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Wetlands Research Program

Regional Guidebook for Applying the Hydrogeomorphic Approach to Assessing Wetland Functions of Northwest Gulf of Mexico Tidal Fringe Wetlands

Deborah J. Shafer, Bryan Herczeg, Daniel W. Moulton, Andrew Sipocz, Kenny Jaynes, Lawrence P. Rozas, Christopher P. Onuf, and Wes Miller

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Assessing Wetland Functions

Regional Guidebook for Applying the Hydrogeomorphic Approach to Assessing Wetland Functions of Northwest Gulf of Mexico Tidal Fringe Wetlands (ERDC/EL TR-02-5)

ISSUE: Section 404 of the Clean Water Act directs the U.S. Army Corps of Engineers to administer a regulatory program for permitting the discharge of dredged or fill material in the "waters of the United States." As part of the permit review process, the impact of discharging dredged or fill material on wetland functions must be assessed. On 16 August 1996, a National Action Plan to Implement the Hydrogeomorphic Approach (NAP) for developing Regional Guidebooks to assess wetland functions was published. This report is one of a series of Regional Guidebooks that will be published in accordance with the National Action Plan.

RESEARCH OBJECTIVE: The objective of this research was to develop a Regional Guidebook for assessing the functions of tidal fringe wetlands in the northwest Gulf of Mexico in the context of the Section 404 Regulatory Program.

SUMMARY: The Hydrogeomorphic (HGM) Approach is a collection of concepts and methods for developing functional indices, and subsequently using them to assess the capacity of a wetland to perform functions relative to similar wetlands in a region. The approach was initially designed to be used in the context of the Clean Water Act Section 404 Regulatory Program permit review sequence to consider alternatives, minimize impacts, assess unavoidable project impacts, determine mitigation requirements, and monitor the success of mitigation projects. However, a variety of other potential applications for the approach have been identified, including determining minimal effects under the Food Security Act, designing mitigation projects, and managing wetlands.

This report uses the HGM Approach to develop a Regional Guidebook for assessing the functions of selected wetland subclasses in the tidal fringe wetlands.

AVAILABILITY OF REPORT: The report is available at the following Web site: *http://www.wes.army.mil/el/wetlands/wlpubs.html.* The report is also available on Interlibrary Loan Service from the U.S. Army Engineer Research and Development Center (ERDC) Research Library, telephone (601) 634-2355, or the following Web site: *http://libweb.wes.army.mil/index.htm.*

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Preface

This Regional Guidebook was authorized by Headquarters, U.S. Army Corps of Engineers (HQUSACE), as part of the Characterization and Restoration of Wetlands Research Program (CRWRP). It is published as an Operational Draft for field testing for a 2-year period. Comments should be submitted via the Internet at the following address: *http://www.wes.army.mil/el/wetlands/hgmhp.html.* Written comments should be addressed to Department of the Army, Research and Development Center, CEERD-EE-W, 3909 Halls Ferry Road, Vicksburg, MS 39180-6199.

The work was performed under Work Unit 32985, "Technical Development of HGM," for which Dr. Ellis J. Clairain, Jr., Environmental Laboratory (EL), Vicksburg, MS, U.S. Army Engineer Research and Development Center (ERDC), was the Principal Investigator. Mr. Dave Mathis, CERD-C, was the CRWRP Coordinator at the Directorate of Research and Development, HQUSACE; Ms. Colleen Charles, CECW-OR, served as the CRWRP Technical Monitor's Representative; Dr. Russell F. Theriot, EL, Vicksburg, MS, ERDC, was the CRWRP Program Manager; and Dr. Clairain was the Task Area Manager.

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At the time of publication of this report, Dr. James R. Houston was Director of ERDC, and COL John W. Morris III, EN, was Commander and Executive Director.

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1 Introduction

The Hydrogeomorphic (HGM) Approach is a collection of concepts and methods for developing functional indices, and subsequently using them to assess the capacity of a wetland to perform functions relative to similar wetlands in a region. The approach was initially designed to be used in the context of the Clean Water Act Section 404 Regulatory Program permit review sequence to consider alternatives, minimize impacts, assess unavoidable project impacts, determine mitigation requirements, and monitor the success of mitigation projects. However, a variety of other potential applications for the approach have been identified, including determining minimal effects under the Food Security Act, designing mitigation projects, and managing wetlands.

On 16 August 1996 a National Action Plan to Implement the Hydrogeomorphic Approach (NAP) was published (National Interagency Implementation Team 1996). The NAP was developed cooperatively by the U.S. Army Corps of Engineers (USACE), U.S. Environmental Protection Agency (USEPA), Natural Resources Conservation Service (NRCS), Federal Highways Administration (FHWA), and U.S. Fish and Wildlife Service (USFWS). Publication of the NAP was designed to outline a strategy and promote the development of Regional Guidebooks for assessing the functions of regional wetland subclasses using the HGM Approach, to solicit the cooperation and participation of Federal, State, and local agencies, academia, and the private sector in this effort, and to update the status of Regional Guidebook development.

The sequence of tasks necessary to develop a Regional Guidebook was outlined in the NAP and was used to develop this Regional Guidebook (see Development Phase). The National Guidebook for Application of Hydrogeomorphic Assessment to Tidal Fringe Wetlands (Shafer and Yozzo 1998) served as the starting point for an initial workshop held at Galveston, TX on 21-24 May 1996. The workshop was attended by fisheries biologists, soil scientists, wildlife biologists, and plant ecologists with extensive knowledge of tidal fringe wetlands of the Texas coast. Based on the results of the workshop, a regional wetland subclass was defined and characterized, a reference domain was defined, wetland functions were selected, model variables were identified, and conceptual assessment models were developed. Subsequently, field work was conducted to collect data from reference wetlands. These data were used to revise and calibrate the conceptual assessment models.

The objectives of this Regional Guidebook are to: (a) characterize the tidal fringe wetlands of the Northwest Gulf of Mexico reference domain, (b) provide a rationale to select functions for the regional tidal fringe subclass, (c) provide a rationale to select model variables and metrics, (d) provide a rationale to develop assessment models, (e) provide data from reference wetlands and document their use in calibrating model variables and assessment models, and (f) outline the necessary protocols for applying the functional indices to the assessment of wetland functions.

This document is organized in the following manner. Chapter 2 outlines the background, objectives, and organization of the document, and provides a brief overview of the major components of the HGM Approach and the Development and Application Phases required to implement the approach. Chapter 3 characterizes the Tidal Fringe Subclass in the northwestern Gulf of Mexico in terms of geographical extent, climate, geomorphic setting, hydrology, vegetation, soils, and other factors that influence wetland function. Chapter 4 discusses each of the wetland functions, model variables, and functional indices. This discussion includes a definition of the function, a quantitative, independent measure of the function for the purposes of validation, a description of the wetland ecosystem and landscape characteristics that influence the function, a definition and description of model variables used to represent these characteristics in the assessment model, a discussion of the assessment model used to derive the functional index, and an explanation of the rationale used to calibrate the index with reference wetland data. Chapter 5 outlines the steps of the assessment protocol for conducting a functional assessment of tidal fringe wetlands in the northwestern Gulf of Mexico. Appendix A lists the team members involved in the development of this HGM assessment protocol. Appendix B provides copies of the field forms needed to collect field data. Appendix C summarizes functions, assessment models, variables, and variable measures, and Appendix D contains the data collected at reference wetlands. Appendix E provides expanded discussions on how to measure selected assessment variables.

While it is possible to assess the functions of tidal fringe wetlands in the northwestern Gulf of Mexico using only the information contained in Chapter 5 and Appendix D, it is suggested that potential users familiarize themselves with the information in Chapters 1-4 prior to conducting an assessment.

2 Overview of the Hydrogeomorphic Approach

As indicated in Chapter 1, the Hydrogeomorphic (HGM) Approach is a collection of concepts and methods for developing functional indices, and subsequently using them to assess the capacity of a wetland to perform functions relative to similar wetlands in a region. The approach was initially designed to be used in the context of the Clean Water Act Section 404 Regulatory Program permit review sequence to consider alternatives, minimize impacts, assess unavoidable project impacts, determine mitigation requirements, and monitor the success of mitigation projects. However, a variety of other potential applications for the approach have been identified, including determining minimal effects under the Food Security Act, designing mitigation projects, and managing wetlands.

The HGM Approach includes four integral components: (a) the HGM Classification, (b) reference wetlands, (c) assessment models and functional indices, and (d) assessment protocols. During the Development Phase of the HGM Approach, these four components are integrated in a Regional Guidebook for assessing the functions of a regional wetland subclass. Subsequently, during the Application Phase, end users, following the assessment protocols outlined in the Regional Guidebook, assess the functional capacity of selected wetlands. Each of the components of the HGM Approach and the Development and Application Phases are discussed below. More extensive treatment of these topics can be found in Brinson (1993, 1995), Brinson et al. (1995, 1996, 1998), Smith et al. (1995), and Hauer and Smith (1998).

The advantage of the HGM approach is that a given site may be assessed for its entire suite of functions or a subset of functions, depending upon the ultimate management objective. The HGM approach requires basic information on the site that can be generated without significant expense. Knowledge about the relationships between form and function upon which these models are based can also be used to assist with planning habitat restoration and/or creation efforts and would allow for the emphasis to be placed on the entire suite of functions or selected functions.

Hydrogeomorphic Classification

Wetland ecosystems share a number of common attributes including relatively long periods of inundation or saturation, hydrophytic vegetation, and hydric soils. In spite of these common attributes, wetlands occur under a wide range of climatic, geologic, and physiographic situations and exhibit a wide range of physical, chemical, and biological characteristics and processes (Cowardin et al. 1979; Mitsch and Gosselink 1993). The variability of wetlands makes it challenging to develop assessment methods that are both accurate (i.e., sensitive to significant changes in function) and practical (i.e., can be completed in the relatively short time frame available for conducting assessments). Existing "generic" methods, designed to assess multiple wetland types throughout the United States, are relatively rapid, but lack the resolution necessary to detect significant changes in function. However, one way to achieve an appropriate level of resolution within the available time frame is to reduce the level of variability exhibited by the wetlands being considered (Smith et al. 1995).

The HGM Classification was developed specifically to accomplish this task (Brinson 1993). It identifies groups of wetlands that function similarly using three criteria that fundamentally influence how wetlands function. These criteria are geomorphic setting, water source, and hydrodynamics. Geomorphic setting refers to the landform and position of the wetland in the landscape. Water source refers to the primary water source in the wetland such as precipitation, overbank floodwater, or groundwater. Hydrodynamics refers to the level of energy and the direction that water moves in the wetland. Based on these three criteria, any number of "functional" wetland groups can be identified at different spatial or temporal scales. For example, at a continental scale, Brinson (1993) identified five hydrogeomorphic wetland classes. These were later expanded to the seven classes described in Table 1 (Smith et al. 1995). In many cases, the level of variability in wetlands encompassed by a continental scale hydrogeomorphic class is still too great to develop assessment models that can be rapidly applied while being sensitive enough to detect changes in function at a level of resolution appropriate to the 404 review process. For example, at a continental geographic scale the depression class includes wetlands as diverse as California vernal pools, prairie potholes in North and South Dakota, playa lakes in the high plains of Texas, kettles in New England, and cypress domes in Florida.

To reduce both inter- and intra-regional variability the three classification criteria are applied at a smaller, regional geographic scale to identify regional wetland subclasses. In many parts of the country, existing wetland classifications can serve as a starting point for identifying these regional subclasses (Stewart and Kantrud 1971; Golet and Larson 1974; Stout 1984). Regional subclasses, like the continental classes, are distinguished on the basis of geomorphic setting, water source, and hydrodynamics. In addition, certain ecosystem or landscape characteristics may also be useful for distinguishing regional subclasses in certain regions. For example, depression subclasses might be based on water source (i.e., groundwater versus surface water) or the degree of connection between the wetland and other surface waters (i.e., the flow of surface water in or out of the depression through defined channels). Tidal fringe subclasses might be based on salinity gradients (Shafer and Yozzo 1998). Slope subclasses might be based on the degree of slope, landscape position, source of water (i.e., interflow versus groundwater), or other factors. Riverine subclasses might be based on water source, position in the watershed, stream order, watershed size, channel gradient, or floodplain width. Examples of potential regional subclasses are shown in Table 2, Smith et al. (1995), and Rheinhardt, Brinson, and Farley (1997).

Table 1
Hydrogeomorphic Wetland Classes at the Continental Scale

HGM Wetland Class	Definition
Depression	Depression wetlands occur in topographic depressions (i.e., closed elevation contours) that allow the accumulation of surface water. Depression wetlands may have any combination of inlets and outlets or lack them completely. Potential water sources are precipitation, overland flow, streams, or groundwater/interflow from adjacent uplands. The predominant direction of flow is from the higher elevations toward the center of the depression. The predominant hydrodynamics are vertical fluctuations that range from diurnal to seasonal. Depression wetlands may lose water through evapotranspiration, intermittent or perennial outlets, or recharge to groundwater. Prairie potholes, playa lakes, vernal pools, and cypress domes are common examples of depression wetlands.
Tidal Fringe	Tidal fringe wetlands occur along coasts and estuaries and are under the influence of sea level. They intergrade landward with riverine wetlands where tidal current diminishes and river flow becomes the dominant water source. Additional water sources may be groundwater discharge and precipitation. The interface between the tidal fringe and riverine classes is where bidirectional flows from tides dominate over unidirectional flows controlled by floodplain slope of riverine wetlands. Because tidal fringe wetlands frequently flood and water table elevations are controlled mainly by sea surface elevation, tidal fringe wetlands seldom dry for significant periods. Tidal fringe wetlands lose water by tidal exchange, by overland flow to tidal creek channels, and by evapotranspiration. Organic matter normally accumulates in higher elevation marsh areas where flooding is less frequent and the wetlands are isolated from shoreline wave erosion by intervening areas of low marsh. *Spartina alterniflora* salt marshes are a common example of tidal fringe wetlands.
Lacustrine Fringe	Lacustrine fringe wetlands are adjacent to lakes where the water elevation of the lake maintains the water table in the wetland. In some cases, these wetlands consist of a floating mat attached to land. Additional sources of water are precipitation and groundwater discharge, the latter dominating where lacustrine fringe wetlands intergrade with uplands or slope wetlands. Surface water flow is bidirectional, usually controlled by water-level fluctuations resulting from wind or seiche. Lacustrine wetlands lose water by flow returning to the lake after flooding and evapotranspiration. Organic matter may accumulate in areas sufficiently protected from shoreline wave erosion. Unimpounded marshes bordering the Great Lakes are an example of lacustrine fringe wetlands.
Slope	Slope wetlands occur in association with the discharge of groundwater to the land surface or sites with saturated overland flow with no channel formation. They normally occur on sloping land ranging from slight to steep. The predominant source of water is groundwater or interflow discharging at the land surface. Precipitation is often a secondary contributing source of water. Hydrodynamics are dominated by downslope unidirectional water flow. Slope wetlands can occur in nearly flat landscapes if groundwater discharge is a dominant source to the wetland surface. Slope wetlands lose water primarily by saturated subsurface flows, surface flows, and by evapotranspiration. Slope wetlands may develop channels, but the channels serve only to convey water away from the slope wetland. Slope wetlands are distinguished from depression wetlands by the lack of a closed topographic depression and the predominance of the groundwater/interflow water source. Fens are a common example of slope wetlands.

(Continued)

Table 1 (Concluded)	
HGM Wetland Class	**Definition**
Mineral Soil Flats	Mineral soil flats are most common on interfluves, extensive relic lake bottoms, or large floodplain terraces where the main source of water is precipitation. They receive virtually no groundwater discharge, which distinguishes them from depressions and slopes. Dominant hydrodynamics are vertical fluctuations. Mineral soil flats lose water by evapotranspiration, overland flow, and seepage to underlying groundwater. They are distinguished from flat upland areas by their poor vertical drainage due to impermeable layers (e.g., hardpans), slow lateral drainage, and low hydraulic gradients. Mineral soil flats that accumulate peat can eventually become organic soil flats. They typically occur in relatively humid climates. Pine flatwoods with hydric soils are an example of mineral soil flat wetlands.
Organic Soil Flats	Organic soil flats, or extensive peatlands, differ from mineral soil flats in part because their elevation and topography are controlled by vertical accretion of organic matter. They occur commonly on flat interfluves, but may also be located where depressions have become filled with peat to form a relatively large flat surface. Water source is dominated by precipitation, while water loss is by overland flow and seepage to underlying groundwater. They occur in relatively humid climates. Raised bogs share many of these characteristics but may be considered a separate class because of their convex upward form and distinct edaphic conditions for plants. Portions of the Everglades and northern Minnesota peatlands are examples of organic soil flat wetlands.
Riverine	Riverine wetlands occur in floodplains and riparian corridors in association with stream channels. Dominant water sources are overbank flow from the channel or subsurface hydraulic connections between the stream channel and wetlands. Additional sources may be interflow, overland flow from adjacent uplands, tributary inflow, and precipitation. When overbank flow occurs, surface flows down the floodplain may dominate hydrodynamics. In headwaters, riverine wetlands often intergrade with slope, depressional, poorly drained flat wetlands, or uplands as the channel (bed) and bank disappear. Perennial flow is not required. Riverine wetlands lose surface water via the return of floodwater to the channel after flooding and through surface flow to the channel during rainfall events. They lose subsurface water by discharge to the channel, movement to deeper groundwater (for losing streams), and evapotranspiration. Peat may accumulate in off-channel depressions (oxbows) that have become isolated from riverine processes and subjected to long periods of saturation from groundwater sources. Bottomland hardwoods on floodplains are an example of riverine wetlands.

Table 2
Potential Regional Wetland Subclasses in Relation to Geomorphic Setting, Dominant Water Source, and Hydrodynamics

			Potential Regional Wetland Subclasses	
Geomorphic Setting	**Dominant Water Source**	**Dominant Hydrodynamics**	**Eastern USA**	**Western USA/Alaska**
Depression	Groundwater or interflow	Vertical	Prairie pothole marshes, Carolina bays	California vernal pools
Fringe (tidal)	Ocean or estuary	Bidirectional, horizontal	Chesapeake Bay and Gulf of Mexico tidal marshes	San Francisco Bay marshes
Fringe (lacustrine)	Lake	Bidirectional, horizontal	Great Lakes marshes	Flathead Lake marshes
Slope	Groundwater	Unidirectional, horizontal	Fens	Avalanche chutes
Flat (mineral soil)	Precipitation	Vertical	Wet pine flatwoods	Large playas
Flat (organic soil)	Precipitation	Vertical	Peat bogs; portions of Everglades	Peatlands over permafrost
Riverine	Overbank flow from channels	Unidirectional, horizontal	Bottomland hardwood forests	Riparian wetlands

Regional Guidebooks include a thorough characterization of the regional wetland subclass in terms of its geomorphic setting, water sources, hydrodynamics, vegetation, soil, and other features that were taken into consideration during the classification process.

Reference Wetlands

Reference wetlands are the wetland sites selected to represent the range of variability that occurs in a regional wetland subclass as a result of natural processes and disturbance (e.g., succession, channel migration, fire, erosion, and sedimentation) as well as cultural alteration. The reference domain is the geographic area occupied by the reference wetlands (Smith et al. 1995). Ideally, the geographic extent of the reference domain will mirror the geographic area encompassed by the regional wetland subclass; however, this is not always possible due to time and resource constraints.

Reference wetlands serve several purposes. First, they establish a basis for defining what constitutes a characteristic and sustainable level of function across the suite of functions selected for a regional wetland subclass. Second, they establish the range and variability of conditions exhibited by model variables and provide the data necessary for calibrating model variables and assessment models. Finally, they provide a concrete physical representation of wetland ecosystems that can be repeatedly observed and measured.

Reference standard wetlands are the subset of reference wetlands that perform the suite of functions selected for the regional subclass at a level that is characteristic in the least-altered wetland sites in the least-altered landscapes. Table 3 outlines the terms used by the HGM Approach in the context of reference wetlands.

Assessment Models and Functional Indices

In the HGM Approach, an assessment model is a simple representation of a function performed by a wetland ecosystem. It defines the relationship between one or more characteristics or processes of the wetland ecosystem or surrounding landscape and the functional capacity of a wetland ecosystem. Functional capacity is simply the ability of a wetland to perform a function compared to the level of performance in reference standard wetlands.

Model variables represent the characteristics of the wetland ecosystem and surrounding landscape that influence the capacity of a wetland ecosystem to perform a function. Model variables are ecological quantities that consist of five components (Schneider 1994). These include: (a) a name,

Table 3
Reference Wetland Terms and Definitions

Term	Definition
Reference domain	The geographic area from which reference wetlands representing the regional wetland subclass are selected (Smith et al. 1995).
Reference wetlands	A group of wetlands that encompass the known range of variability in the regional wetland subclass resulting from natural processes and disturbance and from human alteration.
Reference standard wetlands	The subset of reference wetlands that perform a representative suite of functions at a level that is both sustainable and characteristic of the least human-altered wetland sites in the least human-altered landscapes. By definition, the functional capacity indices for all functions in reference standard wetlands are assigned a 1.0.
Reference standard wetland variable condition	The range of conditions exhibited by model variables in reference standard wetlands. By definition, reference standard conditions receive a variable subindex score of 1.0.
Site potential	The highest level of function possible, given local constraints of disturbance history, land use, or other factors. Site potential may be less than or equal to the levels of function in reference standard wetlands of the regional wetland subclass.
Project target (mitigation project context)	The level of function identified or negotiated for a restoration or creation project.
Project standards (mitigation context)	Performance criteria and/or specifications used to guide the restoration or creation activities toward the project target. Project standards should specify reasonable contingency measures if the project target is not being achieved.

(b) a symbol, (c) a measure of the variable and procedural statement for quantifying or qualifying the measure directly or calculating it from other measurements, (d) a set of values (i.e., numbers, categories, or numerical estimates) that are generated by applying the procedural statement, and (e) units on the appropriate measurement scale. Table 4 provides several examples.

Model variables occur in a variety of states or conditions in reference wetlands. The state or condition of the variable is denoted by the value of the measure of the variable. For example, tree basal area, the measure of the tree biomass variable, could be large or small. Similarly, recurrence interval, the measure of overbank flood frequency variable, could be frequent or infrequent. Based on their condition (i.e., value of the metric), model variables are assigned a variable subindex. When the condition of a variable is within the range of conditions exhibited by reference standard wetlands, a variable subindex of 1.0 is assigned. As the condition deviates from the reference standard condition (i.e., the range of conditions that the variable occurs in reference standard wetlands), the variable subindex is assigned based on the defined relationship between model variable condition and functional capacity. As the condition of a variable deviates from the conditions exhibited in reference standard wetlands, it receives a progressively lower subindex reflecting its decreasing contribution to functional capacity. In some cases, the variable subindex drops to zero. For example, when no trees are present, the subindex for tree basal area is zero. In other cases, the subindex for a variable never drops to zero. For example,

Table 4
Components of a Model Variable

Name (Symbol)	Measure/Procedural Statement	Resulting Values	Units (Scale)
Redoximorphic Features (V_{REDOX})	Status of redoximorphic features/ visual inspection of soil profile for redoximorphic features	Present Absent	Unitless (nominal scale)
Floodplain Roughness (V_{ROUGH})	Manning's Roughness Coefficient (*n*)/ observe wetland characteristics to determine adjustment values for roughness component to add to base value	0.01 0.1 0.21	Unitless (interval scale)
Tree Biomass (V_{TBA})	Tree basal area/measure diameter of trees in sample plots (cm), convert to area (m^2), and extrapolate to per-hectare basis	5 12/8 36	m^2/ha (ratio scale)

regardless of the condition of a site, Manning's Roughness Coefficient (*n*) will always be greater than zero.

Model variables are combined in an assessment model to produce a Functional Capacity Index (FCI) that ranges from 0.0-1.0. The FCI is a measure of the functional capacity of a wetland relative to reference standard wetlands in the reference domain. Wetlands with an FCI of 1.0 perform the function at a level that is sustainable and characteristic of reference standard wetlands. A decrease in FCI usually indicates that the capacity of the wetland to perform the function is less than that which is characteristic of reference standard wetlands. In some cases, however, higher levels of function may occur under conditions that are considered unsustainable or atypical of reference standard wetlands. For example, high values for V_{EDGE}, the variable that measures the ratio of marsh-water interface to total marsh area, are considered to indicate reference standard conditions and would be assigned a variable subindex of 1.0. Very high values for V_{EDGE} are characteristic of subsiding or drowning marshes, an unsustainable condition, and would therefore be assigned a subindex value less than 1.0, even though these marshes may be highly productive in terms of fisheries utilization over the short term.

Assessment Protocol

The final component of the HGM Approach is the assessment protocol. The assessment protocol is a series of tasks, along with specific instructions, that allow the end user to assess the functions of a particular wetland area using the functional indices in the Regional Guidebook. The first task is characterization, which involves describing the wetland ecosystem and the surrounding landscape, describing the proposed project and its potential impacts, and identifying the wetland areas to be assessed. The second task is collecting the field data for model variables. The final task is analysis, which involves calculation of functional indices.

Development Phase

The Development Phase of the HGM Approach is ideally carried out by an interdisciplinary team of experts known as the "Assessment Team," or "A-Team." The product of the Development Phase is a Regional Guidebook for assessing the functions of a specific regional wetland subclass (Figure 1). In developing a Regional Guidebook, the A-Team will complete the following major tasks. After organization and training, the first task of the A-Team is to classify the wetlands within the region of interest into regional wetland subclasses using the principles and criteria of the Hydrogeomorphic Classification (Brinson 1993, Smith et al. 1995). Next, focusing on the specific regional wetland subclass selected, the A-Team develops an ecological characterization or functional profile of the subclass.

The A-Team then identifies the important wetland functions, conceptualizes assessment models, identifies model variables to represent the characteristics and processes that influence each function, and defines metrics for quantifying model variables. Next, reference wetlands are identified to represent the range of variability exhibited by the regional subclass. Field data are then collected from the reference wetlands and used to calibrate model variables and verify the conceptual assessment models. Finally, the A-Team develops the assessment protocols necessary for regulators, managers, consultants, and other end users to apply the indices to the assessment of wetland functions. The following list provides the detailed steps involved in the general sequence described above.

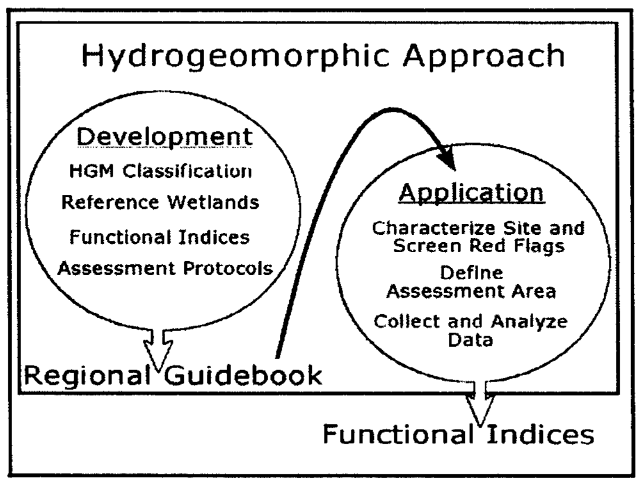

Figure 1. Development and application phases of the HGM Approach

Task 1: Organize the A-Team
- A. Identify A-Team members
- B. Train A-Team in the HGM Approach

Task 2: Select and Characterize Regional Wetland Subclass
- A. Identify/prioritize regional wetland subclasses

B. Select regional wetland subclass and define reference domain
C. Initiate literature review
D. Develop preliminary characterization of regional wetland subclass
E. Identify and define wetland functions

Task 3: Select Model Variables and Metrics and Construct Conceptual Assessment Models
A. Review existing assessment models
B. Identify model variables and metrics
C. Define initial relationship between model variables and functional capacity
D. Construct conceptual assessment models for deriving functional capacity indices (FCI)
E. Complete Precalibrated Draft Regional Guidebook (PDRG)

Task 4: Conduct Peer Review of Precalibrated Draft Regional Guidebook
A. Distribute PDRG to peer reviewers
B. Conduct interdisciplinary, interagency workshop of PDRG
C. Revise PDRG to reflect peer review recommendations
D. Distribute revised PDRG to peer reviewers for comment
E. Incorporate final comments from peer reviewers on revisions into the PDRG

Task 5: Identify and Collect Data from Reference Wetlands
A. Identify reference wetland field sites
B. Collect data from reference wetland field sites
C. Analyze reference wetland data

Task 6: Calibrate and Field Test Assessment Models
A. Calibrate model variables using reference wetland data
B. Verify and validate (optional) assessment models
C. Field test assessment models for repeatability and accuracy
D. Revise PDRG based on calibration, verification, validation optional), and field testing results into a Calibrated Draft Regional Guidebook (CDRG)

Task 7: Conduct Peer Review and Field Test of Calibrated Draft Regional Guidebook
A. Distribute CDRG to peer reviewers
B. Field test CDRG
C. Revise CDRG to reflect peer review and field test recommendations
D. Distribute CDRG to peer reviewers for final comment on revisions
E. Incorporate peer reviewers' final comments on revisions
F. Publish Operational Draft Regional Guidebook (ODRG)

Task 8: Technology Transfer
A. Train end users in the use of the ODRG
B. Provide continuing technical assistance to end users of the ODRG

Application Phase

The Application Phase involves two steps. The first is using the assessment protocols outlined in the Regional Guidebook to carry out the following tasks (Figure 1).

a. Define assessment objectives.
b. Characterize the project site.
c. Screen for red flags.
d. Define the Wetland Assessment Area.
e. Collect field data.
f. Analyze field data.

The second step involves applying the results of the assessment, the FCI, to the appropriate decision-making processes of the permit review sequence, such as alternatives analysis, minimization, assessment of unavoidable impacts, determination of compensatory mitigation, design and monitoring of mitigation, comparison of wetland management alternatives or results, determination of restoration potential, or identification of acquisition or mitigation sites.

Examples of Practical Applications

Prior to completion of the first regional guidebooks, the HGM Approach principles were used to aid in defining appropriate mitigation requirements on several complex permit applications. During a review of a permit application for a large residential housing complex in south central Florida, these principles were used to more clearly quantify the mitigation required, leading to the completion of an extended permit application that had been stalled for several years due to the lack of identifiable and quantifiable mitigation requirements. A similar situation occurred in the Louisville District where the HGM principles were used to clearly identify and describe characteristics that influence wetland functions at a large surface mining site. Changes in these characteristics as a result of project implementation were then identified and used to determine appropriate mitigation requirements, facilitating completion of a complex permit application and minimizing project impacts.

Following completion of several regional guidebooks, Corps of Engineers Districts and other Federal agencies have used them to assess project impacts and mitigation requirements. The Corps of Engineers Louisville District has been using the regional guidebook for assessment of riverine wetlands in western Kentucky to assess impacts of permitted projects since it was published in 1999. The National Resources Conservation Service (NRCS) has used the first draft of the prairie pothole regional guidebook to determine minimum impacts as part of the Food Security Act; the Corps of Engineers Omaha District has also used this guidebook for wetland assessment. The Corps of Engineers Alaska District has signed a Memorandum of Understanding (MOU) with several other Federal and state agencies to use regional guidebooks as they become available; three regional guidebooks for Alaska wetlands are currently nearing completion and should be in use by the time this document is published. The Mobile District will be using the regional guidebook on wet pine flatwoods in the southeastern United States to assess mitigation banking credits. The states of Mississippi and Alabama are working with the Mobile District and ERDC to develop additional regional guidebooks to aid in wetland assessment and determination of mitigation banking credits.

3 Characterization of Tidal Fringe Wetlands in the Northwestern Gulf of Mexico Regional Wetland Subclass

Introduction

This Regional Guidebook was developed to assess the functions of tidal fringe wetlands in the northwestern Gulf of Mexico. For the purposes of this approach, the term "tidal fringe wetlands" applies only to vegetated habitats occupying the intertidal zone of marine, estuarine, or riverine systems. Specifically, these wetlands occur along the fringe of drowned river valleys, barrier islands, lagoons, and other coastal waterways, receive their water primarily from marine or estuarine sources, and are affected by astronomical tidal action. Included in this group are wetlands commonly known as intertidal marshes, salt marshes, forested riverine swamps, and mangrove swamps and correspond to the emergent, scrub-shrub, and forested wetland class designations used by Cowardin et al. (1979). The dominant hydrodynamic is bidirectional water flow generated by tidal action. Additional water sources may be riverine flow, groundwater discharge, and precipitation. Tidal fringe wetlands lose water by tidal exchange, by saturated overland flow to tidal creek channels, and by evapotranspiration. Organic matter normally accumulates in higher elevation marsh areas where flooding is less frequent and the wetlands are protected from shoreline wave erosion by intervening areas of low marsh. *Spartina alterniflora* salt marshes are a common example of tidal fringe wetlands.

By definition (Cowardin et al. 1979), the entire intertidal zone includes the vertical range between the extreme annual high- and low-water levels of spring tides. Spring tides are tides of greater-than-average range that occur around the times of new and full moon. However, the northwestern Gulf of Mexico has small (<1 m) to microtidal (<0.5 m in south Texas) meteorologically dominated (wind) mixed tides. Therefore, wind tides as well as astronomical tides must be considered in relation to the classification of Texas tidal-fringe wetlands. Along coastal rivers, tidal wetlands extend horizontally to the upstream limits of tidal influence and may or may not be exposed to fluctuating salinity (e.g., tidal swamps and freshwater marshes).

Early in the development of these models, regional tidal fringe wetland subclasses were proposed based on differences in elevation and salinity (Shafer and Yozzo 1998). Low marshes occupy the vertical range between the extreme low water levels of spring tides up to the mean daily high water zone and are frequently flooded due to low elevation and proximity to open water. High marshes occupy the vertical range between the mean daily high water zone up to the mean annual high water zone of spring high tides and are infrequently flooded due to higher elevation and distal location from open bay/estuary waters. In the field, differences between these proposed subclasses were often difficult to identify and delineate. Usually, except where abrupt elevation changes occur, distinct boundaries do not exist and transitional zones may show only subtle changes in plant assemblages as one moves higher in elevation and away from sources of salt water. Also, topographic irregularity (mounds, ridges, swales, depressions, etc.) and altered hydrologic characteristics (levees, roads, ditches, drains, water-control structures) often result in uncharacteristically diverse or somewhat atypical plant assemblages. Similarly, since many tidal marsh plant species are tolerant of salinities ranging from saline to fresh, separation of wetland subclasses along a salinity gradient based on vegetation characteristics is not always reliable. Since many of the model variables focus on geomorphological or landscape characteristics, separation of wetland subclasses based on elevation and salinity did not seem justified.

According to Smith et al. (1995), the reference domain is the geographic area occupied by the reference wetland sites. Based on differences in climate, rainfall, and amount of freshwater input, the A-team chose to subdivide the Texas coast into three sub-regions (Figure 2). The Mid-Coast region includes the Trinity-San Jacinto (Galveston Bay), Brazos-San Bernard, Lavaca-Colorado (Lavaca and Matagorda Bays), and Guadalupe-San Antonio (San Antonio Bay) estuaries. The Coastal Bend region includes the Mission-Aransas (Copano and Aransas Bays), and Nueces (Nueces and Corpus Christi Bays) estuaries. The Laguna Madre estuary constitutes the Lower Coast sub-region. Although the potential reference domain includes those tidally influenced coastal wetlands along the Texas coast from Galveston Bay westward to the Mexican border, most of the reference data were collected within the Galveston Bay estuary, with a few other data collection sites located in Matagorda Bay, Mesquite Bay, and Corpus Christi Bay (Appendix D, Table D1). Until additional reference data can be collected from sites along the lower coast, initial application of this method should be limited to tidal fringe marshes in the vicinity of the Galveston Bay estuary or Mid-Coast region of Texas.

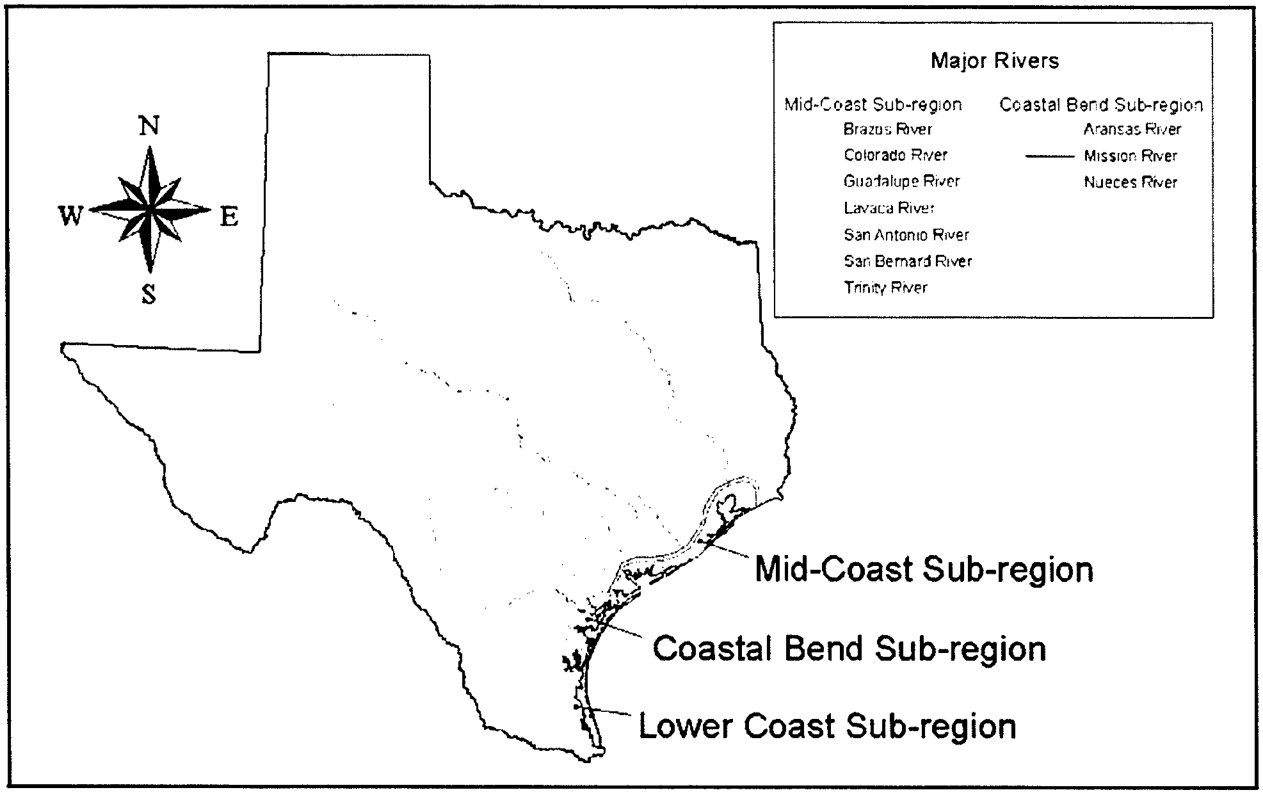

Figure 2. Potential reference domain

Description of the Regional Subclass

Geology and geomorphology

Approximately 60 to 100 million years ago, the entire region that would later become the Texas coastal plain was beneath the waters of the Gulf of Mexico. Sediments eroded from the Rocky Mountains and other continental highlands extended the edge of the continent some 400 km into the Gulf, depositing gravel, sand, silt, and clay in layers up to 12,000 m thick. The modern Texas coastal landscape developed as the result of repeated changes in sea level during the Ice Ages of the Pleistocene Epoch. About 135,000 years ago, during a warm inter-glacial period, sea level was about 8 m higher than present and the Gulf shore was 20 miles inland of its current position. Much of the soil of the coastal plain was deposited by rivers on floodplains and deltas inland of the present coastline. During the maximum glaciation of the last Ice Age (about 18,000 years ago), sea level dropped 30 to 45 m below present level and the coastline was perhaps 50 or more miles seaward of today's coast. During that period, coastal rivers eroded deep valleys down through the recently exposed soft coastal plain sediments. Since then, sea level has risen due to climate warming, reaching the present level about 5,000 years ago. The rising seawater flooded the deeply eroded coastal river valleys, forming bays and lagoons along the new coastline. Coastal rivers adjusted to the new sea level by filling in their deep valleys. The largest Texas rivers, the Brazos, Colorado, and Rio Grande, carried enough sediment to fill their once extensive bays and form deltaic plains at the river mouths. Over the last 9,000 years, some of the bays that have not yet filled in have received over 24 m of sediment. These sediments formed the soils that support the fringe of tidal wetlands around the bay shores of the upper and mid-coasts.

As longshore currents reworked those sediment deposits, several sets of barrier islands have formed and eroded along the coast over the last 1 million years as successive rises and falls of sea level caused shifts in the position of the coastline. Remnants of an ancient chain of barrier islands, formed when sea level was higher than today, can still be seen along parts of the mainland shore. As world climate warmed from 10,000 to 5,000 years ago, sea level rose to within 5 m of the present sea level. As the continental shelf was again submerged, sand that had been deposited in now-submerged river channels, river mouths, and along ancient shorelines was reworked towards shore by tides, waves, storms, and longshore currents to form the modern barrier islands.

Along the lower coast, from south of Kingsville to the Rio Grande, prevailing southeasterly winds have blown sand from the ancient barrier dunes inland, creating the Coastal Sand Plain. The semiarid climate has resulted in sparse vegetative cover. In areas where the groundwater is near the surface, wetlands occur in the swales between dunes. The Laguna Madre is filling with wind-blown sand from Padre Island; the central Laguna, called the Land-Cut, filled in with sand about 150 years ago. The Land-Cut region divides the upper and lower Laguna Madre and is usually dry except when strong winds cover the broad wind-tidal flat with very shallow water. Along the shore of the hyperhaline Laguna Madre, a lack of rainfall or other fresh water results in the formation of mostly unvegetated wind-tidal salt flats. These flats replace the emergent marshes of the more northern, less arid bay shores.

Climate

Tidal fringe wetlands occurring along the Texas coast are influenced by local hydrological and climatic conditions such as freshwater input from riverine sources, rainfall, and evapotranspiration rates. Moving southward from Galveston Bay to the Rio Grande, there is a decrease in rainfall and an increase in average temperatures and rates of evapotranspiration. The average annual rainfall ranges from about 54 in. along the upper coast near Beaumont to 26 in. near South Padre Island (White et al. 1985). From Galveston Bay to Corpus Christi Bay, major rivers supply freshwater inflow to large bays (except the Brazos River and San Bernard River estuary) that have major Gulf inlets between the barrier islands or peninsulas. The upper and lower Laguna Madre, on the other hand, have no major source of freshwater inflow (low rainfall and no major drainages between the Nueces River and Rio Grande) and restricted Gulf inlets. With little freshwater input, evapotranspiration generally exceeds precipitation south of the Bay City-Freeport area (White et al. 1985). Therefore, the Laguna Madre is normally hyperhaline. Also, because of restricted Gulf inlets and microtides, tides of the Laguna are more wind-dominated than elsewhere along the Texas coast.

Hydrology

The tidal range in Galveston Bay is generally less than 0.45 m, with an average of 0.3 m (Webb and Dodd 1976). Maximum tidal currents (outside navigation channels) are approximately 0.3 m/sec (Bobb and Boland 1970). In the Galveston Bay area, predominant winds are southeasterly for much of the year (March-November), with occasional strong northerly winds from December through February (White et al. 1985). In some areas, wind-driven tidal circulation may obscure astronomical tidal cycles, making site hydrology less predictable.

Vegetation communities

Low Salt Marsh. Low salt marshes occur at lower elevations and in closer proximity to sources of bay-estuary salt water than high salt marshes. Mid-coast low salt marshes often have as dominants one or more of the following species: *Spartina alterniflora*, *Batis maritima*, *Salicornia virginica*, *Juncus roemerianus*, *Distichlis spicata*, *Suaeda* spp., *Monanthochloe littoralis*, *Scirpus maritimus*, *Lycium carolinianum*, and *Aster* spp. Avicennia germinans is found along the mid-coast, primarily in the Guadalupe-San Antonio estuary. Areas of slightly higher elevation within the low salt marsh may support shrubs like *Iva frutescens*, *Borrichia frutescens*, and *Tamarix* spp. *Spartina alterniflora* occurs only in the low marsh assemblage because it is largely restricted to the daily intertidal zone. Coastal Bend low salt marshes may support any of the species listed for mid-coast salt marshes except *Juncus roemerianus*. *Avicennia germinans* is more common than on the mid-coast. Laguna Madre low salt marshes may support any of the species listed for Coastal Bend marshes, but *Spartina alterniflora* is only a minor component of the low salt marsh assemblage. *Avicennia germinans* is more common than along the Coastal Bend.

High Salt Marsh. Mid-coast high salt marshes may support any of the species listed for low salt marsh except *Spartina alterniflora*. *Borrichia*, *Monanthochloe*, *Distichlis*, *Suaeda* spp., *Iva* spp., and *Aster* spp. are likely to be more common than in low salt marsh. *Sporobolus virginicus*, *Machaeranthera phyllocephala*, *Sesuvium portulacastrum*, *Heliotropium curassavicum*, *Limonium carolinianum*, and *Salicornia* spp. may also be present. Species such as *Spartina patens* and *S. spartinae*, more typical of brackish marsh, may also be present. Coastal Bend high salt marshes are similar to mid-coast high salt marshes. Laguna Madre high salt marshes are similar to Coastal Bend high salt marshes. *S. portulacastrum* and *H. curassavicum* are common in high salt marshes on the lower coast.

High salt marsh may grade nearly imperceptibly into vegetated saline flats of sparse *Monanthochloe, Salicornia, Batis,* and *Distichlis.*

Low Brackish Marsh. Brackish marsh assemblages are generally inland of salt marshes. Brackish marshes are transitional between salt marshes and freshwater-influenced environments and therefore support more diverse plant assemblages than salt marshes with species typical of both ends of the salinity scale. The transitional zones that grade from salt to brackish to intermediate to fresh marsh may be dynamic with respect to their width, location, and other environmental parameters.

Mid-coast low brackish marshes may support *Spartina alterniflora, Spartina cynosuroides* (Trinity-San Jacinto estuary), *Scirpus maritimus, Scirpus pungens, Scirpus americanus, Juncus roemerianus, Bacopa monnieri, Aster* spp., *Typha domingensis, Paspalum vaginatum*, and *Fimbristylis castanea.* Species typical of low salt marsh such as *Monanthochloe, Distichlis, Batis, Salicornia,* and *Lycium* may also be present. Coastal Bend low brackish marsh may support most of the species found on the mid-coast except *Scirpus americanus* and *Juncus roemerianus. Scirpus californicus,* more typical of fresher water, may also be present. Laguna Madre low brackish marsh may support *Scirpus maritimus, Scirpus pungens, Paspalum vaginatum, Typha domingensis, Monanthochloe, Batis, Distichlis,* and *Salicornia* spp.

High Brackish Marsh. High brackish marsh in all regions is characterized by *Spartina patens, Spartina spartinae, Borrichia, Aster* spp., *Lycium carolinianum, Paspalum vaginatum, Phragmites australis, Baccharis halimifolia, Iva* spp., *Tamarix* spp., *Limonium carolinianum, Sporobolus virginicus, Solidago sempervirens, Panicum virgatum, Monanthochloe, Distichlis, Batis, Salicornia* spp., *Suaeda* spp., *Heliotropium curassavicum,* and *Sesuvium portulacastrum.* In many areas, the transitional zone between salt and brackish marsh seems to be dominated by *Spartina spartinae, Spartina patens,* and *Borrichia frutescens* (White et al. 1983).

Low Intermediate Marsh. Intermediate marsh generally occurs inland of brackish marsh and may be dominated by species that are more typical of freshwater than brackish-water environments. Low intermediate marsh on the mid-coast might have as dominant species *Scirpus californicus, Scirpus americanus, Scirpus pungens, Alternanthera philoxeroides, Bacopa monnieri, Phragmites australis, Eleocharis* spp., *Typha* spp., *Cyperus articulatus* and other *Cyperus* spp., *Hydrocotyle bonariensis, Paspalum lividum,* and *Cladium jamaicense.* In the Coastal Bend region, *Alternanthera* would drop out as a major component as would *Scirpus americanus* (Olney's bulrush). Along the Laguna Madre, intermediate marsh is uncommon due to a lack of freshwater inflow. Intermediate marsh assemblages may occur inland along tidally influenced rivers.

High Intermediate Marsh. High intermediate marsh supports species such as *Spartina spartinae, Spartina patens, Cyperus* spp., *Hydrocotyle bonariensis, Phragmites australis, Aster spinosus, Centella asiatica,* *Paspalum vaginatum*, *Solidago sempervirens*, *Setaria* spp., *Polygonum* spp., and shrubs or saplings of species like *Borrichia frutescens*, *Tamarix* spp., *Baccharis halimifolia*, *Iva* spp., *Sesbania* spp., *Parkinsonia aculeata*, *Sapium sebiferum*, *Salix nigra*, and perhaps *Acacia smallii*, and *Prosopis glandulosa*.

Freshwater Marsh. Many of the species listed for intermediate marsh also occur in freshwater marshes. Common herbaceous species that indicate freshwater environments include *Ludwigia* spp., *Sagittaria* spp., *Pontederia cordata*, *Paspalum lividum*, *Xyris* spp., *Marsilea* spp., *Hymenocallis* spp., *Kosteletzkya virginica*, *Carex* spp., *Echinodorus* spp., *Polygonum* spp., *Typha latifolia*, *Zizaniopsis miliacea*, *Eichhornia crassipes*, *Lemna* spp.; also the shrubs *Sesbania* spp., *Parkinsonia aculeata*, *Cephalanthus* spp., and *Sapium sebiferum*; and along drainages the trees *Taxodium distichum* (mid-coast), *Planera aquatica* (mid-coast), *Carya aquatica* (mid-coast), *Nyssa aquatica* (mid-coast), *Fraxinus* spp., *Celtis laevigata*, and *Salix nigra*.

Transitional Zones. Zones of transition between higher wetlands and uplands may be vegetated by species typical of both wetlands and uplands and may cover extensive areas. Common species include *Spartina spartinae*, *S. patens*, *Baccharis* spp., *Juncus* spp., *Carex* spp., *Cyperus* spp., *Eleocharis* spp., *Sesbania* spp., *Rhynchospora* spp., *Iva annua*, *Aster spinosus*, *Centella asiatica*, *Paspalum monostachyum*, *Paspalum plicatulum*, *Aristida* spp., *Setaria* spp., *Andropogon glomeratus*, *A. virginicus*, *Schizachyrium scoparium*, *Panicum* spp., *Arundo donax*, *Ambrosia* spp., *Eupatorium* spp., *Pluchea camphorata*, *Sabatia* spp., *Euthamia leptocephala*, *Polygonum* spp., *Phyla* spp., *Coreopsis tinctoria*, *Conoclinium betonicifolium*, *Croton* spp., *Cassia fasciculata*, *Helianthus* spp., *Sorghum halepense*, *Cynodon dactylon*, *Polypogon monspeliensis*, *Flaveria* spp., *Acacia smallii*, *Prosopis glandulosa*, *Parkinsonia aculeata*, *Sapium sebiferum*, and *Tamarix* spp.

Soils

Tidal fringe wetlands in the Texas Gulf Coast Saline Prairie Major Land Resource Area have soils that formed on flood basin, coastal marsh, and beach sediments deposited near bays and the Gulf of Mexico. Surface textures range from sand to clay, and in some soils mucky clay and mucky clay loam also occur. Soil salinity in these landscapes ranges from 0.5 to 18 parts per thousand. In the Texas Gulf Coast Saline Prairie, the high marsh roughly corresponds to elevations of 0.6 to 2.5 m. Soils in the high marsh include the sandy Mustang soil, the loamy Nass, Sievers and Veston soils, and the clayey Surfside soil. The low marsh roughly corresponds to elevations of less than 1 m. Soils in the low marsh include the loamy Follett, Karankawa, Tatlum and Veston soils, and the clayey Harris, Placedo, Tracosa, and Velasco soils.

Anthropogenic alterations

Common anthropogenic alterations affecting this reference domain include oil and gas extraction, impoundment, grazing, and conversion to uplands. These changes may have profound impacts on marsh geomorphology, hydrologic regime, vegetation community, and habitat quality. Extraction of oil, gas, and groundwater has contributed to accelerated rates of subsidence. Flood control structures on many of the major rivers such as the Trinity, San Jacinto, Colorado, and Nueces Rivers have reduced sediment loads into Texas estuaries. As a result, subsidence rates in some tidal fringe wetlands may be compounded by sediment starvation. Deep-draft navigation channels within the reference domain may increase salinity levels within the estuaries. Erosion from vessel wakes results in loss of some tidal fringe wetlands along navigation channels such as the Gulf Intracoastal Waterway. Nutrient loading from agricultural sources and wastewater discharges into watershed rivers may affect species composition and biomass production of tidal fringe marsh plant communities.

4 Wetland Functions and Assessment Models

The following functions performed by Northwest Gulf of Mexico tidal fringe marshes were selected for assessment:

1. Shoreline stabilization.
2. Sediment deposition.
3. Nutrient and organic carbon exchange.
4. Resident nekton utilization.
5. Non-resident nekton utilization.
6. Maintain invertebrate prey pool.
7. Provide wildlife habitat.
8. Maintain characteristic plant community composition.
9. Plant biomass production.

The following sequence is used to present and discuss each of these functions:

Definition: defines the function and identifies an independent quantitative measure that can be used to validate the functional index.

Rationale for selecting the function: provides the rationale for why a function was selected and discusses onsite and offsite effects that may occur as a result of diminished or atypical functional capacity.

Characteristics and processes that influence the function: describes the characteristics and processes of the wetland and the surrounding landscape that influence the function and lay the groundwork for the description of model variables.

Description of model variables: defines and discusses model variables and describes how each model variable is measured.

Functional capacity index: describes the assessment model from which the functional capacity index is derived and discusses how model variables interact to influence functional capacity.

Function 1: Shoreline Stabilization

Definition

Shoreline stabilization is the ability of a wetland to maintain existing shorelines against erosion and subsidence due to relative sea level rise. It is influenced by several factors. Emergent macrophytic vegetation of tidal fringe wetlands baffles wave energy, slowing water movement and allowing suspended material to be deposited on the marsh substrate. The roots help to bind the deposited sediments to the marsh substrate. A quantitative unit of measure of this function would be net hectares of marsh gained or lost/year/km of coastline. This function includes components of both the Wave Energy Attenuation and Shoreline Stabilization functions listed in the National Tidal Fringe Guidebook (Shafer and Yozzo 1998).

Rationale for selecting the function

Significant anthropogenic changes such as construction of navigation channels through wetlands (i.e. the Gulf Intracoastal Waterway), subsidence resulting from groundwater extraction, oil and gas extraction, and reduced freshwater sediment inflows due to flood control structures have accelerated erosion of wetlands within Texas estuaries. Along parts of the Texas coast, erosion may cause shorelines to retreat at rates of up to 3 m/year^{-1} (White et al. 1985). The ability of marsh vegetation to stabilize sediments and reduce shoreline erosion has long been recognized by coastal engineers (U.S. Army Corps of Engineers 1954), and some of the first planted salt marshes in Texas were established for this purpose (Webb 1982). Vegetated intertidal wetlands provide a measure of protection against the destructive effects of wave energy associated with storm surges, wind-generated waves, and vessel wakes.

Characteristics and processes that influence the function

The ability of a tidal wetland to attenuate wave energy is a function of the frictional resistance characteristics of the vegetation, surface obstructions or micro-topography, and marsh width (Knutson, Allen, and Webb 1990). Emergent stems function as a flexible baffle to dampen wave energy and detain water. Stems may also trap organic debris ranging in size from leaves and twigs to logs. Trapped debris may induce additional drag and decrease water velocity. Detailed protocols for field determination of Manning's friction coefficient (n) on densely vegetated nontidal freshwater flood plains are presented in Gardiner and Dackombe (1983) and Arcement and Schneider (1989). Miller (1988) used Manning's coefficient to characterize frictional resistance attributed to intertidal marsh vegetation. A Manning's n of 0.06 was assigned for short *Spartina alterniflora* in a South Carolina salt marsh; *Juncus roemerianus* was assigned a Manning's n of 0.125, and is therefore considered more effective in dissipating tidal surges. *Spartina alterniflora* may reduce wave heights by as much as 71-94 percent and wave energy by 92-100 percent, while the roots of marsh vegetation serve to bind the marsh substrate (Wayne 1976; Knutson, Allen, and Webb 1990). The composition, density, and height of vegetation contribute the greatest effect to roughness of the marsh surface in Texas tidal fringe wetlands. Other landscape features, such as hummocks, patches of high marsh interspersed with low marsh, and the presence of shell berms also contribute to roughness and should also be considered. Typically, Texas tidal fringe marshes will have smooth to minor surface irregularities.

The wave climate at any given site is influenced by fetch, shoreline geometry, wind speed and duration, sediment grain size, water depth, and proximity to boat traffic (Knutson, Ford, and Inskeep 1981; Knutson and Inskeep 1982; Knutson and Woodhouse 1983). A number of methods can be used to evaluate wave climate, but many of these are beyond the scope of a rapid assessment method such as this one. Knutson, Ford, and Inskeep (1981) devised a simple, rapid method that included measures of average and longest fetch distances, sediment grain size, and shoreline geometry characteristics. The relative exposure indices (REI) calculated by Keddy (1982) include estimates of wind speed and duration as well as fetch distances, and have been shown to be highly correlated with sediment grain size parameters. The REI's provide a biologically meaningful and quantitative method for exploring relationships between wave energy regime and sediment type, vegetation community composition, epibenthic faunal communities, and seagrass bed structure (Keddy 1982, Pihl 1986, Fonseca and Bell 1998). These indices have been modified for use in Texas tidal fringe wetlands and may be useful for assessing the relative risk of site erosion.

The predominant wind direction in Galveston Bay is from the south-southeast, except during the months from December through February, when winter storms can generate brief but strong northerly winds (Webb and Dodd 1976). Mean annual wind speed and directional percent frequency were calculated for eight stations located along the Texas coast from Galveston Bay to Corpus Christi Bay (Figure 3) for use in calculating relative exposure indices (Appendix E, Table E1). For this initial effort, wind data summaries for only a single year (1997) were generated. Summary statistics calculated over multiple

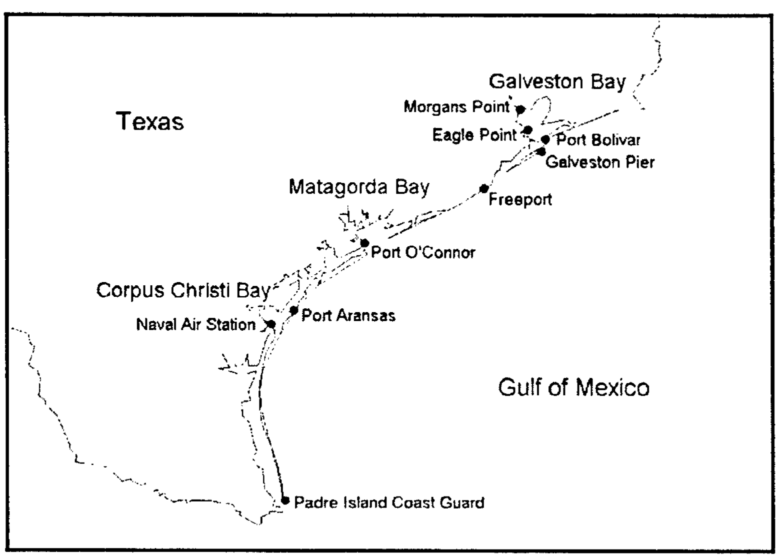

Figure 3. Locations of wind data stations

years would more accurately represent long-term wind patterns along the coast.

The potential for shoreline erosion due to vessel traffic will depend on the magnitude of the waves produced, traffic frequency, and the distance between the shoreline and the passing ships (Knutson and Woodhouse 1983). The height of waves produced by a ship is primarily dependent on its velocity; other factors such as hull design, draft, and water depth will also have a lesser effect (Sorenson 1973). Limited data suggest that wave heights are reduced by 25-50 percent at a distance of 150 m (Sorenson 1973). An example of vessel-wake-induced erosion of tidal fringe wetlands can be seen along the Gulf Intracoastal Waterway in Sundown Bay in the Aransas National Wildlife Refuge.

Large expansive marshes are more effective at dissipating the effects of wave energy than narrow fringing marshes because wave energy diminishes as the crest moves landward across the marsh surface. Marsh width generally depends on regional geomorphologic characteristics, tidal range, and slope of the shoreline. The loss of wave energy is directly related to marsh width (Knutson, Allen, and Webb 1990); however, field experiments on wave dissipation in *Spartina alterniflora* marshes have demonstrated that this relationship is nonlinear (Knutson et al. 1982). Based on a survey of marshes planted for shoreline erosion control, a minimum marsh width of at least 10 m was recommended by Knutson, Allen, and Webb (1990).

Belowground plant roots and rhizomes also play an important role in sediment stabilization, particularly during the winter months when the aboveground portions may be significantly reduced. Collection and processing of belowground samples require more time and effort than are typically available for a rapid assessment technique such as this one. Therefore, this component is not included as a model variable at this time, but could be considered if time and resources were available.

Description of model variables

The five variables in this function are only evaluated in cases where there is evidence of shoreline erosion within the project area. Examples of this include slumping banks, undercut banks, exposed root mats, or vertical bluffs along the shoreline (see Figure D1 for examples). If any of these features exist, assign a variable subindex to each of the following five variables using the procedures outlined below and calculate a functional capacity index for this function. Otherwise, assign a default functional capacity index of 1.0 to this function, indicating the presence of a stable, non-eroding shoreline.

Shoreline slope (V_{SLOPE}). Water depth affects the stabilization ability of tidal fringe wetlands by affecting the height of waves breaking on shore. Wetlands adjacent to navigation channels will also be subject to ship-generated waves. The variable assumes that tidal fringe wetlands having a steeply sloped shoreline or proximity to deep water have a lower shoreline stabilization potential (Knutson and Woodhouse 1983). The model for shoreline stabilization attempts to evaluate the erosion potential from vessel wakes and/or steep shorelines by estimating the distance from the shoreline to water depths of at least 2 m MLW.

Estimate V_{SLOPE} using the following procedure.

a. Estimate the distance from the shoreline needed to reach water depths of at least 2 m MLW using field reconnaissance, maps, or bathymetry charts.

b. Using Table 5, assign a variable subindex based on the distance measured or estimated above.

Table 5
Relationship Between Shoreline Slope and Functional Capacity

Distance to Navigation Channel or Water Depths ≥2 m	Variable Subindex
<50 m	0.1
50-150 m	0.5
>150 m	1.0

Average marsh width (V_{WIDTH}). This variable describes the distance that water must travel across intervening tidal fringe wetland (distance from the shoreline). Large expansive marshes are more effective at dissipating the effects of wave energy than narrow fringing marshes because wave energy diminishes as the crest moves landward across the marsh surface.

Measure average marsh width using the following procedure.

a. Using a recent aerial photo or direct field survey, establish a baseline along the lengthwise axis that runs roughly parallel to the shoreline and/or perpendicular to the topographic gradient.

b. Draw a series of transects perpendicular to this baseline from the shoreline to the nearest upland and estimate the average width of the marsh in meters (Figure 4). The number of transects is determined by the length of the baseline (Table 6).

c. Assign a variable subindex based on the chart in Figure 5.

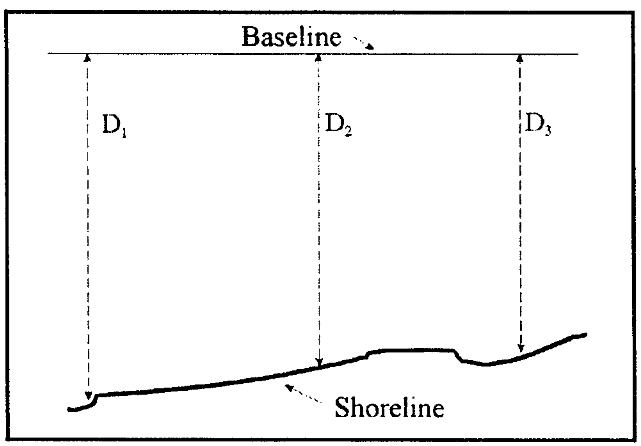

Figure 4. Determining mean marsh width

Table 6 Number of Transects for Estimating Mean Marsh Width	
Baseline Length (m)	**Number of Transects**
<300	3
300-1,500	5
1,500-3,000	7
>3,000	9

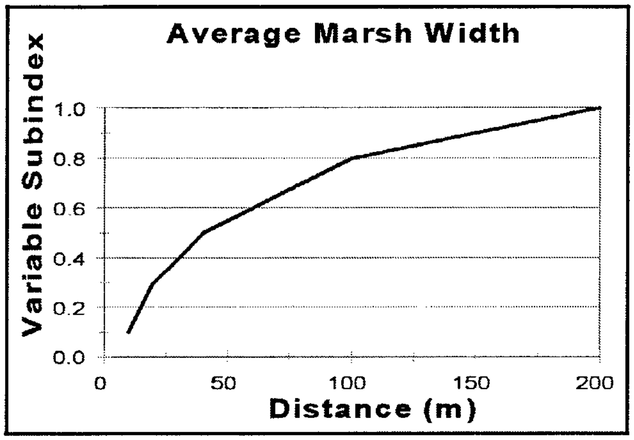

Figure 5. Relationship between mean marsh width and functional capacity

In Galveston Bay reference wetlands, average marsh widths ranged from less than 10 m to more than 1,000 m (Appendix D, Table D3). Based on the range of values at reference standard sites, a variable subindex of 1.0 is assigned to average marsh widths greater than or equal to 200 m (Figure 5). Marshes with mean widths less than 30 m sometimes displayed evidence of shoreline erosion, or were located in areas subject to local subsidence (White et al. 1985). The lower limit of the curve was established based on the recommendations of Knutson, Allen, and Webb (1990).

Exposure (V_{EXPOSE}). This variable estimates the potential for shoreline erosion due to wind-generated wave energy. Sites with high exposure indices will be subject to greater wave energy, and therefore have a higher potential for shoreline erosion.

Calculate a relative exposure index for each site using the following method as adapted from Keddy (1982).

a. Using aerial photographs, maps, or other charts, measure and record fetch distances for each of the 16 compass bearings (Figure 6). Fetch distances in some directions will be zero.

b. Select the wind data station closest to the site (Figure 3).

c. Using annual wind summary data from the appropriate wind station (Appendix E), calculate an exposure index according to the following formula. The wind summary data are available in spreadsheet format to facilitate rapid calculation of this variable.

$$\text{Exposure Index} = \sum_{i=1}^{16} V_i \times F_i \times P_i$$

where

V_i = mean annual wind speed (km/hr)
F_i = fetch distance (km)
P_i = proportion of time wind blows from each of 16 compass directions

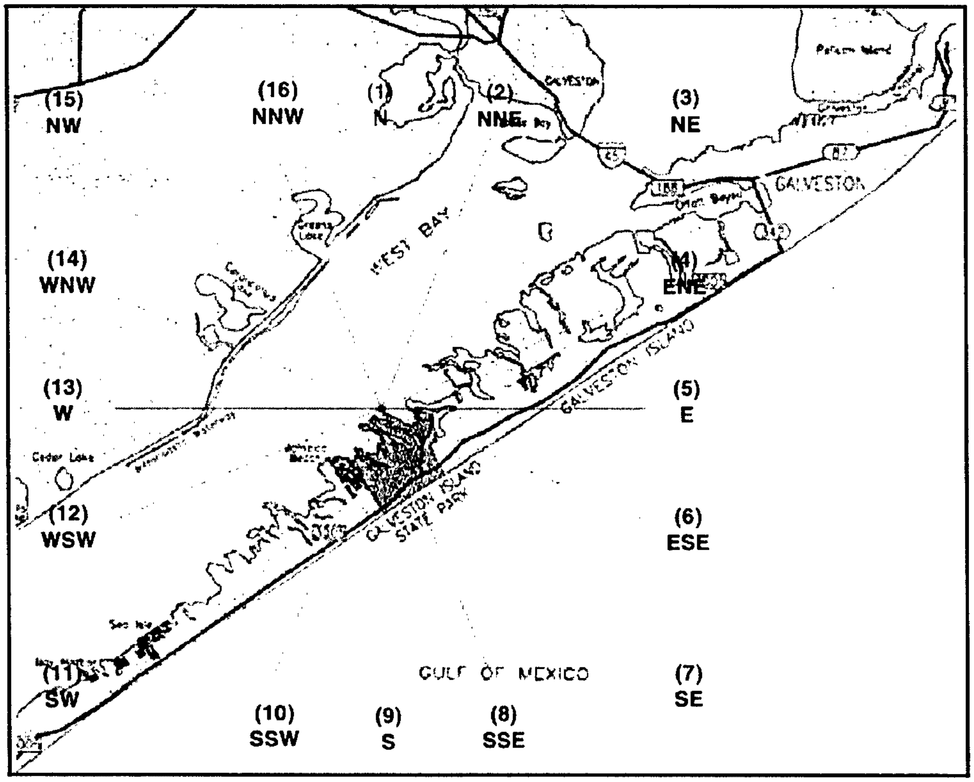

Figure 6. Measurement of fetch distances

d. Using Figure 7, assign a variable subindex based on the exposure index calculated above.

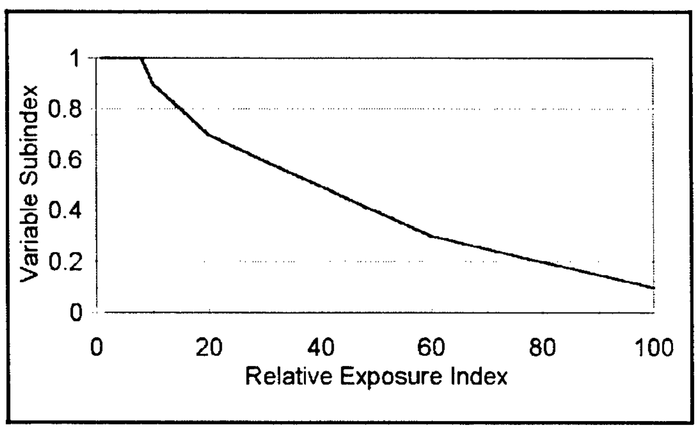

Figure 7. Relationship between relative exposure index and functional capacity

Exposure indices for tidal marshes in Galveston Bay ranged from less than 1 to 95 (Appendix D, Table D2). The exposure indices were significantly negatively correlated with soil silt-clay content (Spearman's nonparametric correlation, $p \leq 0.045$), indicating that sediments at exposed sites contained a higher proportion of coarse, sandy material than less exposed sites. Exposure indices were related to the landscape position of the individual sites, and could be classified into two groups. Sites with index values less than 6 were located on bayous, creeks, channels, or protected embayments and correspond to the low to moderate wave energy climates described by Knutson, Allen, and Webb (1990).

Sites with index values ranging from 9 to 95 were located in higher energy areas along roughly linear or convex stretches of shoreline exposed to greater fetch distances across the bays (i.e., Bolivar Peninsula sites). Marshes with very high relative exposure index values were typically characterized by the presence of an exposed peat shelf bayward of the marsh, and the formation of a raised berm composed of shell hash at the outer marsh edge. The highest exposure index values were calculated for two created marshes on Bolivar Peninsula. Although portions of these sites have persisted for more than 20 years, there is evidence of erosion along the bay shoreline. The lack of natural marshes along the same stretch of shoreline suggests that exposure indices ranging from 80-100 may be approaching some maximum threshold above which natural marshes do not occur in the landscape.

Based on these observations, sites with a relative exposure index less than or equal to 8 were assigned a variable subindex of 1.0 (Figure 7). Sites with exposure indices greater than 8 were assigned decreasing subindex values, with minimum subindex values at exposure indices greater than or equal to 100 (Figure 7).

Soil texture (V_{SOIL}). The potential for marsh erosion will be affected by the grain size distribution of the soils. Clay soils are cohesive and tend to resist erosion whereas sandy soils may be more easily eroded.

a. Determine the predominant soil texture for the site either directly in the field (see Appendix E for methodology) or from a county soil survey.

b. Based on the predominant soil texture, assign a variable subindex using Table 7.

Table 7
Relationship Between Soil Texture and Functional Capacity

Soil Texture	Variable Subindex
Sandy	0.2
Sandy loam	0.4
Loam	0.6
Clay loam	0.8
Clay	1.0

Surface roughness (V_{ROUGH}). This variable describes the potential effects of emergent vegetation on the hydrodynamics of tidal floodwaters. The baffling effect of emergent vegetation dissipates wave energy and slows water movement allowing deposition of sediments that contribute to accretion and stabilizing shorelines. Roughness coefficients represent resistance to flow. The density, diameter, and height of emergent macrophyte stems are major contributors to site roughness. Variation in surface microtopography may also contribute to roughness characteristics.

a. Estimate Manning's *n* roughness coefficients using the following equation (modified from Gardiner and Dackombe (1983)) for the determination of *n* values on vegetated tidal marsh surfaces:

$$n = (n_{BASE} + n_{TOPO} + n_{VEG})$$

where:

n_{BASE} = base value of *n* for the marsh's bare soil surface

n_{TOPO} = a correction factor for the effect of topographic relief on the marsh surface

n_{VEG} =a value for vegetation on the marsh surface

b. Using the information provided in Table 8, determine the appropriate value for the three variables in the previous equation. For the variable n_{VEG}, use the value for the mean total percent cover of the site (V_{COVER}) to determine the appropriate column. Then assign a value for n_{VEG} according to the predominant vegetation group present onsite. If there is more than one predominant vegetation type (i.e., 40 percent cover of *Spartina alterniflora* and 40 percent cover *Batis maritima*), then choose a value intermediate between these two groups.

c. Assign a variable subindex based on Figure 8.

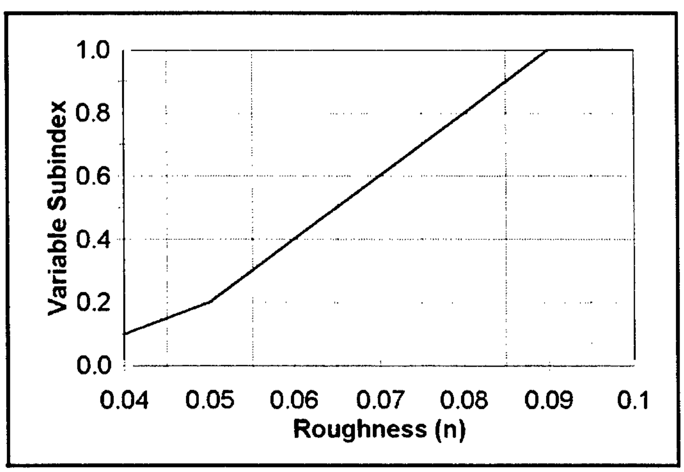

Figure 8. Relationship between surface roughness (*n*) and functional capacity

Table 8
Adjustment Values for Roughness Components Contributing to Manning's Roughness Coefficient (*n*)

Roughness Component	Adjustment to *n* Value			Description of Terms
Sediment surface (n_{BASE})	0.025			Base value for bare marsh soil.
	0.03			More than 25% of sediment surface covered with gravel or broken shell.
Topographic relief (n_{TOPO})	0.001			Representative area is flat with essentially no microtopographic (i.e., hummocks) or macrotopographic relief (i.e., berms, tidal channels, ridges and swales, ponds).
	0.005			Microtopographic (i.e., hummocks) or macrotopographic relief (i.e., berms, tidal channels, ridges and swales, ponds) cover 5-25% of a representative area.
	0.010			Microtopographic (i.e., hummocks) or macrotopographic relief (i.e., berms, tidal channels, ridges and swales, ponds) cover 26-50% of a representative area.
	0.020			Microtopographic (i.e., hummocks) or macrotopographic relief (i.e., berms, tidal channels, ridges and swales, ponds) cover >50% of a representative area.
Roughness Component	**Percent Cover**			**Description of Conditions**
	<50	**50-75**	**76-100**	
Vegetation (n_{VEG})	0.025	0.030	0.035	Representative area predominantly short, flexible-stemmed grasses (i.e., short *Spartina alterniflora, S. patens, Distichlis spicata*).
	0.035	0.040	0.050	Vegetation in representative area predominantly short, with stiff, trailing stems (i.e., *Batis maritima, Salicornia virginica*).
	0.050	0.060	0.070	Representative area predominantly tall, flexible-stemmed grasses (i.e., tall *Spartina alterniflora, S. cynosuroides, Scirpus sp.*).
	0.070	0.100	0.160	Vegetation in representative area predominantly tall, with stiff leaves (i.e., *Juncus roemerianus*) or mixed woody shrubs (i.e., mangroves).

Note: Adapted from Arcement and Schneider (1989) and Gardiner and Dackombe (1983).

Functional capacity index

The assessment model for calculating the functional capacity index is as follows:

$$FCI = (V_{SLOPE} + V_{WIDTH} + V_{EXPOSE} + V_{ROUGH} + V_{SOIL})/5$$

In the model, the capacity of a tidal fringe wetland to stabilize shorelines and attenuate wave energy is based on five characteristics. The variables are combined in a simple arithmetic mean, indicating that all variables are assumed to be equally important. The first variable V_{SLOPE} evaluates offshore slope. Tidal fringe wetlands with steeply sloping shorelines or close proximity to navigation channels will have a greater potential for shoreline erosion. The variable V_{WIDTH} evaluates the width of the vegetated marsh surface. Wide, expansive marshes are able to dissipate more wave energy as the wave crest travels across the marsh surface. V_{EXPOSE} estimates the potential for site erosion due to wind-generated waves based on the geomorphology and landscape position of the wetland. V_{ROUGH} represents the contributions of vegetation and topographic relief to the frictional resistance characteristics (Manning's n) of the marsh surface. V_{SOIL} indicates the potential for soil erosion based on sediment grain size distribution.

Function 2: Sediment Deposition

Definition

This function refers to the potential deposition and retention of inorganic and organic particulates from the water column, primarily through physical processes. A quantitative measure of this function would be millimeters sediment/square meter/year.

Rationale for selecting the function

Tidal marshes accrete vertically and expand horizontally across the coastal landscape by accumulating sediments. If sediment availability is reduced, or if accretion rates are insufficient to maintain pace with relative sea-level rise or storm-induced erosion, marsh loss will result. The ability of a tidal marsh to maintain an adequate rate of sediment deposition is critical to maintaining its integrity in a highly dynamic coastal setting.

Characteristics and processes that influence the function

Tidal marshes maintain their vertical and horizontal position in the coastal landscape by achieving a balance between two processes: 1) the accretion of mineral and organic sediments, and 2) coastal submergence due to the combined effects of sea-level rise and subsidence. Along transgressive coastlines, the vertical position of the marsh surface relative to mean sea level is determined by sediment supply and the frequency of tidal flooding events. Deposition occurs when the marsh surface is inundated, and suspended sediment settles onto the marsh surface. Most material settles out in the low marsh and along tidal creeks, forming natural levees; the least amount of material settles out in the high marsh, where peat accumulation is the dominant accreting process. If the accretion rate is sufficient, fringing marshes will accrete horizontally, as well as vertically to encroach upland mainland slopes, maintaining their areal extent as erosion occurs along the seaward edge (Kastler and Wiberg 1996).

Marsh sediments originate from a variety of potential sources, including terrestrial drainage, erosion of headlands or shore deposits, eolian transport, and seaward sources such as washover and longshore drift. The mineral fraction of marsh sediments includes both sand and finer silt and clay components. Organic constituents include plant detrital material, benthic micro- and macro-invertebrates, organic films, and animal fecal pellets (Kastler and Wiberg 1996). While it is recognized that sediment accretion in tidal marshes is critical to maintaining their position in the presence of subsidence and sea level rise, sediment accretion is not directly measured in this model due to a lack of rapid, accurate, and consistent techniques for the measurement of this variable.

Several factors may potentially affect the process of sediment accumulation in tidal marshes including elevation, flooding duration, suspended solid concentration, flow baffling by vegetation, and proximity to source (DeLaune, Baumann, and Gosselink 1983; Cahoon and Reed 1995; Leonard and Luther 1995; Leonard 1997). No single factor dominates; instead they act synergistically to control marsh sedimentation rates (Leonard 1997). High levels of function are associated with low elevation, high concentration of suspended sediment in floodwaters, and low organic content of the suspended sediments. Gleason et al. (1979) report sediment accretion rates to be a positive nonlinear function of stem density in laboratory experiments on newly planted *Spartina alterniflora* marshes.

The behavior of sediment particles with respect to site-specific tidal hydraulics influences the potential for marsh accretion. Settling lag, the tendency for a particle to continue moving with a fluid beyond the point where the current is competent to suspend it, is an important factor. Scour lag, the related process by which a particle remains in place even after its critical velocity has been reached, must also be considered. Tidal velocity asymmetry, the difference in the length of ebb and flood phases of the lunar tidal cycle, also affects the potential for accretion of sediment due to its effect on flooding duration and current velocity (Kastler and Wiberg 1996). Many of these processes are certainly beyond the scope of the HGM Approach.

Coastal storm events may have significant influence on sedimentation rates, especially in micro-tidal systems along the Texas coast, where storms result in extensive and prolonged inundation and canopy flow. In these systems, the effects of major storms may far surpass the amount of sedimentation attributed to daily tidal events (Wolaver et al. 1988; Leonard, Hine, and Luther 1995).

Description of model variables

Surface roughness (V_{ROUGH}). The baffling effect of emergent marsh vegetation and microtopographic relief, which retards surface water flow, allows suspended particulates to settle out of suspension (Stumpf 1983; Leonard, Hine, and Luther 1995). Emergent macrophyte roots and rhizomes serve to stabilize sediments, reducing the potential for tidal resuspension and transport. See Function 1 for guidance on the calculation of Manning's *n* roughness values for tidal marshes.

Hydrologic regime (V_{HYDRO}). The length of time during which the marsh surface is inundated has been demonstrated to affect sedimentation rates (Wolaver et al. 1988; Cahoon and Reed 1995). The opportunity for particles to settle out of suspension increases with increasing flooding duration. An accurate determination of flooding frequency and duration requires the installation and monitoring of water level recorders. Hydroperiod may also be estimated using surveyed elevations and tide gauge data.

In most cases, information on tidal flooding frequency and duration for each site is lacking. Since the collection of accurate hydrologic data is beyond the scope of a rapid assessment procedure such as this one, this variable is evaluated by determining if the WAA is free and open to exchange of tidal floodwaters. If normal tidal flooding is restricted due to the presence of berms, culverts, or other blockages, then the opportunity for sediment input is correspondingly reduced. Therefore, the subindex value for this variable is assumed to be 1.0 unless some type of obstruction prevents free and open exchange of tidal waters.

Estimate a value for the hydrologic regime using the following method.

a. Visually inspect the site and determine if hydrological restrictions are present. Examples of hydrologic restrictions include weirs, culverts, berms, or other structures that limit tidal flow. The value of the variable subindex V_{HYDRO} is assumed to be 1.0 unless tidal flow is restricted.

b. Match site condition with variable subindex value from Table 9.

Table 9
Relationship Between Hydrologic Regime (V_{HYDRO}) and Functional Capacity

Site Description	Subindex
Site is open to free exchange of tidal waters. No obvious hydrologic alteration or restrictions present.	1.0
Moderate hydrologic restriction present (i.e., presence of low-elevation berm, which is frequently overtopped by high tide events or has multiple breaches or large culverts).	0.6
Severe hydrologic restriction present (i.e., presence of high-elevation berm, which is infrequently overtopped by high-tide events or has a single opening, breach, or small culvert).	0.3
Site receives tidal floodwaters only during extreme storm tide events.	0.1
Site is isolated from tidal exchange. The principal source of flooding is water sources other than tidal action (i.e., precipitation or groundwater). *Note: If this condition exists, another wetland assessment model should be strongly considered unless the site was formerly a tidal wetland prior to hydrologic modification.*	0.0

Functional capacity index

The assessment model for calculating the functional capacity index is as follows:

$$FCI = (V_{ROUGH} \times V_{HYDRO})^{1/2}$$

The surface roughness characteristics slow the velocity of floodwaters, allowing suspended sediments to settle out. A geometric mean is used to average the effects of the two variables. The use of a geometric mean indicates that if the subindex of a single variable drops to zero, the results from the entire model will be zero. For example, if there is no tidal flooding, then there is no opportunity for sediment transport due to tidal action. Therefore it is appropriate that the FCI for this function computes as zero if $V_{HYDRO} = 0$.

Function 3: Nutrient and Organic Carbon Exchange

Definition

This function describes the ability of a tidal wetland to export or import nutrients and organic carbon via tidal flushing, deposition, and erosion. Even though net fluxes on an annual basis may be small, changes in form or timing may be highly consequential (in - inorganic, out - organic; uptake - spring, release -fall). A quantitative measure of this function is mass of nutrient or dissolved and particulate carbon transformed per unit area per unit time ($g/m^2/yr$).

Rationale for selecting the function

Tidal nutrient flux via surface water or groundwater is important in maintaining the high levels of primary productivity characteristic of tidal wetlands. In turn, the large stores of plant material produced supply organic carbon in a wide variety of forms for consumption or accumulation within the system or export to the adjoining estuary or littoral fringe. These systems may either import or export nutrients and organic carbon, depending on specific geomorphologic characteristics, time of year, marsh age, and other factors. Characterizing the magnitude and direction of nutrient and organic carbon fluxes in tidal wetlands is important in determining the wetland's ability to mediate water quality and to maintain characteristic plant communities. The latter is particularly relevant to newly created or developing tidal wetlands, where nutrient limitation often dictates project success or failure.

Characteristics and processes that influence the function

The Tidal Nutrient Exchange Potential and Particulate Organic Carbon Exchange Potential functions of the National Guidebook (Shafer and Yozzo 1998) were seen as behaving indistinguishably in the Northwest Gulf of Mexico region. Therefore, the two functions have been combined for regional application. Also, a large proportion of organic carbon exchanged between tidal wetlands and other systems can be in dissolved form rather than particulate. Since the characteristics governing exchange of dissolved organic carbon are the same as for nutrients and particulate organic carbon (flooding regime, potential for plant growth, amount of surface exposed to water for exchange), dissolved organic carbon exchange was assumed to be covered by the function as well.

Nutrients can enter tidal wetlands by precipitation, upland runoff, groundwater flow, and tidal exchange. Once in the wetland, nutrients may be deposited on the bottom, adsorbed to particles, or taken up and fixed in the tissues of rapidly growing vascular plants. The nutrients may be incorporated or otherwise transformed by microbial assemblages associated with the complex of surfaces provided by the sediment, live plants, litter, and detritus. This index considers nutrient exchanges and fluxes of organic carbon mediated by tidal conditions. The index incorporates variables that are believed to represent site characteristics that would logically contribute to tidal exchange of surface waters and any associated nutrients and organic carbon. These variables include flooding duration, which is an indicator of the amount of time that surface waters reside upon the marsh surface, during which infiltration of nutrients to the root zone may occur and the water comes in contact with the microbial films covering sediment, live plant, and litter surfaces. The other variable, vegetative structure, expresses the vegetative structural characteristics of emergent plants on the marsh surface. The model deviates from the National Guidebook in not including a variable for emergent macrophyte community composition, because, in the experience of the A-team, although the region's tidal marshes vary widely in species composition, a marsh's effect on nutrient and organic carbon exchanges does not necessarily depend upon the number of species present. The A-team could envision comparable performance of this function by single or multispecies marshes within the same salinity and inundation zone. Furthermore, the A-team saw the variable as highly scale-dependent, so that it would present difficulties in formulating reproducible comparisons to reference standards.

Odum (1974) proposed that nutrient inputs via tidal waters were important in maintaining the characteristic high productivity of *Spartina alterniflora* in creekside salt marshes. This occurs as a result of direct infiltration of nutrient-laden surface waters, horizontal recharge driven by rise and fall of the tide, and in some cases, vertical recharge from below the root zone. Salt marsh vegetation is primarily nitrogen limited, with ammonium nitrogen being the form most readily available in interstitial waters for uptake by plant roots. Phosphorus is abundant in saline waters and marsh soils, and is generally not considered a limiting nutrient in salt or brackish marsh systems. Numerous studies have attributed variation in *Spartina alterniflora* growth form to gradients in chemical and physical characteristics of tidal marshes, including nutrient availability (Valiela and Teal 1974; Broome, Woodhouse, and Seneca 1975; DeLaune and Pezeshki 1988). This is particularly true for developing or created salt marshes. Other workers suggest that, in mature marshes, edaphic factors affecting nutrient uptake are the primary determinants of *Spartina* growth form. Variables known to stress plants (high soil salinity and sulfide concentrations, waterlogging, low dissolved oxygen) reduce the uptake efficiency of both ammonium and nitrate at the root-pore water interface, especially when multiple stressors are present.

Nutrient exchange capacity in tidal wetlands may be considered a function of marsh age. Older, well-developed marshes are generally characterized as having fine-grained, nutrient-rich organic soils; these systems tend to export nutrients to the adjacent estuary. In contrast, newly developed marshes characterized by coarse, sandy soils generally lack well-developed nutrient pools and are devoid of binding sites associated with soil organic matter. In these younger wetlands, direct nutrient limitation is important and a net import of nutrients generally occurs. This has been demonstrated by fertilization experiments in salt marshes (Broome, Woodhouse, and Seneca 1975; Osgood and Zieman 1993) in which *Spartina alterniflora* plants in newly developed marshes exhibited an enhanced growth response relative to plants in older marshes.

Previous efforts to characterize nutrient exchanges in tidal marshes have yielded varying results, and seasonal differences in nutrient exchange are often pronounced. Major fluxes of nitrogen in the Great Sippewisset marsh in Massachusetts were attributed to nitrate in groundwater and dissolved organic nitrogen (DON) in tidal surface waters. A net export of DON was documented; however, inputs and outputs of total nutrients were approximately equal (Valiela et al. 1978; Kaplan, Valiela, and Teal 1979). Woodwell et al. (1979) observed a net export of ammonium nitrogen during summer and fall from a Long Island, New York salt marsh. During winter and spring, the marsh imported ammonium and nitrate nitrogen. Wolaver et al. (1983) observed strong seasonal trends in tidal exchanges of nitrogen and phosphorus in a Virginia salt marsh, with considerable export of DON during fall, and a net import of phosphorus during most of the year. Aurand and Daiber (1973) observed a net import of inorganic nitrogen for a Delaware salt marsh over a single year, and Stevenson et al. (1977) reported a yearly net export of nitrogen and phosphorus from a Chesapeake Bay tidal marsh.

Nutrient flux in tidal freshwater wetlands has not been studied as extensively as in salt marshes; however, based on geomorphologic similarities and other ecosystem attributes, it has been suggested that, like salt marshes, they can function either as sources, sinks, or transformers of nutrients (Mitsch and Gosselink 1993). Simpson et al. (1983) found that tidal freshwater marshes in New Jersey imported nitrogen and phosphorus during spring, primarily from upland runoff. During late spring and throughout the summer, most nutrients were tied up in plant biomass. During fall, there was a rapid and significant export of nutrients associated with the rapid senescence and decomposition of plant material. Thus, overall, a net export of nitrogen and phosphorus occurred across the entire year.

The role of subsurface hydrology in controlling nutrient fluxes in tidal wetlands is not specifically addressed in the present function. Only recently have ecologists investigated the role of subsurface nutrient flux in determining gradients in primary productivity. Nutrient-rich marsh pore waters are exchanged at interfaces such as marsh creeks; subsurface exchanges may account for as much as 50 percent of nutrient export from tidal marshes (Jordan and Correll 1985). Childers, Cofer-Shabica, and Nakashima (1993) postulated that marshes in coastal areas with tidal ranges in excess of 1 m were dominated by subtidal horizontal exchanges of porewaters, whereas solute exchange in marshes characterized by tidal ranges of less than 1 m occurred primarily within marsh surface waters. Because the embayments of the Northwest Gulf of Mexico region are microtidal (<0.5 m) to low tidal (<1 m) and greater excursions in elevation are associated only with infrequent meteorological events, it is assumed that subsurface water exchanges can be ignored for regional applications. It has been demonstrated that, in areas of increased porewater flux, *Spartina alterniflora* production is enhanced. Regularly flooded tidal marshes, especially those in an early stage of development, tend to exhibit increased rates of nutrient flux relative to older marshes, in which distinct, isolated zones of subsurface hydrology are recognized, and within which internal nutrient cycles may predominate.

Tidal wetlands are known to export organic carbon to nearshore coastal waters in both dissolved (DOC) and particulate (POC) forms. This process has been the focus of a dominant paradigm in coastal ecology, the "outwelling hypothesis" (Odum 1980). Although the importance and general applicability of this paradigm have been challenged in recent years, particularly with regard to the role of phytoplankton production in coastal waters, it has formed the basis of much subsequent research and numerous management strategies in coastal systems. Many studies have attempted to estimate net flux of detrital material to coastal waters, although such estimates are often subject to considerable error, primarily due to tidal cycle asymmetry. The relative importance of DOC versus POC is still largely unknown, due to the difficulty in estimating leaching rates of DOC from decomposing macrophytes and other sources (phytoplankton, benthic algae). For the purposes of this assessment procedure, the marsh structure variables are assumed equally good indicators of availability of POC and DOC for tidal exchange.

The index incorporates those variables that are believed to represent biological and hydrologic characteristics of a site that would logically contribute to the production, suspension, and removal or capture of detrital particles and DOC via the overlying water column. A number of potentially important factors are not considered in the present index, including decomposition rates, which can vary seasonally among plant species and even between different parts of the same plant. Decomposition of labile, broad-leaved emergent vegetation, such as *Peltandra virginica* or *Sagittaria* spp. in tidal freshwater marshes occurs more rapidly than breakdown of salt marsh species such as *Spartina patens* or *Juncus roemerianus*, which are characterized by high carbon:nitrogen ratios, and thus decompose gradually (Odum and Heywood 1978). Water and air temperature are key determinants of the rate of organic matter decomposition. Microbial activity associated with decomposing marsh vegetation is mediated by temperature decreases in winter. The rate of decomposition of detrital material is inversely related to particle size. Large fragments of plant tissue are broken down rapidly by invertebrate grazers, either via passage through the gut or mechanical fragmention by chewing. Storm events are not considered here; however, they are certainly responsible for the transport of considerable amounts of suspended organic and inorganic materials in tidal marsh systems.

Wiegert, Christian, and Wetzel (1981) developed an ecosystem model for salt marshes on Sapelo Island, Georgia. In their model, the tidal export coefficient estimated a combined POC and DOC export value of approximately 1,000 $gC/m^2/yr$. Hackney and De la Cruz (1979) determined that a single tidal creek near Bay St. Louis, Mississippi was responsible for a net *import* of particulate organic matter (38.32 kg/yr). The authors suggested that individual creeks may actually serve to dampen long-term oscillations in detrital availability to nearshore waters rather than providing a constant source of detrital material.

Description of model variables

Nutrient cycling and organic carbon exchange in tidal systems are mediated by physical, chemical, and biological factors. Many of the factors affecting nutrient cycling and organic carbon exchange are either poorly understood or beyond the scope of a rapid assessment method such as this one. The variables chosen for this functional index represent those factors that are both practical to measure and are presumed to affect nutrient and organic carbon exchanges in tidal systems. High levels of function are assumed to occur at those sites having flooding frequency, duration, and plant biomass (percent cover and height) similar to reference standard sites.

Hydrologic regime (V_{HYDRO}). Nutrients infiltrate to the root zone during periods of inundation. Increases or decreases in the flooding duration at a particular site relative to reference standard sites in the region may change nutrient cycling and organic carbon exchange patterns within the marsh. See Function 2 for a description of variable measurement.

Vegetative structure (V_{VEGSTR}). Emergent macrophytic plants take up and transform minerals, nutrients, and other compounds, which are later released as the plants begin to senesce and decay. They provide substrate for bacterial and fungal growth and are an important source of organic carbon, which may be exported to adjacent ecosystems.

Structural complexity of the vegetative community is assessed using a weighted combination of mean height and percent cover measurements. A weighted height index is believed to be more representative of vegetation structural complexity than either percent cover or plant height measurements individually. In addition, species level identifications are not necessary. Weighted height indices ≥30 are assigned a variable subindex of 1.0 based on values obtained at reference standard wetlands. For reference marshes in Galveston Bay, the most frequently observed heights for *Spartina alterniflora* were between 45 and 60 cm. The most frequently occurring heights for *Distichlis spicata* were 20-35 cm, and the most frequently occurring heights for *Batis and Salicornia* sp. were 35-45 cm (Appendix D, Figure D10). These measurements were obtained in the spring, and may be subject to seasonal variability.

If multiple persons will be performing vegetation estimates, it would be prudent for the entire team to spend some time in the field prior to performing an assessment to ensure that the estimates performed by different persons on the same plot are within an acceptable margin of error.

Measure this variable using the following procedure.

a. Select one or more representative areas within the site for sampling. Beginning at the shoreward edge of the marsh, establish one or more 30-m transects perpendicular to the shoreline or along the hydrologic gradient (e.g. increasing elevation). If there are multiple vegetation community types within the Wetland Assessment Area (WAA), the transect should intersect each vegetation community, in order to ensure a representative sample.

b. Using a standard 1-m^2 frame, estimate total percent cover by both live and dead emergent macrophytic plant species at intervals along the transect, excluding any areas where water depths are too deep to support the growth of emergent vegetation. The number of transects and plots needed will depend on the size and heterogeneity of the site; a minimum of 10 plots should be used.

c. If the 1-m^2 sample plot above contains more than one species (i.e., *Spartina* and *Distichlis*), estimate the proportion of the 1-m^2 plot area covered by each species, omitting any species that occupies less than 10 percent cover. If the total percent cover above was estimated at 80 percent, the sum of the percent cover of each individual species should be 80 percent. There may be cases where there are several species that individually account for more than10 percent cover, but collectively amount to 10 percent. In these cases, estimate the cumulative percent cover for the species group.

d. For each species identified above, estimate the height in centimeters (rounded to the nearest 5 cm) at which the bulk of the biomass occurs (i.e., the most frequently occurring height) and record this value. For those species with trailing stems, the height should be measured in situ rather than extended vertically. Record an estimated height for the species group, if necessary.

e. Calculate a vegetative structure index for each plot using the equation below.

$$\text{Index} = \sum_{i=1}^{n} \textit{Height } Sp_i \times \textit{Percent Cover } Sp_i$$

where n = number of species present in each 1-m^2 plot

f. Calculate the average of the vegetative structure indices for all plots to obtain a mean total site vegetative structure index.

g. Using Figure 9, determine the variable subindex that corresponds to the mean total site vegetative structure index.

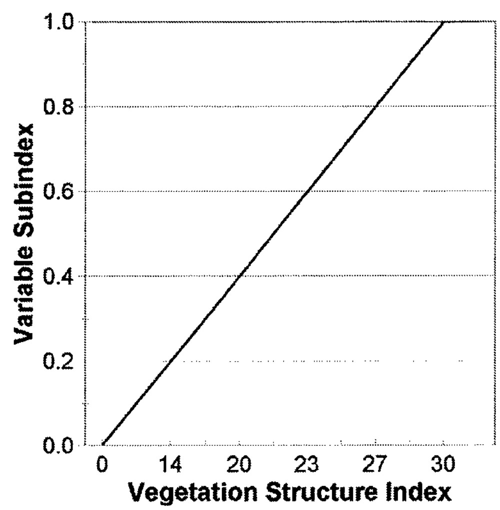

Figure 9. Relationship between vegetation structure index and functional capacity

Functional capacity index

The assessment model for calculating the functional capacity index is as follows:

$$\text{FCI} = (V_{HYDRO} \times V_{VEGSTR})^{1/2}$$

Site hydrology and the aggregate of the plant variables representing vegetative structure were considered of equal weight. Site hydrology and vegetative structure were aggregated using a geometric mean so that a zero value for either would result in an FCI of zero, as required by logic. (Regardless how much biomass there may be, there can be no exchange if there is no flooding. Conversely, regardless how appropriate the flooding regime, there can be no marsh-mediated exchange if there is no biomass.)

Function 4: Resident Nekton Utilization

Definition

This function describes the potential utilization of a marsh by resident (nonmigratory) fish and macrocrustacean species. A quantitative measure of this function would be abundance (or biomass) of resident nekton per square meter.

Rationale for selecting the function

Tidal marshes provide forage habitat, spawning sites, and a predation refuge for resident fishes and macrocrustaceans. These organisms are typically year-round residents of intertidal marshes and adjacent subtidal shallows. The ubiquitous killifishes (*Fundulus* spp.) and grass shrimps (*Palaemonetes* spp.) are characteristic residents of Atlantic and Gulf coast intertidal wetlands. These organisms are consumed by nektonic and avian predators and are considered to represent an important link in marsh-estuarine trophic dynamics.

Characteristics and processes that influence the function

The importance of tidal marshes as habitat for both resident and nonresident nekton species is one of the most-often-cited functions of this wetland type (see also "Function 5, Nonresident Nekton Utilization"). Most evidence suggests that resident organisms (e.g., killifishes, grass shrimps) utilize the entire marsh surface across the range from low to high elevations for foraging, reproduction, and as a refuge from predators. Although a number of factors are believed to determine utilization of these areas by nekton, these variables are often difficult to quantify and may not necessarily be supported by available research. The variables used in the model are based on documentation in the primary literature. The model includes the following factors: habitat complexity, access to and availability of "aquatic edge," and the duration of tidal flooding. It is assumed that the potential utilization of a site by resident nekton will change as a direct function of each of these variables.

Resident nekton are widely distributed throughout the lower intertidal marsh early and late in the tidal cycle (Kneib 1984a; Rozas and Reed 1993). Field experimentation has shown that the mummichog (*Fundulus heteroclitus*) requires access to the marsh surface for foraging to maintain normal growth rates (Weisberg and Lotrich 1982; Kneib 1997). For larval mummichogs, growth is positively related and mortality rates are negatively related to flooding duration (Kneib 1993). Gulf killifish (*Fundulus grandis*) consume more prey when they have access to the marsh surface than when they are confined to subtidal areas by low tides (Rozas and LaSalle 1990). Resident nekton will make extensive use of high marsh when spring tide conditions facilitate access to the upper intertidal zone. Kneib (1993) found that when high and low *Spartina alterniflora* marshes were flooded for equal periods (5 to 6 hr), growth rates and survival of mummichog larvae were greater in the high marsh, presumably due to greater availability of preferred invertebrate prey. The dense vegetation characteristic of high marsh habitats may also offer greater protection from natant predators in comparison to low marshes. Several resident killifish species, including *Fundulus heteroclitus,* rely on availability of high intertidal marsh, coincident with spring tidal events, for use as spawning sites (Taylor et al. 1979, Taylor and DiMichele 1983, Greeley and MacGregor 1983).

Tidal creeks and channels are used as "staging areas" for resident and nonresident nekton at low tide and represent corridors between the marsh surface and deeper, subtidal habitats (Rozas, McIvor, and Odum 1988). In tidal freshwater marshes, the presence of dense submerged aquatic vegetation (SAV) provides foraging opportunities and a predation refuge to resident nekton confined to subtidal areas at low tide (Rozas and Odum 1987a, 1987b). The shallow pools that remain in intertidal channels may also provide a low-tide refuge for resident species (Kneib and Wagner 1994). Shallow, water-filled depressions and rivulets distributed across the marsh surface provide habitat for small resident organisms and allow them to remain there at low tide (Kneib 1978, 1984a, 1987, 1997).

Resident nekton are not confined to the marsh "edge" or marsh/open water interface due to their mobility, small size, and broad environmental tolerances. However, densities of most species tend to decrease substantially with distance from the marsh/open water interface. Therefore, resident nekton abundance across the intertidal marsh surface may be positively correlated with the amount of marsh "edge" available (Kneib and Wagner 1994; Minello, Zimmerman, and Medina 1994; Peterson and Turner 1994; Zimmerman and Minello 1984).

Description of model variables

Aquatic edge (V_{EDGE}). The amount of marsh/water interface or edge is considered to be an important factor governing the exchange of organisms. With the availability of digital imagery produced from aerial photography, it is possible to obtain measurements of edge between the marsh and adjacent bodies of water, including the edge along open water such as rivers or bays, edges of ponds, and edges along banks of tidal creeks using GIS software. Measurements of edge may be subject to large variations depending on the scale of measurement used, however. Therefore, it is crucial that the scale of measurement be the same for all sites if sites are to be compared. The measurement scale should also be chosen so that relatively small water bodies and patches of marsh are visible. At a scale of 1 cm = 48 m (1:4800 or 1 in. = 400 ft), water bodies and patches of marsh as small as 1 m in diameter may be detected.

A GIS-based analysis of digital imagery is recommended as the method of choice for estimating the relative amount of edge among several sites. However, most regulatory personnel who are involved in routine wetland assessments may not have access to GIS software or time available for detailed GIS measurements of each site. Therefore, a simple pattern recognition technique is proposed as an alternative, based on the degree of landscape complexity of a site. This approach has been used in other rapid assessment techniques, such as the Wetland Evaluation Technique (WET) (Adamus et al. 1991), and the Wetland Value Assessment (WVA) (Coastal Wetlands Planning, Protection, and Restoration Act Environmental Work Group 1998).

Using GIS-based analysis of digital imagery, the edge:area ratio was measured for a series of reference sites in the Mid-Coast subregion of the reference domain (Table D5). Natural tidal marshes with edge:area ratios more than 800 m/ha were observed in areas that had been subject to the effects of moderate to severe subsidence (i.e., Swan Lake, Bolivar Peninsula near Elmgrove Point, and Dickinson Bayou); therefore, sites with edge:area ratios more than 800 m/ha were assigned a variable subindex of 0.8. The relatively high subindex value acknowledges the importance of these sites in terms of fisheries access, but also recognizes the fact that these sites do not represent the reference standard condition, and therefore should not be assigned a variable subindex of 1.0. Those sites with "high" edge:area ratios between 400 and 800 m/ha were considered the reference standard condition and are assigned a variable subindex of 1.0. Lower subindex values are assigned to those sites with edge:area ratios less than 400 m/ha (Table 10).

Measure or estimate V_{EDGE} using one of the following techniques.

a. Using aerial photography at a scale of 1 cm = 48 m (1 in. = 400 ft), assign a subindex value for the site using the qualitative descriptions provided in Table 10. See the pictorial key in Appendix D for specific examples.

b. Alternatively, using aerial photography at the same scale, measure all visible marsh/water interfaces, including edges of tidal creeks (both banks), ponds, creeks, and open bay shoreline. Determine the total marsh area in hectares, and express the total amount of edge in meters as a function of total marsh area. Assign a subindex value using the data in Table 10.

Table 10
Estimating V_{EDGE} Based on the Amount of Marsh/Water Interface Present

Site Description	Edge:Area		Subindex
	Qualitative	Quantitative (m/ha)	
1) Marsh shows signs of deterioration due to subsidence (i.e. highly fragmented with large amounts of open water. Vegetation occurs mainly in isolated hummocks or on natural levees along tidal creeks). Although edge:area is very high, this condition is not considered sustainable in the long term (Figure D2).	Very High	>800	0.8
1) Well-developed tidal drainage network present (Figure D3), OR **2)** Simple tidal drainage network (may consist of only a single channel) present with isolated ponds and depressions present in the marsh interior (Figures D4 and D5). **3)** Atypical geomorphic configuration with a large amount of shoreline relative to total area (i.e. small island or narrow peninsula) (Figure D7).	High	350-800	1.0
1) Simple tidal drainage network (may consist of only a single channel). Isolated ponds and depressions are few or lacking **2)** Narrow fringe marsh that lacks tidal creeks. One lengthwise shoreline is exposed to tidal waters. Area of marsh is small relative to shoreline length (Figures D6 and D8).	Moderate	200-350	0.7
Marsh lacks both tidal creeks and isolated ponds and depressions. Shoreline is generally linear or smooth curvilinear without embayments or convolutions. Area of marsh is large relative to shoreline length (Figure D9).	Low	<200	0.4

Hydrologic regime (V_{HYDRO}). Since resident nekton are only able to access the surface of the marsh when it is flooded, the potential utilization of a site by these species is directly related to the length of time that the marsh surface is inundated. See Function 2 for guidance on estimating this variable.

Nekton habitat complexity (V_{NHC}). Habitat complexity is a measure of the heterogeneity of a site, based on comparison of the number of habitats actually present at a site relative to the number of possible habitats known to occur in the appropriate regional subclass. Different marsh vegetation types (i.e., low, mid, high marsh), water bodies (e.g., ponds, tidal creeks, and channels), physical structures (e.g., coarse woody debris, oyster reefs), and the presence of submerged aquatic vegetation in adjacent subtidal areas all contribute to the habitat complexity of a site, and may affect utilization by resident nekton species. Since it is highly unlikely that all possible habitat types can be detected from aerial photos of the site, a field visit will be required to obtain the data necessary to calculate this variable. The user should refer to the reference standard data set for the particular regional subclass in question to determine the possible habitat types that may be present.

There were a total of 11 possible different habitat types identified in reference wetlands in the Galveston Bay area (Table 11). Based on the conditions observed at reference standard sites, a variable subindex of 1.0 is assigned if at least six of these habitat types are present onsite or within a 30-m radius of the site. Assign a variable subindex based on Figure 10.

Table 11 Possible Nekton Habitat Types
Low marsh (i.e., *Spartina alterniflora*)
High marsh (i.e., *Spartina patens, Batis maritima*)
Subtidal creeks/channels
Intertidal creeks/channels
Ponds or depressions
Submerged aquatic vegetation (i.e., *Ruppia maritima, Halodule wrightii*)
Oyster reef
Unvegetated flats
Algal mats
Mangroves
Coarse woody debris

Functional capacity index

The assessment model for calculating the functional capacity index is as follows:

$$FCI = (V_{EDGE} + 2\ V_{HYDRO} + 0.5\ V_{NHC})/3.5$$

The weighting factors assigned to each variable in the model equation reflect the perceived relative importance of each of these factors to resident nekton utilization of the marsh. The hydrologic regime was deemed most important, followed by the amount of marsh/water interface or edge. Habitat complexity was subjectively determined to be of lesser importance than either of the two previous variables.

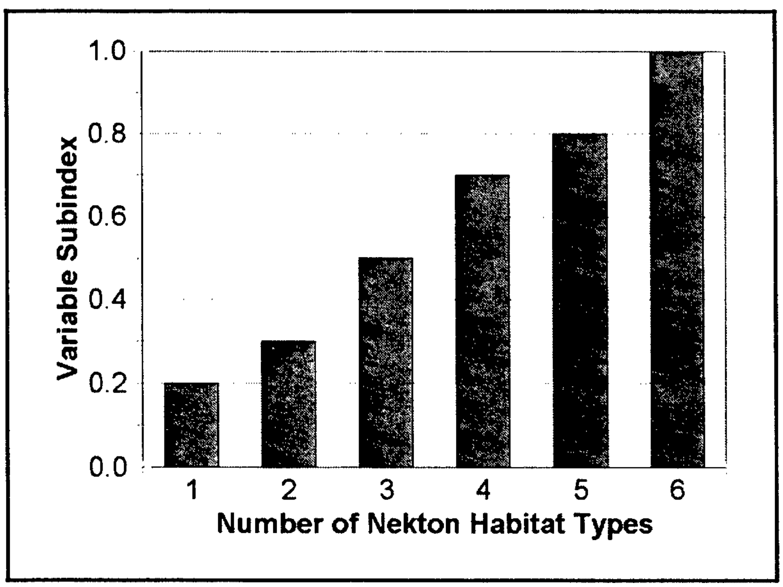

Figure 10. Relationship between nekton habitat complexity and functional capacity

Function 5: Nonresident Nekton Utilization

Definition

This function describes the potential utilization of a site by seasonally occurring adults or juveniles of marine or estuarine-dependent fisheries species. A quantitative measure of this function would be abundance (or biomass) of nonresident nekton per square meter.

Rationale for selecting the function

Tidal marshes provide foraging opportunities and a predation refuge for a variety of estuarine-dependent fisheries species. Most of these organisms are seasonal inhabitants, entering tidal marshes as juveniles in the spring, and leaving in the fall. Several important commercial fisheries in the United States (i.e., Southeast and Gulf coast penaeid shrimp) are critically dependent on the availability of suitable tidal marsh nursery habitat.

Characteristics and processes that influence the function

Nonresident, or transient, fishes and macrocrustaceans utilize tidal wetlands as forage sites and for protection from predators. This model is based on both the opportunity and the means by which transient nekton access a tidal wetland and the attributes of the wetland that provide prey resources and refuge from predation. The model incorporates variables that include measurement of the proximity of a site to subtidal source channels, access to the site via the tidal drainage channel network, the nature and extent of "aquatic edge," the duration of tidal flooding, and a measure of habitat complexity. It is assumed that the potential utilization of a site by transient nekton will change as a direct function of each of these variables.

Nonresident nekton utilize tidal wetlands on a seasonal basis and typically do so only for part of their life cycle. In most cases, it is the juvenile forms that utilize these habitats as nurseries and refugia from large predators. Unlike resident nekton (e.g. killifishes, grass shrimps), which utilize intertidal wetlands for most of their life histories and who utilize the entire elevational range of these habitats (i.e., low to high elevation zones), nonresident nekton are more restricted in their access to these areas. These organisms invade coastal marshes on rising tides, access the marsh almost exclusively through the tidal channel system, utilize the interior marsh surface only during longer, higher tides, and usually vacate all tidal channels during tidal exposure (Zimmerman and Minello 1984; Kneib 1991; Rakocinski, Baltz, and Fleeger 1992; Rozas and Reed 1993; Baltz, Rakocinski, and Fleeger 1993; Peterson and Turner 1994).

Most transient nekton species found in coastal marshes originate from subtidal habitats (mainstream and large distributary channels, deepwater bay or ocean) that are linked to marshes by the tidal drainage system. Although resident nekton may occupy residual waters in tidal channels within or adjacent to the marsh (see "Function 4, Resident Nekton Utilization"), nonresident nekton tend not to remain in shallow microhabitats and must retreat to deeper water on most ebb tides. Thus, the tidal channels linking the marsh drainage system and the subtidal refuge constitute corridors between the two habitats (Rozas, McIvor, and Odum 1988). Although information on recurring movement patterns is lacking, the current belief is that transient nekton have no strong fidelity toward a particular wetland site, but tend to move about an estuary or between estuaries.

Description of model variables

Aquatic edge (V_{EDGE}). See Function 4 for description of variable measurement.

Opportunity for marsh access (V_{OMA}). V_{OMA} is estimated by calculating the percentage of edge that is tidally connected (channels, embayments, ponds). The percentage of connected waterways across the WAA is an indirect measure of the surface of the marsh that is occupied by access routes for aquatic organisms. Unlike aquatic edge, which includes all possible interfaces (including areas that lack a tidal connection to the estuary; e.g., isolated ponds, pans), this variable estimates the contribution that water bodies with connections to the estuary alone have on the potential access of transient organisms, thereby reflecting the assumed relative importance of this form of edge over others. The proportion of the total edge that was tidally connected ranged from 50-100 percent in reference standard wetlands in Galveston Bay.

Measure or estimate V_{OMA} using one of the following methods (in order of preference).

a. Using recent aerial photography at a scale of 1 cm = 48 m (1 in. = 400 ft), and GIS mapping, determine the proportion of the total edge that is tidally connected (excludes isolated ponds and depressions).

b. Using recent aerial photography at a scale of 1 cm = 48 m (1 in. = 400 ft), and an English area grid, determine the proportion of the total edge that is tidally connected (all tidal creeks and shorelines with a direct tidal connection, excluding isolated ponds and depressions).

c. Using Table 12, assign a variable subindex for V_{OMA}.

Hydrologic regime (V_{HYDRO}). The opportunity for transient nekton to access the tidal channel system, as well as the marsh surface from the tidal channels, is primarily determined by the duration of tidal flooding. Transient species may have to wait longer for sufficient water to accumulate before they access the marsh surface, and must vacate the marsh surface earlier than resident nekton on falling tides. Individual species may vary considerably in the degree to which they use the flooded intertidal marsh

Table 12
Relationship Between Opportunity for Marsh Access (V_{OMA}) and Functional Capacity

Tidally Connected Edge: Total Edge Ratio	Variable Subindex
50-100%	1.0
35-50%	0.7
25-35%	0.5
1-25%	0.2
No tidally connected edge present.	0.0

surface; however, it appears that maximum utilization (in terms of abundance and species richness) occurs at slack high water (Kneib and Wagner 1994). See Function 2 for guidance on estimating this variable.

Nekton habitat complexity (V_{NHC}). Use of the marsh by nonresident nekton may also be influenced by the structural attributes of the intertidal and adjacent subtidal habitats. Many nekton species, such as penaeid shrimp, exhibit preferences for certain attributes of marsh vegetation, such as stem density or height, which may mediate susceptibility to predation (Minello and Zimmerman 1983, Zimmerman and Minello 1984). Other structures, such as submerged vegetation (Rozas and Minello 1998; Sogard and Able 1991; Thomas, Zimmerman, and Minello 1990; Orth and van Montfrans 1987), coarse woody debris (Everett and Ruiz 1993), oyster reefs (Crabtree and Dean 1982), and the prop roots of red mangroves (Thayer, Colby, and Hettler 1987) also appear to attract transient nekton. Shallow ponds and ditches in the mid- to upper intertidal marsh may also attract transient nekton, but access will be limited to those organisms that can penetrate interior marshes on higher tides. See Function 4 for a description of variable measurement.

Functional capacity index

The assessment model for calculating the functional capacity index is as follows:

$$FCI = [((V_{EDGE} + 2\ V_{HYDRO} + 0.5\ V_{NHC})/3.5) \times V_{OMA}]^{1/2}$$

The first part of the model is the same as the previous model for resident nekton utilization, reflecting a similarity in habitat requirements for these species. An important difference between these two models, however, is the addition of the "Opportunity for Marsh Access" variable. Access to tidally connected channels is critical for these species, which spend only a portion of their life cycle in the marsh, and use tidal channels as corridors between the marsh and subtidal areas. A geometric mean was considered more appropriate for this portion of the model.

Function 6: Maintain Invertebrate Prey Pool

Definition

This function describes the potential for the wetland to produce and maintain a characteristic benthic and epiphytic invertebrate prey pool. A quantitative measure of this function would be abundance of invertebrate species per square meter.

Rationale for selecting the function

Benthic and epiphytic invertebrates represent a critical link in the trophic transfer of energy (in the form of secondary production) to near-coastal waters. Resident nektonic predators (e.g., killifish, caridean shrimp) access the intertidal marsh surface on rising tides to forage on macroinfauna and epifauna. These consumers, in turn, are preyed upon by larger, predatory fishes in adjacent subtidal habitats.

Characteristics and processes that influence the function

The spatial distribution of benthic and epiphytic invertebrates in tidal marshes is known to be nonrandom. Important factors that may determine invertebrate distribution and abundance include predation, competition, and variation in environmental conditions. Physical variables, such as macrophyte stem height/density, and microtopography may also influence aggregation patterns of intertidal benthic organisms (Bell 1979, Van Dolah 1978, Osenga and Coull 1983, Rader 1984). Macroinfauna are often more abundant in association with dense marsh vegetation (e.g., *Spartina*), relative to bare or sparsely vegetated intertidal habitats. Small benthic organisms that are able to exploit the root or culm surface benefit from increased area for colonization. Nonrandom recruitment of larvae and post-larvae may result from the hydrodynamic effects of *Spartina* culms. The structural complexity of emergent macrophytes may inhibit predation by natant macrofauna (e.g., killifishes and caridean shrimp) on benthic and epiphytic invertebrates, resulting in differential postrecruitment mortality in vegetated versus unvegetated habitats (Rader 1984).

Small-scale patterns of invertebrate distribution have been attributed to the patchy distribution of microbial food sources (Findlay 1981; Decho and Castenholz 1986) and the influence of biogenic structures (Bell, Watzin, and Coull 1978; Osenga and Coull 1983). Certain taxa (e.g., nematodes) may be locally abundant around structures such as fiddler crab burrows; others, such as copepods, may exhibit reduced densities in the vicinity of biogenic structures. Microtopographic features, such as intertidal pools and rivulets, or elevated plant hummocks influence distribution patterns, abundance, and composition of small benthic and epifaunal invertebrates in tidal freshwater wetlands (Yozzo and Smith 1995).

Heat and/or dessication stress have been suggested as possible limiting factors in the distribution of intertidal invertebrate populations. However, the water-retaining properties and associated evaporative cooling of salt marsh peat enhances survival of benthic invertebrates during extended low tide/high temperature conditions (Van Dolah 1978) and most tidal marsh taxa tolerate a broad range of environmental conditions. Similarly, while sediment composition and texture may exert considerable influence on the distribution of certain benthic organisms in deeper aquatic or marine habitats, the distribution patterns of most common salt marsh benthic invertebrates are apparently not determined by sediment composition (Kneib 1984b). However, Wenner and Beatty (1988) indicated that the most important variables affecting the distribution and abundance of benthic and epifaunal invertebrates in South Carolina salt marshes included sediment composition, along with type and density of vegetation, amount of flooding and hydroperiod, and water circulation. Low water circulation and prolonged conditions of oxygen depletion are detrimental to colonization by many intertidal invertebrates.

Predation may exert significant influence on the abundance and population size structure of benthic and epibenthic fauna. Peak densities for salt marsh invertebrates seem to occur in spring or autumn with lowest densities occurring in mid-summer (Bell 1979, 1980, 1982; Cammen 1979; Kneib and Stiven 1982), when the abundance of the most common natant marsh predators (primarily killifish, *Fundulus* spp. and caridean shrimp) is highest. Predation effects are complex, difficult to quantify, and may be confounded by environmental factors (Wenner and Beatty 1988). Year-to-year variability in infaunal densities may be pronounced, suggesting that changes in certain environmental parameters (e.g., porewater and surface water salinity, dissolved oxygen concentrations, and soil pH) are important in determining seasonal and longer-term population dynamics of benthic and epiphytic invertebrates in tidal marshes.

Exclusions

The model excludes large filter-feeding bivalves (oysters, clams, and mussels). Because these organisms are less vulnerable to predation by most small natant marsh predators, and because their turnover rates are relatively low, these populations are not expected to fit the model. Some other large, conspicuous marsh invertebrates (e.g. periwinkles, *Littorina spp.* and fiddler crabs, *Uca* spp.) are also not likely to be represented in the model. Meiofauna (benthic organisms that pass through a 500-μm sieve but are retained on a 63-μm sieve) are also not explicitly considered in the model. Although certain meiofaunal taxa (e.g., harpacticoid and cyclopoid copepods, ostracods) are known to be important prey resources for larval and juvenile fish and macrocrustaceans in tidal marshes (Bell and Coull 1978; Ellis and Coull 1989; Feller, Coull, and Hentschel 1990; Kneib 1993; Yozzo and Smith 1995), the sample processing and taxonomic resources required for validation of meiofaunal population characteristics are probably beyond the scope of HGM efforts. Tidal marsh meiofauna population dynamics are largely determined by the same factors that influence macrofaunal distribution (e.g., salinity, inundation frequency, vegetation characteristics, disturbance/predation); thus meiofauna probably do not warrant separate consideration from the macrofauna in the model.

The model only considers production of prey resources on the intertidal marsh surface, as this habitat is the primary forage habitat for natant predators such as killifish and grass shrimps. However, it should be recognized that the creeks and channels draining the marsh also contain a diverse and abundant infaunal community, often (in the case of tidal freshwater marshes) in association with submerged aquatic vegetation. In consideration of the widely accepted view of the intertidal marsh surface as an important source of energy for estuarine consumers (Bell and Coull 1978, Kneib and Stiven 1982, McIvor and Odum 1988, Kneib and Wagner 1994, Kneib 1997), and in maintaining consistency with the other HGM functions that focus primarily on processes occurring on the vegetated intertidal marsh surface, the model does not consider prey resources in subtidal habitats.

Macrofaunal production estimates typically require detailed information on size-frequency distributions and age-specific growth rates. However, assuming similar turnover rates, simple estimation of standing stocks (biomass) may reflect relative production of many small benthic macroinvertebrates (polychaetes, oligochaetes, ostracods, tanaids, amphipods, etc.) commonly found in tidal wetlands. For model validation purposes, it is feasible to obtain a relatively quick estimate of standing stocks rather than calculate secondary production.

Description of model variables

Hydrologic regime (V_{HYDRO}). Standing stocks of macrobenthic invertebrates in tidal marshes are controlled largely by the availability of suitable, moist habitat and the effect of aquatic predators. In the absence of predators, macrobenthic standing stocks should be relatively high in areas that are inundated regularly. See Function 2 for measurement and scaling of this variable.

Aquatic edge (V_{EDGE}). This variable is a direct linear measure of the amount of edge between the intertidal marsh surface and adjacent aquatic habitats. Intertidal and subtidal creeks and shallow embayments represent "staging areas" for natant marsh predators. A large amount of edge is assumed to provide these organisms greater access to foraging areas on the intertidal marsh surface. For a description of how to measure V_{EDGE}, refer to Function 4.

Total percent vegetative cover (V_{COVER}). This variable is a measure of the relative proportion of the site that is covered with emergent macrophytic vegetation. Vegetation provides structure that increases the available habitat and can mediate the effects of predation, so the presence of vegetation, especially dense vegetation, should have a positive effect on macrofaunal standing stocks.

Measure this variable using the following procedure.

a. Select one or more representative areas within the site for sampling. Beginning at the shoreward edge of the marsh, establish one or more 30-m transects perpendicular to the shoreline or along the hydrologic gradient (e.g., increasing elevation). If there are multiple vegetation community types within the WAA, the transect should intersect each vegetation community, in order to ensure a representative sample.

b. Using a standard 1-m^2 frame, estimate total percent cover by both live and dead emergent macrophytic plant species at intervals along the transect, excluding any areas where water is too deep to support the growth of emergent vegetation. The number of transects and plots needed will depend on the size and heterogeneity of the site; a minimum of 10 plots should be used. Beginning at the shoreward edge of the marsh, establish a 30-m transect perpendicular to the shoreline. Using a standard 1-m^2 frame, estimate total percent cover by emergent macrophytic plant species at regularly spaced intervals of 3 m along the transect, excluding tidal creeks. The number of transects needed will depend on the size and heterogeneity of the site; for some sites, one transect may be sufficient.

c. Calculate the average of all total percent cover estimates.

d. Using Figure 11, determine the variable subindex that corresponds to the mean percent cover estimate.

If multiple persons will be estimating vegetation cover, it would be prudent for the entire team to spend some time in the field prior to performing an assessment to ensure that estimates performed by different persons on the same plot are within an acceptable margin of error.

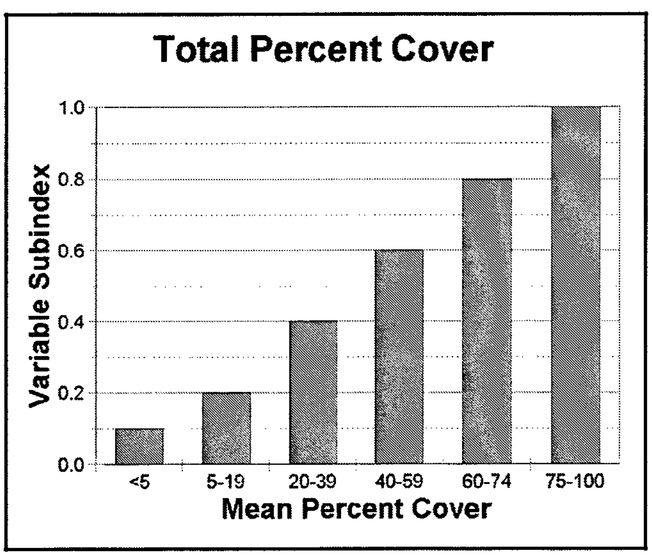

Figure 11. Relationship between mean total percent cover and functional capacity

Functional capacity index

The assessment model for calculating the functional capacity index is as follows:

$$FCI = (V_{HYDRO} + V_{EDGE} + V_{COVER})/3$$

The form of this model is a simple arithmetic mean, reflecting the assumed equal contributions of all three variables to total functional capacity.

Function 7: Provide Wildlife Habitat

Definition

This function describes the potential utilization of the marsh by resident and migratory avifauna, herpetofauna, and mammals. A quantitative measure of this function would be abundance of birds, herps, and mammals per unit area.

Rationale for selecting the function

A variety of birds, mammals, and herpetofauna, including many threatened or endangered species, utilize tidal fringe wetland habitats, either as permanent residents or occasional visitors. Many wildlife species are important consumers in tidal wetlands and may figure prominently as trophic links to adjacent terrestrial or aquatic/marine ecosystems.

Characteristics and processes that influence the function

This model is intended to represent the general habitat quality of tidal fringe wetlands for wetland-dependent species of avifauna, herpetofauna, and mammals, with the recognition that individual species within these groups may have different, even conflicting, habitat requirements. The great variability in habitat requirements makes founding a model upon presence of specific habitat type or measurements of habitat quality impractical. Use of tidal fringe wetlands by wildlife varies in terms of the type and number of activities in which the species engages (e.g., feeding, breeding) and the amount of time spent in the wetland (i.e., year-round residents to occasional visitors). Some species may spend their entire lives in marshes (e.g., clapper rails, rice rats), others migrate seasonally to breed or feed there, and still others are occasional users of these areas as stopover points during migration. Attempts to identify key factors governing the use of marshes by these organisms are further complicated by the differential use of elevation zones or portions of a marsh by different species or groups of species (e.g., wading birds, shorebirds) across either single or multiple purposes. For example, shorebirds and wading birds typically feed in different parts of a marsh: shorebirds preferring the open shoreline edge of a marsh, adjacent mudflats, or large tidal creeks; wading birds preferring the shallow water along creeks and pools. Species that breed and feed in marshes may use different zones for each activity; the clapper rail prefers to nest in low marsh zones, while it feeds across the entire marsh. Further, because the model considers all birds, herpetofauna, and mammals as a group, factors that favor one group may have the opposite effect on another. The presence of an adjacent upland, for example, may provide a suitable and beneficial high-tide refuge for mammals that may use the marsh as feeding grounds (e.g., raccoons, skunks), but at the same time, might exert a negative effect on ground-nesting birds that these mammals may prey upon. In addition, species and densities of wildlife will not be uniform across sites within the same subclass of coastal fringe wetland simply due to factors not related to the level of degradation or function impairment.

The number of species increases in direct relation to the size of an area sampled; therefore, as patch size (or wetland area) increases so does (generally) the number of species utilizing that wetland. Wetland shape, or the amount of wetland (or similar habitat) interior, is also important for the preservation of species that are adversely affected by the creation of edge habitats. An individual wetland that is functionally linked to the habitats surrounding it can be effectively enlarged with an abundance of nearby habitat similar in form (i.e., salt marsh and coastal prairie). This nearby similar habitat can be either directly connected to the WAA wetland or not connected, but is near enough for use by mobile species of wildlife. Direct connections would be important for species such as turtles, but not so important for more mobile species such as alligators.

Regional habitat subtypes selected for the variable V_{WHC} are based on their functional role in supporting wildlife communities (Table 13). Because the habitat types likely to be represented in the regional reference set are principally defined by vegetation type, this variable embodies a variety of concepts of importance to wildlife, including hydrology, exotic plants, vegetation structure, and refugia. Identification of habitat types should include the full range of habitat types and edge characteristic of the project vicinity, which may or may not be included in specific project boundaries. For example, marsh-dependent animals require refuge from low-tide events (pools), as well as high-tide events (hummocks and/or adjacent uplands).

Table 13
Examples of Habitat Types and Associated Wildlife

Habitat Type	Associated Species Groups
Submerged aquatic vegetation	Waterfowl
Unvegetated subtidal bottom	Wading birds, diving ducks, furbearers, herps
Shellfish beds	Wading birds, shorebirds, marine mammals
Mudflats	Wading birds, shorebirds, raptors, waterfowl
Other wetland subclasses	Marsh resident birds, marsh resident mammals, songbirds, waders, raptors, large shorebirds, furbearers, herps
Tidal ponds	Furbearers, herps, wading birds, shorebirds
Tidal channels	Wading birds, resident birds, shorebirds, furbearers, herps
Unvegetated pans	Shorebirds
Vegetated pans	Shorebirds
Slough (nontidal channel)	Waterfowl, herps, furbearers, wading birds, and shorebirds
Supratidal habitats (i.e., hummocks, large logs, muskrat lodges, prairie, forested or scrub-shrub uplands,etc.)	Refuge and nesting habitat for all groups

Description of model variables

This model quantifies habitat quality by examining factors related to the amount of degradation that has occurred to those attributes that affect all wildlife species, and to factors that are responsible for maintaining the wetland in its present, most sustainable, form in the face of a constantly changing coastal environment. Specifically, it has chosen landscape level factors identified by island biogeography theory as controlling species diversity and abundance across all taxonomic groups. These factors include patch size, connectedness, and distance to other patches of suitable habitat (other populations). These factors will hopefully reflect the level of man-induced habitat degradation. The model also includes two other variables that typically reflect the presence of environmental conditions resulting in degraded wildlife habitat. These degradation indicators are wildlife habitat complexity and percent cover of typical vegetation.

Total effective patch size (V_{SIZE}). This variable measures the size of the area that an animal is likely to be able to traverse during its daily movements without encountering significant barriers or risk of predation. This includes the core wetland patch of which the WAA is a part as well as any other patches of suitable habitat that are connected via wildlife corridors. For the purposes of this method, the core wetland is defined as a contiguous patch of tidal fringe wetland. Other wetlands connected to the core wetland via corridors may be of the same wetland subclass (e.g., tidal fringe) or other wetland subclasses.

The primary reason for this is that wildlife will utilize the entire wetland complex and will not be confined to or deterred by project boundaries. A single habitat patch rarely supplies all of the needs of a particular wildlife species throughout the year. A yearly home range may consist of one large habitat block but often consists of a collection of habitat patches. Predatory wildlife, especially, need large annual home ranges to avoid depleting prey populations. In addition, wildlife must access adjoining home ranges when breeding or dispersing.

Movement between habitat patches undertaken by dispersing individuals may occur over very large distances and is assumed to be unimpeded, as this wetland subclass always occurs near large water bodies that can afford long-distance travel. This type of movement is given no further consideration. On a more local scale, travel between patches is facilitated by the presence of corridors. Corridors are areas of unsuitable habitat that may afford the opportunity to travel between suitable patches with a minimal risk of mortality. The more formidable the barriers to movement incurred, the less likely it is that a wildlife species will be able to utilize these nearby habitat patches. Therefore, the contribution of a nearby wetland to total effective patch size is weighted according to the ability of different classes of wildlife (highly mobile and less mobile) to traverse between patches. The more classes of wildlife that are blocked by lack of an effective corridor, the lower the multiplier value.

This variable's subindex decreases as the total effective patch size decreases and is scaled to the daily home ranges of different classes of wildlife. A tract that supplies all the needs of the animal with the largest daily home range is considered the reference standard condition, while a tract that supplies only the minimal needs of a transitory species is assigned a lower subindex. The needs and abilities of the following four wildlife classes were considered in scaling this variable:

a. A highly mobile animal (e.g., river otter (*Lutra canadensis*)) with a large home range.

b. A moderately mobile animal (e.g., clapper rail (*Rallus longirostris*)) with a moderate home range.

c. A weakly mobile animal (e.g., marsh wren (*Cistothorus palustris*)) with a small home range.

d. A highly mobile animal that uses the wetland subclass only as one of several possible foraging habitats (e.g., great egret (*Casmerodius albus*)).

This variable is measured using the following procedure.

a. Using aerial photography, determine the patch size (in hectares) of the core wetland. The core wetland includes the entire contiguous tidal fringe wetland. For instance, if the wetland assessment area encompasses only 2 ha of a 100-ha tidal fringe wetland, record the core wetland size as 100 ha. For the purposes of this function, tidal flats as well as tidal creeks and ponds less than 1 m deep are included out to a distance of 20 m from the emergent wetland edge. This not only makes assessing V_{SIZE} in highly interspersed marshes possible with aerial photography, it also takes into account aquatic habitats heavily used by wildlife especially during winter when tidal flats and creek banks are exposed for long periods.

b. Next, determine the size (in hectares) of any patches of other wetlands that are connected to the above patch via wildlife corridors. For each of these wetland patches, assess the degree of corridor connectivity according to the descriptions in Table 14.

c. Multiply the size of each of these wetland patches by the appropriate multiplier according to the degree of corridor connectivity (Table 14).

d. To determine the total effective patch size, sum the products obtained in Step *c* and add this number to the initial patch size recorded in Step *a*.

e. Assign a variable subindex for total effective patch size based on the information in Table 15.

Table 14
Assessing the Degree of Corridor Connectivity

Corridor Type	Corridor Description	Multiplier
Contiguous corridor	**1)** Open water stretches <60 m (regardless of depth). **2)** Unvegetated stretches of shoreline or strips of other wetland subclasses <60 m in length that have an aquatic shelf at least 3 m wide and are <0.3 m deep at MSL. This discounts most tidal creeks and coves or unvegetated stretches of shoreline abutting uplands as barriers to wildlife that are traveling through their daily home range.	1.00
Partially impeded corridor	**1)** Open water stretches from 60-300 m (regardless of depth). **2)** Unvegetated shorelines or strips of other wetland subclasses from 60-500 m in length that have an aquatic shelf at least 3 m wide with water depths <0.3 m at MSL. Deeper stretches of water that interrupt the shelves are not considered impeding if they are <60 m wide. **3)** Stretches of undeveloped upland that are <30 m in width.	0.75
Impeded corridor	**1)** Shoreline shelves or wetland strips 500-1200 m long. **2)** Stretches of undeveloped upland 30-300 m in width.	0.50
Corridor absent or barrier present	**1)** Open water stretches or undeveloped upland >300 m in width. **2)** Shorelines >300 m long that contain no shelf with waters 0.3 m deep (i.e., long stretches of bulkheading). **3)** Roadways with >100 vehicle crossings per day that are unbridged or have a bridge opening <3 m wide. **4)** Highly developed urban, residential, or industrial areas.	0.0

Table 15
Relationship Between Total Effective Patch Size (V_{SIZE}) and Functional Capacity

Total Effective Patch Size	Variable Subindex
>200 ha	1.00
5-200 ha	0.75
1-5 ha	0.50
0.2-1 ha	0.25
<0.2 ha	0.10

Table 16
Possible Wildlife Habitat Types

Low marsh (i.e., *Spartina alterniflora*)	Oyster reef
High marsh (i.e., *Spartina patens, Batis maritima*)	Unvegetated flats
Subtidal creeks/channels	Coarse woody debris
Intertidal creeks/channels	Algal mats
Ponds or depressions	Mangroves
Submerged aquatic vegetation (i.e., *Ruppia maritima, Halodule wrightii*)	Forested uplands
Unvegetated beach berm	Shrub/scrub
Supratidal habitats (hummocks, logs)	Pasture/Grasslands

Wildlife habitat complexity (V_{WHC}). Habitat heterogeneity is believed to increase the diversity of wildlife species utilizing a site. This variable is a measure of the heterogeneity of a site, based on comparing the number of habitats actually present at a site to the number of possible habitats known to occur in or adjacent to the appropriate regional subclass (Table 16). Separate variables have been defined for V_{NHC} (nekton habitat complexity) and V_{WHC} (wildlife habitat complexity) to reflect differential usage of available habitats by these faunal groups. This variable should help to identify sites that have been degraded by human activity or are not providing the greatest level of this function possible for the hydrogeomorphic setting present. As an example: coastal marshes suffering from man-induced subsidence often lack habitat types such as supratidal marshes or vegetated ponds. Similarly, created coastal marshes usually lack tidal access, aquatic edge, and channels and ponds. The variable is a simple measure of the habitat types listed in Table 16 that are present in or within a 2-km radius of the project area perimeter.

Measure V_{WHC} using the following procedure.

a. Identify the total number of habitat types listed in Table 16 that are present in or within a 2-km radius of the project area perimeter.

b. Assign a variable subindex based on Figure 12.

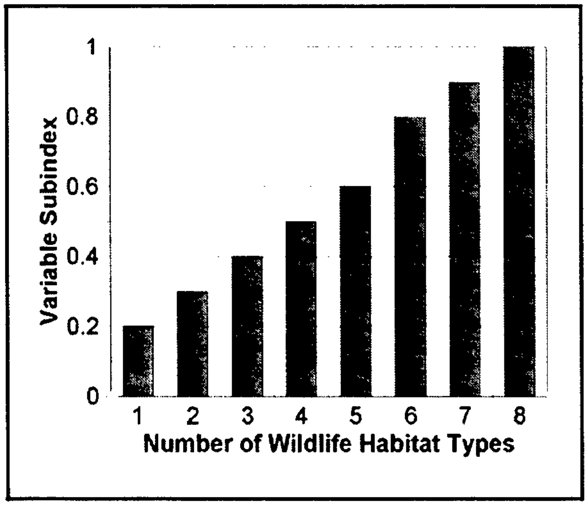

Figure 12. Relationship between wildlife habitat complexity and functional capacity

Total percent vegetative cover (V_{COVER}) or percent cover by typical species ($V_{TYPICAL}$). Wildlife need native plant communities to secure forage, nesting substrates, and sheltering areas. Reduced plant growth will result in less primary productivity and ultimately fewer grazing wildlife species or fewer herbivorous prey species for predatory wildlife. Loss of plant cover also adversely impacts the ability of a habitat to provide shelter and nesting substrates. The presence of atypical plant species may adversely affect herbivorous wildlife species or the prey pool of predatory wildlife as plants vary widely in their quality as forage. In addition, many species of wildlife nest only in specific plant communities. For example, mottled ducks prefer to nest in the dense grass of native salty prairies. The lack of vegetative cover or the presence of atypical plant species may reflect recent heavy disturbance, pollution, or incorrect hydrology or soil conditions. This often translates into reduced wildlife usage or a reduction in the number of species of wildlife present in a habitat. See Functions 6 and 8 for descriptions of the measurement of these variables.

Functional capacity index

The assessment model for calculating the functional capacity index is as follows:

$$FCI = [2\ V_{SIZE} + V_{WHC} + (\text{Minimum}\ (V_{TYPICAL}\ \text{OR}\ V_{COVER}))]/4$$

Due to the relative uniformity within the estuarine fringe plant communities (e.g., lack of vegetative diversity and successional changes in plant community structure), wildlife habitat value is mostly controlled by patch size and other landscape scaled features, such as connectivity to other patches, captured in the V_{SIZE} variable. Therefore, the variable V_{SIZE} is given greater weight than the other variables in computing the functional capacity index. The habitat complexity variable measures the presence of important physical and biotic habitat features, while hydroperiod, plant community composition, and percent cover are measured directly or indirectly by the $V_{TYPICAL}$ and V_{COVER} variables.

Function 8: Maintain Characteristic Plant Community Composition

Definition

This function describes the ability of a wetland to support a native plant community of characteristic species composition. Community composition of wetlands is so varied in the region that there is no suite of species that may be considered characteristic of a reference standard. A wide group of assemblages would have to be treated as representative. Any combination of species or even a single species typical of a subclass (see subclass profiles for a listing of these species) can be the reference standard, if constituting the entire vegetation of a site, but the higher the contribution of exotic or nuisance species or even species characteristic of a different marsh subclass, the lower the functional capacity index will be.

Rationale for selecting the function

The vegetative community is one of the fundamental components of both terrestrial and wetland ecosystems. Changes in the plant species composition and structure may profoundly affect the entire suite of physical, chemical, and biological processes occurring within a site. Although these attributes have already been considered through the incorporation of these variables into many of the other functional indices, maintenance of a characteristic native plant community was deemed sufficiently important to warrant separate consideration.

Characteristics and processes that influence the function

The number of plant species that are able to exist in salt marshes is limited due to environmental stress factors such as the duration, frequency, and depth of flooding and high pore water salinity levels. Salt marsh vegetation is typically dominated by grasses (Poaceae) and sedges (Cyperaceae) or a combination of these families. The plants typically occur in well-defined zones dominated by a single species or species association. Tidal fringe marshes lack the complex multi-layered structure characteristic of forested communities; although a scrub-shrub component may exist, it usually occurs at the upland edges or on elevated hummocks and occupies only a small proportion of the total area. The spatial extent of the major zones of vegetation is largely determined by elevation and the resultant effect on the tidal flooding regime.

Changes in the extent of aerial coverage and species composition of tidal marshes may occur as a direct result of altered hydrology, such as dikes, channels, or impoundments. These changes affect the salinity regime, flooding frequency and flooding duration, and may cause an increase in the extent of brackish species such as *Typha domingensis*, at the expense of more salt-tolerant species such as *Spartina alterniflora* (Sinicrope et al. 1990). Such conditions may also allow the introduction and spread of non-native or undesirable species, such as *Phragmites australis* (Roman, Niering, and Warren 1984).

Smooth cordgrass (*Spartina alterniflora*) is the dominant plant in the intertidal zone along the Atlantic coast and the western Gulf of Mexico. This species generally occurs between mean high water and mean low water and exhibits considerable variation in growth form (i.e., tall, medium, and short), as determined primarily by tidal flooding frequency and duration. Above mean high water, floral composition of salt marshes increases in diversity and varies with latitude. Common species include saltmeadow cordgrass (*S. patens*), saltgrass (*Distichlis spicata*), blackgrass (*Juncus gerardi*), and black needlerush (*Juncus roemerianus*). Unvegetated salt pannes are common intertidal landscape features in Atlantic and Gulf Coast salt marshes, and these pannes may be fringed by halophytes such as glasswort (*Salicornia spp.*) and saltwort (*Batis maritima*).

Brackish marshes generally occur in association with freshwater input from coastal rivers and bayous. Depending on the amount of freshwater input and its effect on the local salinity regime, these marshes may be dominated by either smooth cordgrass or big cordgrass (*S. cynosuroides*). Bulrush (*Scirpus americana*) and pickerelweed (*Pontederia cordata*) may also be present in mixed stands associated with big cordgrass.

Description of model variables

Total percent vegetative cover (V_{COVER}). Extremely low percent vegetative cover values may indicate a number of different undesirable conditions including (a) a subsiding or deteriorating marsh, (b) presence of toxins or other pathological condition, or (c) incorrect elevation range in created marshes. See Function 6 for a description of variable measurement.

Percent cover by typical species ($V_{TYPICAL}$). Nonnative or invasive species are considered indicators of site degradation due to nutrient enrichment or other types of anthropogenic disturbance. This variable serves to downgrade the value of the functional index as the proportionate contribution of typical species for the subclass decreases. Table 17 lists those non-native or potentially undesirable species identified by the regional A-team.

For all reference wetlands sampled in the Galveston Bay area, the percent cover by typical native species was greater than 95 percent, indicating that exotic or invasive species are not a major problem for most tidal fringe wetlands in this region.

Table 17 **Possible Invasive or Undesirable Plant Species**			
Scientific Name	**Common Name**	**Salt/Brackish**	**Intermediate**
Alternanthera philoxeroides	Alligator weed		L
Aster spinosus	Spiny aster		H
Phragmites australis	Common reed	H	L,H
Sesbania drummondii	Drummond's rattlebush		L,H
Typha spp.	Cattail	L	L
H = high marsh, L = low marsh.			

Measure this variable using the following procedure.

a. Visually estimate the percentage of the site that is covered by nontypical, non-native, or otherwise undesirable plant species (See Table 17). Subtract this number from 100 to estimate the percentage of the site that is occupied by plant species typical of the regional subclass. See the subclass profile in Chapter 2 for additional information. Appendix D also lists typical plant species that occur in saline, brackish, and intermediate marshes along the Texas coast.

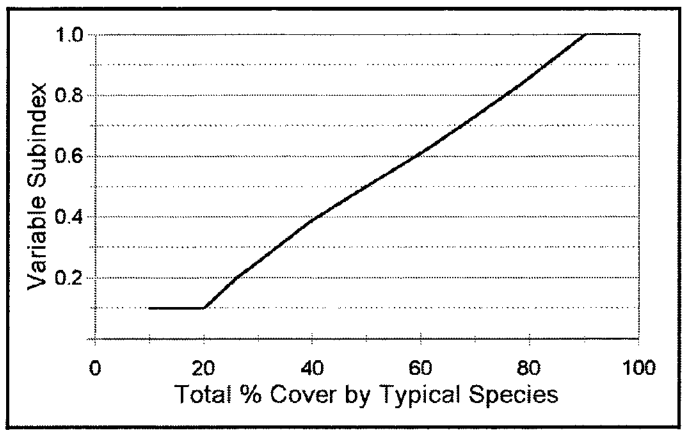

Figure 13. Relationship between percent cover by typical plant species and functional capacity

b. Assign a variable subindex based on Figure 13.

Functional capacity index

The assessment model for calculating the functional capacity index is as follows:

$$FCI = (\text{Minimum } (V_{COVER} \text{ or } V_{TYPICAL})$$

These variables are intended to downgrade the value of the function as a result of (a) plant community composition that differs from that of the regional subclass ($V_{TYPICAL}$), or (b) abnormally low vegetative cover, which may result from a number of different causes (V_{COVER}). Since either of these conditions could contribute to site degradation, the value of the functional index is set to the lower of the two variable subindices.

Function 9: Plant Biomass Production

Definition

This function estimates standing crop as an indicator of the potential for a site to produce plant material. A quantitative measure of this function would be plant biomass per square meter, assumed to be directly related to plant biomass produced per square meter per year.

Rationale for selecting the function

The high productivity of coastal marshes and the physical structure that is one expression of that productivity are the basis for the transformations of matter and energy that we refer to as wetland functions. Thus, this function appears in its entirety as one component of the Nutrient and Organic Carbon Exchange model on the grounds that the biological activity associated with plant growth and structure is the driving force behind the transformation of materials being addressed by that function. Plant biomass and production are similarly important in support of animal assemblages. Although these support roles are captured at least in part in other functions, marsh biomass and biomass production were deemed important in their own right.

Characteristics and processes that influence the function

The high productivity of coastal marshes has long been recognized (Sather and Smith 1984). Although the combination of long periods of soil saturation and variable salinities excludes most plants, those that can tolerate the conditions are the beneficiaries of nutrient subsidy and waste removal afforded by periodic flooding and emersion. The result is a potential for high primary productivity.

Although many factors clearly influence primary productivity (e.g., nutrient availability, sediment properties, soil aeration), only a measure of the standing stock of plant material on a site is included as a model variable. It is assumed that standing crop is a sensitive integrator of all other influences on primary production and is the proximate factor that most directly defines the potential of a site for primary production. Furthermore, direct measurement of nutrient availability and edaphic features, or at the other extreme, of primary productivity itself, is beyond the scope of applications of this methodology.

For wetlands of the same type, it is generally assumed that more pristine, less modified examples will be more productive. This may not hold in the case of wetlands receiving waters elevated in nutrients from sewage discharges or runoff from agricultural operations. Here, very high productivity and standing crop may be indicative of a degenerative condition that is not sustainable.

Description of model variable

Vegetative structure (V_{VEGSTR}). This variable is a composite of cover and height summed over all species. See Function 3 for a description of variable measurement.

Functional capacity index

The assessment model for calculating the functional capacity index is as follows:

$$FCI = V_{VEGSTR}$$

This function contains only a single variable because this variable integrates vegetation structural characteristics that contribute to biomass production; namely, cover and height aggregated over all species. Although this variable is included as a component in other models, it is included here to allow separate comparison of biomass production potential among sites.

Assign a variable subindex using Figure 9.

5 Assessment Protocol

Introduction

Previous sections of this Regional Guidebook provide background information on the HGM Approach and document the variables, measures, and models used to assess the functions of tidal fringe wetlands in the Northwestern Gulf of Mexico. This chapter outlines a protocol for collecting and analyzing the data necessary to assess the functional capacity of a wetland in the context of a 404 permit review or similar assessment scenario.

The typical assessment scenario is a comparison of pre-project and post-project conditions in the wetland. In practical terms, this translates into an assessment of the functional capacity of the WAA under both pre-project and post-project conditions and a subsequent determination of how the FCIs have changed as a result of the project. Data for the pre-project assessment are collected under existing conditions at the project site, while data for the post-project assessment are normally based on the conditions that are expected to exist following proposed project impacts. A conservative and well-documented approach is required in defining post-project conditions. This recommendation is based on the often observed lack of similarity between predicted and actual post-project conditions.

This chapter discusses each of the tasks required to assess tidal fringe wetlands in the northwestern Gulf of Mexico, including:

a. Define assessment objectives.
b. Characterize the project area.
c. Screen for red flags.
d. Define the wetland assessment area.
e. Collect field data.
f. Analyze field data.
g. Apply assessment results.

Define Assessment Objectives

Begin the assessment process by identifying the purpose of conducting the assessment. This may be as simple as stating, "The purpose of this assessment is to determine how the proposed project will impact wetland functions." Often, there will be multiple purposes for conducting the assessment. Other potential objectives include: (a) comparing several wetlands as part of an alternatives analysis, (b) identifying specific actions that could be taken to minimize project impacts, (c) documenting baseline conditions at the wetland site, (e) determining mitigation requirements, (f) determining mitigation success, or (g) evaluating the effects of a wetland management technique. Defining the purpose will facilitate communication and understanding between the people involved in conducting the assessment and will make the purpose clear to other interested parties. In addition, it will help to establish the approach that is taken. The specific approach will vary, depending on whether the project is a Section 404 permit review, an Advanced Identification (ADID), a Special Area Management Plan (SAMP), or some other scenario.

Characterize the Project Area

Characterizing the project area involves describing the project area in terms of climate, surficial geology, geomorphic setting, tidal flooding regime, vegetation, soils, land use, proposed impacts, and any other characteristics and processess that have the potential to influence how wetlands at the project area perform functions. The characterization should be written and should be accompanied by maps and figures that show project area boundaries, buildings, jurisdictional wetlands, the WAA, proposed impacts, roads, ditches, streams, soil types, plant communities, threatened or endangered species habitats, and other important features.

The following list identifies some information sources that will be needed in order to characterize a project area.

a. Recent aerial photographs or digital ortho-photo quadrangle imagery.
b. Topographic and National Wetlands Inventory maps.
c. County Soil Survey.

Screen for Red Flags

Red flags are those features within or near the project area to which special recognition or protection has been assigned on the basis of objective criteria (Table 18).

Table 18
Red Flag Features and Respective Program/Agency Authority

Red Flag Features	Authority[1]
Native lands and areas protected under American Indian Religious Freedom Act	A
Hazardous waste sites identified under CERCLA or RCRA	I
Areas protected by a Coastal Zone Management Plan	B, E, L
Areas providing Critical Habitat for Species of Special Concern	B, C, F
Areas covered under the Farmland Protection Act	K
Floodplains, floodways, or floodprone areas	J
Areas with structures or artifacts of historic or archeological significance	A, D, G
Areas protected under the Land and Water Conservation Fund Act	K
Areas protected by the Marine Protection Research and Sanctuaries Act	B, D
National wildlife refuges and special management areas	B, C, D
Areas identified in the North American Waterfowl Management Plan	C
Areas identified as significant under the RAMSAR treaty	C
Areas supporting rare or unusual plant communities	C, F
Areas designated as Sole Source Groundwater Aquifers	I
Areas protected by the Safe Drinking Water Act	I
City, County, State, and National Parks	C, D, F, G, L
Areas supporting threatened or endangered species	B, C, E, F, I
Areas with unique geological features	D
Areas protected by the Wild and Scenic Rivers Act	C, D
Areas protected by the Wilderness Act	C, D

[1] Program Authority/Agency
A = Bureau of Indian Affairs
B = National Marine Fisheries Service
C = U.S. Fish and Wildlife Service
D = National Park Service
E = State Coastal Zone Office
F = State Dept. of Natural Resources, Fish and Game, etc.
G = State Historic Preservation Officer
H = State National Heritage Offices
I = U.S. Environmental Protection Agency
J = Federal Emergency Management Administration
K = Natural Resource Conservation Service
L = Local Government Agencies

Many red flag features, such as those based on national criteria or programs, are similar from region to region. Other red flag features are based on regional or local criteria. Screening for red flag features represents a proactive attempt to determine if the wetlands or other natural resources in and around the project area require special consideration or attention that may preempt or postpone an assessment of wetland function. The assessment of wetland functions may not be necessary if the project is unlikely to occur as a result of a red flag feature. For example, if a proposed project has the potential to impact a threatened or endangered species or habitat, an assessment of wetland functions may be unnecessary since the project may be denied or modified strictly based on the impacts to threatened or endangered species habitat.

Define the Wetland Assessment Area

The WAA is an area of wetland within a project area that belongs to a single regional wetland subclass and is relatively homogeneous with respect to the site-specific criteria used to assess wetland functions (i.e. hydrologic regime, vegetation structure, topography, soils, successional stage, etc.). In many project areas, there will be just one WAA representing a single regional subclass. However, as the size and heterogeneity of the project area increase, it is more likely that it will be necessary to define and assess multiple WAAs within a project area.

At least two situations necessitate defining and assessing multiple WAAs within a single project area. The first situation exists when spatially separate patches of the same regional subclass occur within the project area. The second exists when a physically contiguous wetland area of the same regional subclass exhibits spatial heterogeneity with respect to hydrology, vegetation, soils, disturbance history, or other factors that translate into a significantly different value for one or more of the site-specific variable measures. These differences may be the result of natural variability or anthropogenic alteration. Designate each of these areas as a separate WAA and conduct a separate assessment on each area.

There are elements of subjectivity and practicality in determining what constitutes "significant" differences in portions of the WAA. Field experience with the regional wetland subclass under consideration should provide a sense of the range of variability that typically occurs and the background necessary to make reasonable decisions about defining multiple WAAs. In general, differences resulting from natural variability should not be used as a basis for dividing a contiguous wetland into multiple WAAs.

Data Collection

The following equipment is necessary to measure or estimate values for model variables:

a. A 30-m measuring tape.
b. A 1-m^2 quadrat for estimating plant percent cover.
c. Recent color infrared aerial photographs or digital ortho-photo quadrangle imagery at a scale of approximately 1 cm = 48 m (1:4800 or 1 in. = 400 ft).

d. Clear mylar overlay sheet marked with the 16 cardinal and subcardinal compass bearings.
e. English area grid.
f. Measuring stick marked in centimeters.
g. Soil probe or sharpshooter shovel.
h. Bathymetry charts.
i. National Wetlands Inventory maps.

Although this method is designed for use by those without access to GIS mapping software, use of a computer mapping software package such as ArcView or ArcInfo will greatly facilitate the measurement of some model variables.

Guidance on How to Use the Model Outputs

The HGM Approach is designed to reflect the long-term sustainability and functional capacity of the wetland, relative to other similar wetland types in the region. It incorporates the assumption that wetlands provide many different functions, a subset of which were selected by the regional A-Team for inclusion in the assessment models. The HGM Approach assumes that not all wetlands perform the same functions or perform those functions to the same level of performance. Although these functions are often intricately interconnected to establish the character of the wetland ecosystem, we must recognize that the state of our knowledge concerning these linkages between the different wetland functions is limited.

The output from the HGM Approach is an index score of 0.0 to 1.0 for each function identified by the A-Team. The HGM Approach has been occasionally criticized for providing results for each function but not providing an approach to combine all functional scores into a single, "bottom line" number, that represents the overall functional capacity of the wetland. Several options are available to create a "bottom line" number to represent the wetland functional capacity, but one must recognize the potential dangers in exercising each option. One option would be to compute an average of all scores. However, since some variables occur in several functional models, these variables could, in effect, be counted multiple times, producing a weighting effect on the final result. Consider another situation in which one function, such as one related to hydrology, may be critical to the sustainability of the wetland. If simply averaged with scores for other functions, its influence on the outcome of the wetland assessment might be minimized. Another option might be to use the result of the lowest scoring wetland funtion to represent the "bottom line" number for the wetland since that might be considered the "weakest link" in the sustainability of the wetland.

In addition to concerns about combining functional scores into a single score to represent the functional capacity of a wetland, some have proposed designing mitigation plans to maximize the score of a single function. This too has its limitations and is counter to the assumptions in the HGM

Approach. One cannot design a mitigation plan to maximize one single function without influencing the ability of the wetland to perform other functions, and therefore, the ecological integrity and sustainability of the wetland could be compromised. For example, one may want to maximize the water storage capacity of a mitigation site, yet doing so could result in nothing more than a very large hole with little semblance to the wetland for which the mitigation was proposed. In another example, one may wish to maximize waterfowl production by providing consistent, shallow water over an area. the result could be a totally different plant community from that which originally existed. As shown in these examples, attempts to maximize an individual function could result in a site that no longer meets the definition of a wetland or one that has little likelihood of sustainability.

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Appendix B
Field Data Forms

FIELD DATA SHEET:
NORTHWEST GULF OF MEXICO TIDAL FRINGE MARSHES

Assessment Team: ____

Project Name/Location: ______________________________ **Date:** __________

Prior to conducting an assessment, establish the project area boundaries and delineate the wetland boundaries within the project area.

Sample variables 1-3 using aerial photos at a scale of 1 cm = 48 m (1: 4800 or 1 in. = 400 ft, digital ortho-photo quadrangle imagery, maps, etc.)

1. V_{EDGE} Degree of marsh dissection/edge:area ratio *Subindex* ________

(1) Using either the quantitative or qualitative approach, measure or estimate the edge:area ratio and assign a subindex value based on Table B1. See pictorial key in Appendix D (Figures D2-D9) for specific examples.

Table B1
Relationship Between Edge:Area and Functional Capacity

Site Description	**Edge:Area** Qualitative	**Edge:Area** Quantitative m/ha	**Subindex**
1) Marsh shows signs of deterioration due to subsidence (i.e., highly fragmented with large amounts of open water. Vegetation occurs mainly in isolated hummocks or on natural levees along tidal creeks). Although edge:area is very high, this condition is not considered sustainable in the long term (Figure D2).	Very High	>800	0.8
1) Well-developed tidal drainage network present (Figure D3), OR **2)** Simple tidal drainage network (may consist of only a single channel) present with isolated ponds and depressions present in the marsh interior (Figures D4 and D5). **3)** Atypical geomorphic configuration with a large amount of shoreline relative to total area (i.e. small island or narrow peninsula) (Figure D7).	High	350-800	1.0
1) Simple tidal drainage network (may consist of only a single channel). Isolated ponds and depressions are few or lacking, OR **2)** Narrow fringe marsh that lacks tidal creeks. One lengthwise shoreline is exposed to tidal waters. Area of marsh is small relative to shoreline length (Figures D6 and D8).	Moderate	200-350	0.7
Marsh lacks both tidal creeks and isolated ponds and depressions. Shoreline is generally linear or smooth curvilinear without embayments or convolutions. Area of marsh is large relative to shoreline length (Figure D9).	Low	<200	0.4

2. V_{OMA} Proportion of tidally connected edge to total edge *Subindex* __________

(1) Assign subindex value based on Table B2.

Table B2
Relationship Between Opportunity for Marsh Access and Functional Capacity

Tidally Connected Edge: Total Edge	Subindex
❑ 50-100%	1.0
❑ 35-50%	0.7
❑ 25-35%	0.5
❑ 1-25%	0.2
❑ No tidally connected edge present	0.0

3. V_{SIZE} Total Effective Patch Size *Subindex* __________

(1) If the core wetland size exceeds 200 ha, assign a variable subindex value of 1.0. The core wetland is defined as a contiguous patch of tidal fringe wetland that contains the WAA.

(2) If the core wetland size <200 ha, identify other patches of wetlands in the surrounding area, and record the size of each patch in hectares. These wetlands may be in a wetland subclass other than tidal fringe. Then, using the descriptions provided in Table B3, determine the degree of connectivity between each wetland patch and the core wetland.

Table B3
Determination of Corridor Connectivity

Corridor Type	Corridor Description	Multiplier
Contiguous corridor	**1)** Open water stretches <60 m (regardless of depth). **2)** Unvegetated stretches of shoreline or strips of other wetland subclasses <60 m in length that have an aquatic shelf at least 3 m wide and are <0.3 m deep at MSL. This discounts most tidal creeks and coves or unvegetated stretches of shoreline abutting uplands as barriers to wildlife that are traveling through their daily home range.	1.00
Partially impeded corridor	**1)** Open water stretches from 60-300 m (regardless of depth). **2)** Unvegetated shorelines or strips of other wetland subclasses from 60-500 m in length that have an aquatic shelf at least 3 m wide with water depths <0.3 m at MSL. Deeper stretches of water that interrupt the shelves are not considered impeding if they are <60 m wide. **3)** Stretches of undeveloped upland that are <30 m in width.	0.75
Impeded corridor	**1)** Shoreline shelves or wetland strips between 500-1200 m long. **2)** Stretches of undeveloped upland 30-300 m in width.	0.50
Corridor absent or barrier present	**1)** Open water stretches or undeveloped uplands >300 m in width. **2)** Shorelines >300 m long that contain no shelf with waters <0.3 m deep (i.e., long stretches of bulkheading). **3)** Roadways with >100 vehicle crossings per day that are unbridged or have a bridge opening <3 m wide. **4)** Highly developed urban, residential, or industrial areas.	0.0

(3) Multiply the size of the patch (ha) by the appropriate connectivity multiplier from the table above.

Size of Wetland (hectares)		Connectivity Multiplier	Product
Core wetland	______	1.0	1.0
Patch A	______	______	______
Patch B	______	______	______
Patch C	______	______	______
Patch D	______	______	______
Patch E	______	______	______

(4) Obtain the sum of all the products above. SUM ______

(5) Using Table B4, assign a variable subindex based on the value of the sum calculated above.

Table B4
Relationship Between Total Effective Patch Size and Functional Capacity

Total Effective Patch Size	Subindex
>200 ha	1.00
5-200 ha	0.75
1-5 ha	0.50
0.2-1 ha	0.25
<0.2 ha	0.10

Sample variables 4-8 based on an onsite field inspection of the project area and WAA.

4. V_{HYDRO} Hydrologic regime *Subindex* _________

(1) Assign subindex value based on Table B5.

Table B5
Relationship Between Hydrologic Regime and Functional Capacity

Site Condition	Subindex
Site is open to free exchange of tidal waters. No obvious hydrologic alteration or restrictions present.	1.0
Moderate hydrologic restriction present (i.e., presence of low-elevation berm, which is frequently overtopped by high tide events or has multiple breaches or large culverts).	0.6
Severe hydrologic restriction present (i.e., presence of high-elevation berm, which is infrequently overtopped by high tide events or has a single opening, breach or small culvert).	0.3
Site receives tidal floodwaters only during extreme storm tide events.	0.1
Site is isolated from tidal exchange. The principal source of flooding is water sources other than tidal action (i.e., precipitation or groundwater). *Note: If this condition exists, another wetland assessment model should be strongly considered unless the site was formerly a tidal wetland prior to hydrologic modification.*	0.0

5. $V_{TYPICAL}$ Percent cover by typical plant species within the WAA _______%

(1) Visually estimate the percentage of the site that is covered by nontypical, nonnative, or otherwise undesirable plant species (see Table B6). Subtract this number from 100 to estimate the percentage of the site that is occupied by plant species typical of the regional subclass. See the subclass profile in Chapter 2 for additional information. Appendix D also lists typical plant species that occur in saline, brackish, and intermediate marshes along the Texas coast.

Table B6
Possible Invasive or Undesirable Plant Species

Scientific Name	Common name	Salt/Brackish	Intermediate
Alternanthera philoxeroides	Alligator weed		L
Aster spinosus	Spiny aster		H
Phragmites australis	Common reed	H	L,H
Sesbania drummondii	Drummond's rattlebush		L,H
Typha spp.	Cattail	L	L
H = high marsh, L = low marsh.			

(2) Assign variable subindex based on Figure B1 *Subindex* _________

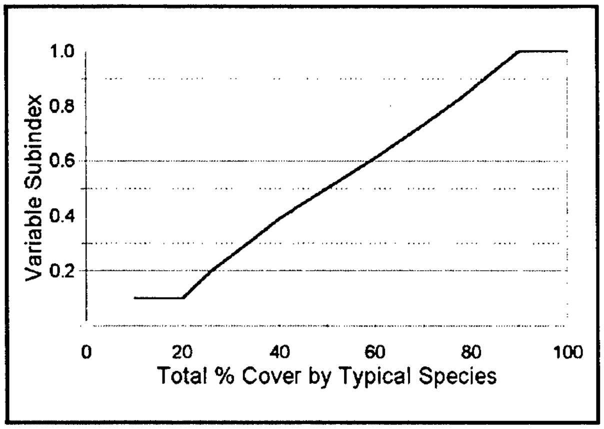


Figure B1. Relationship between percent cover by typical plant species and functional capacity

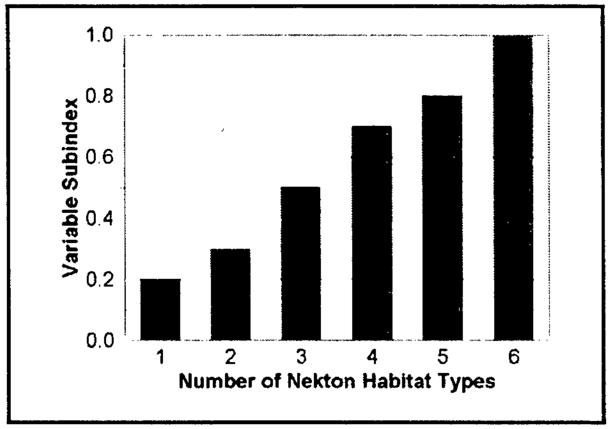


Figure B2. Relationship between nekton habitat complexity and functional capacity

6. V_{NHC} Nekton Habitat Complexity (# different habitat types)

(1) Check the habitats observed on or within a 30-m (100-ft) radius of project area perimeter.

Coarse woody debris	_____	Unvegetated flats	_____	Algal mats	_____
Subtidal creeks/channels	_____	Oyster reef	_____	Mangroves	_____
Intertidal creeks/channels	_____	Low marsh	_____	High marsh	_____
Ponds or depressions	_____	Submerged aquatic vegetation	_____		

(2) Assign variable subindex based on the chart in Figure B2. *Subindex* _________

7. V_{WHC} Wildlife Habitat Complexity (total # different habitat types)

(1) Check those habitat types **IN ADDITION TO THOSE LISTED ABOVE** that are present on or within a 2-km radius of the project area perimeter.

Supratidal habitats (hummocks, logs)	_______	Scrub-Shrub	______	Forested uplands	_____
Unvegetated beach	_______	Grasslands	______		

(2) Assign variable subindex based on the chart in Figure B3. *Subindex* _________

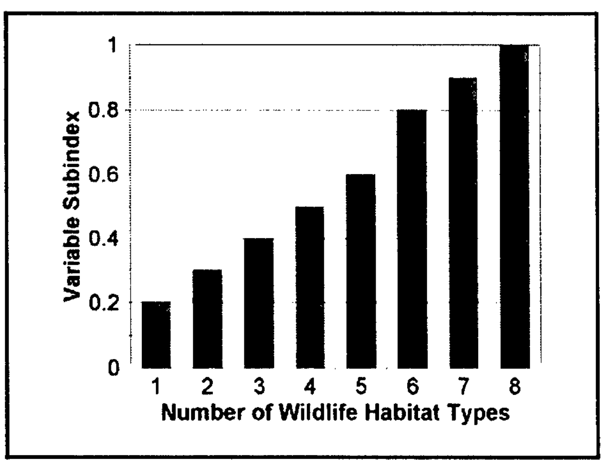


Figure B3. Relationship between wildlife habitat complexity and functional capacity

8. V_{ROUGH} Surface roughness (Manning's *n*)

(1) Choose a value for each of the three variables in the equation below based on the descriptions provided in Table B7.

$$\_\_\_\_ + \_\_\_\_ + \_\_\_\_ = \_\_\_\_$$
$$n_{BASE} + n_{TOPO} + n_{VEG} = n$$

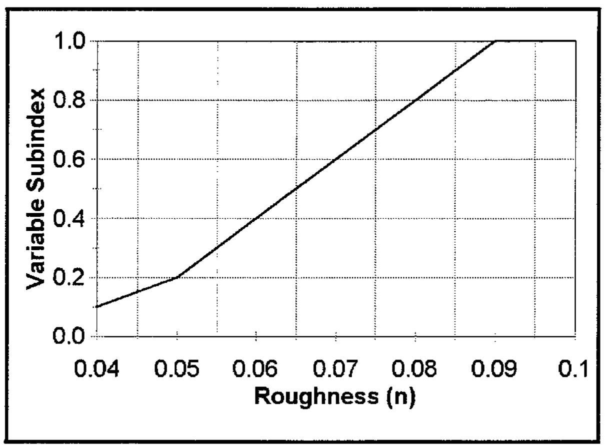
(2) Compute the sum of the three variables in the equation above.

(3) Assign variable subindex based on the chart in Figure B4. *Subindex* ________

Table B7
Relationship Between Roughness and Functional Capacity

Roughness Component	Adjustment to *n* Value			Description of Terms
Sediment surface (n_{BASE})	0.025			Base value for bare marsh soil.
	0.03			More than 25% of sediment surface covered with gravel or broken shell.
Topographic relief (n_{TOPO})	0.001			Representative area is flat with essentially no microtopographic (i.e., hummocks) or macrotopographic relief (i.e., berms, tidal channels, ridges and swales, ponds).
	0.005			Microtopographic (i.e., hummocks) or macrotopographic relief (i.e., berms, tidal channels, ridges and swales, ponds) cover 5-25% of a representative area.
	0.010			Microtopographic (i.e., hummocks) or macrotopographic relief (i.e., berms, tidal channels, ridges and swales, ponds) cover 26-50% of a representative area.
	0.020			Microtopographic (i.e., hummocks) or macrotopographic relief (i.e., berms, tidal channels, ridges and swales, ponds) cover >50% of a representative area.
Roughness Component	**Percent Cover**			
	<50	**50-75**	**76-100**	**Description of Conditions**
Vegetation (n_{VEG})	0.025	0.030	0.035	Representative area predominantly short, flexible-stemmed grasses (i.e., short *Spartina alterniflora, S. patens, Distichlis spicata*).
	0.035	0.040	0.050	Vegetation in representative area predominantly short, with stiff, trailing stems (i.e., *Batis maritima, Salicornia virginica*).
	0.050	0.060	0.070	Representative area predominantly tall, flexible-stemmed grasses (i.e., tall *Spartina alterniflora, S. cynosuroides, Scirpus sp.*).
	0.070	0.100	0.160	Vegetation in representative area predominantly tall, with stiff leaves (i.e., *Juncus roemerianus*) or mixed woody shrubs (i.e., mangroves).
Note: Adapted from Arcement and Schneider (1989) and Gardiner and Dakombe (1983).				

Figure B4. Relationship between surface roughness (*n*) and functional capacity



Sample variables 9 and 10 based on a representative number of locations in the WAA using a series of 1-m² plots arranged along one or more 30-m (100-ft) transects oriented perpendicular to the wetland shoreline or the hydrologic gradient.

9. V_{COVER} Mean Total Percent Vegetative Cover ________%

(1) Select one or more representative areas within the site for sampling. Beginning at the shoreward edge of the marsh, establish one or more 30-m transects perpendicular to the shoreline or along the hydrologic gradient (e.g. increasing elevation). If there are multiple vegetation community types within the WAA, the transect should intersect each vegetation community, in order to ensure a representative sample.

(2) Using a standard 1-m^2 frame, estimate total percent cover by **both live and dead** emergent macrophytic plant species at intervals along the transect, excluding any areas where water depths are too deep to support the growth of emergent vegetation. The number of transects and plots needed will depend on the size and heterogeneity of the site; **a minimum of 10 plots should be used**.

(3) Calculate the average of all total percent cover estimates.

(4) Assign variable subindex for V_{COVER} based on the chart in Figure B5. *Subindex* ________

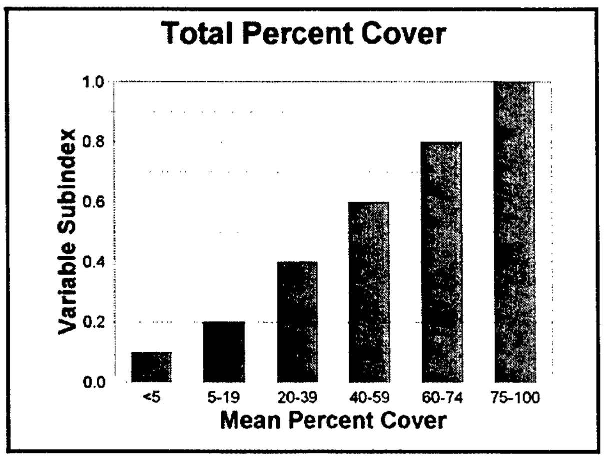


Figure B5. Relationship between mean total percent cover and functional capacity

10. V_{VEGSTR} Mean Vegetative Structure Index

(4) If the 1-m^2 sample plot above contains more than one species (i.e., *Spartina* and *Distichlis*), estimate the proportion of the 1-m^2 plot area covered by each species, omitting any species that occupies <10% cover. If the total percent cover above was estimated at 80%, the sum of the percent cover of each individual species should be 80%. There may be cases where there are several species that individually account for <10% cover, but collectively amount to 10%. In these cases, estimate the cumulative percent cover for the species group.

(5) For each species identified above, estimate the height in centimeters (rounded to the nearest 5 cm) at which the bulk of the biomass occurs (i.e., the most frequently occurring height) and record this value. For those species with trailing stems, the height should be measured in situ rather than extended vertically. Record an estimated height for the species group, if necessary.

(6) Calculate a vegetative structure index for **each** plot using the equation below.

V_{VEGSTR} = ((Hgt$_1$ × Proportion$_1$) + (Hgt$_2$ × Proportion$_2$) + ... (Hgt$_x$ × Proportion$_x$))

where: x = # plant species per plot.

[(_____ × _____) + (_____ × _____) + (_____ × _____)] = _______

(7) Compute the sum of all the vegetative structure indices generated above. ________

(8) Divide by the total number of plots to determine the mean. ________

(9) Assign a subindex value based on the chart in Figure B6. *Subindex* ________

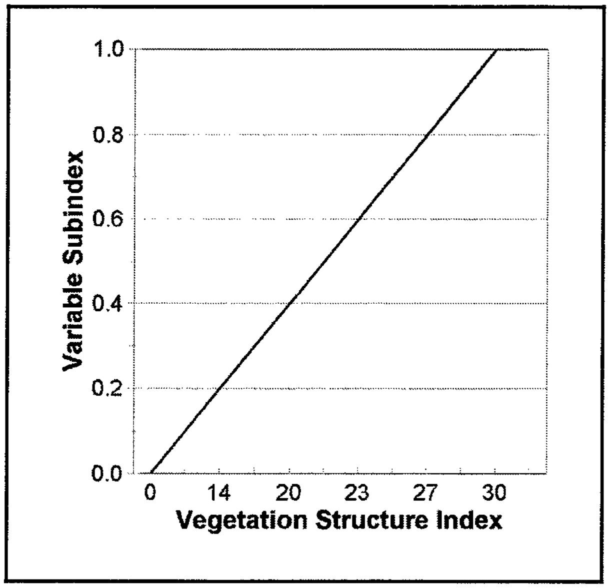


Figure B6. Relationship between vegetation structure index and functional capacity

Variables 11-14 are assessed only for the Shoreline Stabilization Function.

First, conduct a field site inspection and determine if there are any visual indicators of shoreline erosion within the project area. Examples of this include slumping banks, undercut banks, exposed root mats, or vertical bluffs along the shoreline (See Figure D1 for examples). If any of these features exist, assign a variable subindex to each of the following variables using the procedures outlined below and calculate a functional capacity index for this function. Otherwise, assign a default functional capacity index of 1.0 to this function, indicating the presence of a stable, non-eroding shoreline.

11. V_{WIDTH} Mean width of the marsh

(1) Using a recent aerial photo or direct field survey, establish a baseline along the lengthwise axis that runs roughly parallel to the shoreline and/or perpendicular to the topographic gradient.

(2) Draw a series of transects perpendicular to this baseline from the shoreline to the nearest upland and measure or estimate the average width of the marsh in meters (Figure 4). The number of transects is determined by the length of the baseline (Table B8).

Table B8
Number of Transects for Estimating Mean Marsh Width

Baseline Length (m)	Number of Transects
<300	3
300-1,500	5
1,500-3,000	7
>3,000	9

1____ m 2____ m 3____ m 4____ m 5____ m

(3) Determine the average of the widths recorded above.

(4) Assign a variable subindex based on Figure B7. *Subindex* __________

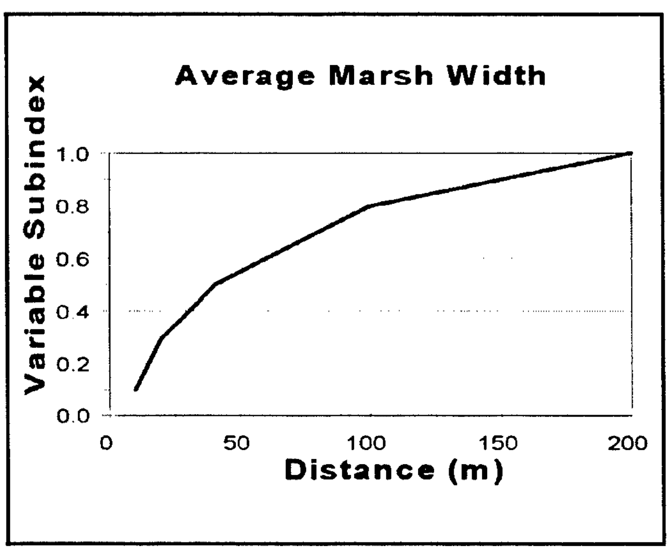


Figure B7. Relationship between average marsh width and functional capacity

12. V_{EXPOSE} Relative Exposure Index (REI)

(1) Measure fetch distances in kilometers for each of the 16 possible compass bearings.

(2) Using the map in Figure 3 in the main text (page 22), select the wind data station closest to your site.

(3) Using the supplemental information in Table E1 on mean annual wind speeds, calculate an REI using the equation below.

$$\text{Relative Exposure Index} = \sum_{i=1}^{16} V_i \times F_i \times P_i$$

where:

V_i = mean annual wind speed (km/hr)
F_i = fetch distance (km)
P_i = proportion of time wind blew from each of 16 cardinal and subcardinal compass directions

(4) Assign variable subindex based on Figure B8. *Subindex* __________

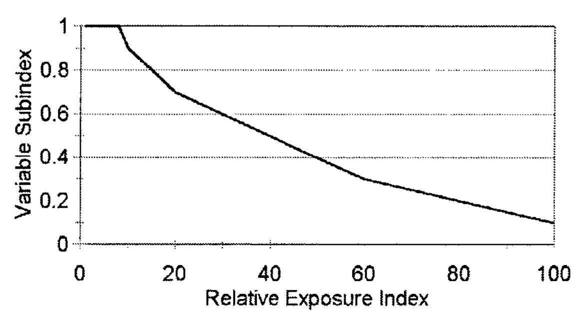


Figure B8. Relationship between relative exposure index and functional capacity

13.. V_{SLOPE} Distance to navigation channel OR water depths ≥2 m *Subindex* __________

Table B9
Relationship Between Shoreline Slope and Functional Capacity

Distance	Subindex
<50 m	0.1
50-150 m	0.5
>150 m	1.0

14. V_{SOIL} Soil texture *Subindex* ________

Table B10
Relationship Between Soil Type and Functional Capacity

Predominant Soil Type	Subindex
Clay	1.0
Clay loam	0.8
Loam	0.6
Sandy loam	0.4
Sandy	0.2

TIDAL FRINGE MARSH HGM FUNCTIONAL ASSESSMENT

Sediment Deposition $= (V_{ROUGH} \times V_{HYDRO})^{1/2}$

$(\_\_\_\_\_\_\_\_ \times \_\_\_\_\_\_\_\_)^{1/2} = \_\_\_\_\_\_\_\_$

Resident Nekton Utilization $= (V_{EDGE} + 2\ V_{HYDRO} + 0.5\ V_{NHC}) / 3.5$

$(\_\_\_\_\_\_\_\_ + 2\ \_\_\_\_\_\_\_\_ + \_\_\_\_\_\_\_\_ /2) / 3.5 = \_\_\_\_\_\_\_\_$

Nonresident Nekton Utilization $= [(V_{EDGE} + 2\ V_{HYDRO} + 0.5\ V_{NHC}) / 3.5)\ V_{OMA}]^{1/2}$

$[(\_\_\_\_\_\_\_\_ + 2\ \_\_\_\_\_\_\_\_ + 0.5\ \_\_\_\_\_\_\_\_ / 3.5) \times \_\_\_\_\_\_\_\_]^{1/2} = \_\_\_\_\_\_\_\_$

Invertebrate Prey Pool $= (V_{HYDRO} + V_{EDGE} + V_{COVER}) / 3$

$(\_\_\_\_\_\_\_\_ + \_\_\_\_\_\_\_\_ + \_\_\_\_\_\_\_\_) / 3 = \_\_\_\_\_\_\_\_$

Nutrient and Organic Carbon Exchange $= (V_{VEGSTR} \times V_{HYDRO})^{1/2}$

$(\_\_\_\_\_\_\_\_ \times \_\_\_\_\_\_\_\_)^{1/2} = \_\_\_\_\_\_\_\_$

Maintain Characteristic Plant Community Composition = V_{COVER} or $V_{TYPICAL}$, whichever is lower

Min (________ or ________) = ________

Plant Biomass Production = V_{VEGSTR} = ________________

Provide Wildlife Habitat $= [2\ V_{SIZE} + V_{WHC} + (\text{Min}\ (V_{COVER}\ \text{OR}\ V_{TYPICAL}))] / 4$

$(2\ \_\_\_\_\_\_\_\_ + \_\_\_\_\_\_\_\_ + \_\_\_\_\_\_\_\_) / 4 = \_\_\_\_\_\_\_\_$

Shoreline Stabilization $= (V_{SLOPE} + V_{WIDTH} + V_{EXPOSE} + V_{ROUGH} + V_{SOIL}) / 5$

$(\_\_\_\_\_\_\_\_ + \_\_\_\_\_\_\_\_ + \_\_\_\_\_\_\_\_ + \_\_\_\_\_\_\_\_ + \_\_\_\_\_\_\_\_) / 5 = \_\_\_\_\_\_\_\_$

Appendix C
Summaries of Functions and Variables

Definitions: Functions and Variables for Northwest Gulf of Mexico Tidal Fringe Marshes

Function 1: Shoreline stabilization

a. Definition. Emergent macrophytic vegetation of tidal fringe wetlands baffles wave energy, slowing water movement and allowing suspended material to be deposited on the marsh substrate. The roots help to bind the deposited sediments to the marsh substrate. Through these processes, tidal fringe wetlands maintain existing shorelines against erosion due to eustatic sea level rise and subsidence. A quantitative unit of measure of this function would be net hectares of marsh gained or lost/year/mile of coastline.

b. Model variables-symbols-measures-units.

1. Shoreline slope - V_{SLOPE} - distance to water depths ≥2 meters - meters.
2. Mean marsh width - V_{WIDTH} - average marsh width - meters.
3. Wave exposure - V_{EXPOSE} - relative exposure index - unitless.
4. Surface roughness - V_{ROUGH} - Manning's roughness coefficient (n) - unitless.
5. Soil texture - V_{SOIL} - predominant soil texture - unitless.

c. Assessment model:

$$FCI = (V_{SLOPE} + V_{WIDTH} + V_{EXPOSE} + V_{ROUGH} + V_{SOIL}) / 5$$

Function 2: Sediment deposition

a. *Definition.* This function refers to the potential deposition and retention of inorganic and organic particulates from the water column, primarily through physical processes. A quantitative measure of this function would be centimeters sediment/m^2/year.

b. *Model variables-symbols-measures-units.*

1. Surface roughness - V_{ROUGH} - Manning's roughness coefficient (*n*) - unitless.
2. Hydroperiod - V_{HYDRO} - site hydroperiod or degree of hydrological modification - unitless.

c. *Assessment model:*

$$FCI = (V_{ROUGH} \times V_{HYDRO})^{1/2}$$

Function 3: Nutrient and organic carbon exchange

a. *Definition.* The ability of a tidal wetland to export or import nutrients and organic carbon via tidal flushing, deposition, and erosion. Even though net fluxes on an annual basis may be small, changes in form or timing may be highly consequential (in - inorganic, out - organic; uptake - spring, release - fall). A quantitative measure of this function is mass of nutrient or dissolved and particulate carbon transformed per unit area per unit time (g/m^2/yr).

b. *Model variables-symbols-measures-units.*

1. Hydroperiod - V_{HYDRO} - site hydroperiod or degree of hydrological modification - unitless.
2. Vegetative structure index - V_{VEGSTR} - A weighted index of plant species percent cover and height - unitless.

c. *Assessment model:*

$$FCI = (V_{HYDRO} \times V_{VEGSTR})^{1/2}$$

Function 4: Resident nekton utilization

a. *Definition.* This function describes the potential utilization of a marsh by resident (non-migratory) fish and macrocrustacean species. A quantitative measure of this function would be abundance (or biomass) of resident nekton per square meter.

b. *Model variables-symbols-measures-units.*

1. Edge - V_{EDGE} - the amount of marsh-water interface - meters/hectare.

2. Hydroperiod - V_{HYDRO} - site hydroperiod or degree of hydrological modification - unitless.
3. Nekton habitat complexity - V_{NHC} - number of nekton habitat types present - integer.

c. *Assessment model:*

$$FCI = (V_{EDGE} + 2\ V_{HYDRO} + 0.5\ V_{NHC}) / 3.5$$

Function 5: Nonresident nekton utilization

a. *Definition.* This function describes the potential utilization of a site by seasonally occurring adults or juveniles of marine or estuarine-dependent fisheries species. A quantitative measure of this function would be abundance (or biomass) of nonresident nekton per square meter.

b. *Model variables-symbols-measures-units.*

1. Edge - V_{EDGE} - the amount of marsh-water interface - meters/hectare.
2. Hydroperiod - V_{HYDRO} - site hydroperiod or degree of hydrological modification - unitless.
3. Nekton habitat complexity - V_{NHC} - number of nekton habitat types present - integer.
4. Opportunity for marsh access - V_{OMA} - percentage of the total edge that is tidally connected - percent.

c. *Assessment model:*

$$FCI = [((V_{EDGE} + 2\ V_{HYDRO} + 0.5\ V_{NHC}) / 3.5)\ V_{OMA}]^{1/2}$$

Function 6: Maintain invertebrate prey pool

a. *Definition.* This function estimates the potential for the wetland to produce and maintain a characteristic benthic and epiphytic invertebrate prey pool. A quantitative measure of this function would be abundance of invertebrate species per unit area.

b. *Model variables-symbols-measures-units.*

1. Hydroperiod - V_{HYDRO} - site hydroperiod or degree of hydrological modification - unitless.
2. Edge - V_{EDGE} - the amount of marsh-water interface - meters/hectare.
3. Mean percent vegetative cover - V_{COVER} - average percent cover by emergent macrophytic vegetation - percent.

c. *Assessment model:*

$$FCI = (V_{HYDRO} + V_{EDGE} + V_{COVER}) / 3$$

Function 7: Provide wildlife habitat

a. *Definition.* Describes the potential utilization of the marsh by resident and migratory avifauna, herpetofauna, and mammals. A quantitative measure of this function would be abundance of birds, herps, and mammals per unit area (hectare).

b. *Model variables-symbols-measures-units.*

1. Total effective patch size - V_{SIZE} - the sum of the sizes (in hectares) of all wetlands in the vicinity of the WAA weighted by their degree of corridor connectivity-hectares.
2. Wildlife habitat complexity - V_{WHC} - total number of different wildlife habitat types present - integer.
3. Mean percent vegetative cover - V_{COVER} - average percent cover by emergent macrophytic vegetation-percent.
4. Percent cover by typical vegetation - $V_{TYPICAL}$ - proportion of the site that is covered by vegetation typical of the regional subclass - percent.

c. *Assessment model:*

$$FCI = [2\ V_{SIZE} + V_{WHC} + (Min\ (V_{COVER}\ \text{OR}\ V_{TYPICAL}))] / 4$$

Function 8: Maintain characteristic plant community composition

a. *Definition.* The ability of a wetland to support a native plant community of characteristic species composition. The higher the contribution of exotic or nuisance species or even species characteristic of a different marsh subclass, the lower the functional capacity index will be. A second possible indicator of deviation from characteristic plant community composition is total percent vegetative cover.

b. *Model variables-symbols-measures-units.*

1. Mean percent vegetative cover - V_{COVER} - average percent cover by emergent macrophytic vegetation - percent.
2. Percent cover by typical vegetation - $V_{TYPICAL}$ - proportion of the site that is covered by vegetation typical of the regional subclass - percent.

c. *Assessment model:*

$$FCI = \text{Minimum}\ (V_{COVER}\ \text{OR}\ V_{TYPICAL})$$

Function 9: Plant biomass production

a. Definition. A measure of standing crop used as an indicator of the potential for a site to produce plant material. A quantitative measure of this function would be plant biomass per square meter, assumed to be directly related to plant biomass produced per square meter per year.

b. Model variables-symbols-measures-units.

   1. Vegetative structure index - V_{VEGSTR} - A weighted index of plant species percent cover and height - unitless.

c. Assessment model:

   $FCI = V_{VEGSTR}$

Summary of Model Variables: Measure/Units, Methods and Scaling

1. Shoreline slope (V_{SLOPE})

Measure/units: Distance in meters from the marsh shoreline to water depths of at least 2 m.

Method: (1) Estimate the distance from the seaward edge of the project perimeter needed to reach water depths of at least 2 m using field reconnaissance, maps, or bathymetry charts.

(2) Report the distance in meters.

Variable scaling:

Table C1
Relationship Between Shoreline Slope and Functional Capacity

Distance	Subindex
<50 m	0.1
50-150 m	0.5
>150 m	1.0

2. Mean marsh width (V_{WIDTH})

Measure/Units: Average width of the marsh in meters.

Method: (1) Using a recent aerial photo or direct field survey, establish a baseline by identifying the lengthwise axis closest to the upland boundary or marsh perimeter that runs roughly parallel to the shoreline and/or perpendicular to the topographic gradient.

(2) Draw a series of transects from this baseline perpendicular to the shoreline and measure the distances to estimate the average width of the marsh. The number of transects is determined by the length of the baseline.

Variable Scaling:

Table C2
Number of Transects for Estimating Mean Marsh Width

Baseline Length (m)	Number of Transects
<300	3
300-1,500	5
1,500-3,000	7
>3,000	9

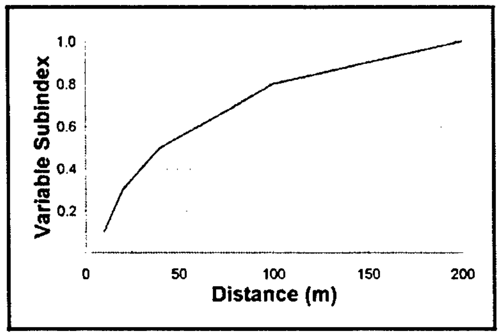


Figure C1. Relationship between mean marsh width and functional capacity

3. Wave exposure (V_{EXPOSE})

Measure/Units: An index of the relative wave exposure of a site - unitless.

Method: (1) Using aerial photographs, maps, or other charts, measure and record fetch distances for each of the 16 possible compass bearings. Fetch distances in some directions will be zero.

(2) Select the wind data station closest to the site (Figure 3).

(3) Using annual wind summary data from the appropriate wind station (Table E1), calculate an exposure index according to the formula:

$$\text{Relative Exposure Index} = \sum_{i=1}^{16} V_i \times F_i \times P_i$$

where:

V_i = mean annual wind speed (km/hr)
F_i = fetch distance (km)
P_i = proportion of time wind blew from each of 16 cardinal and subcardinal compass directions

Variable Scaling:

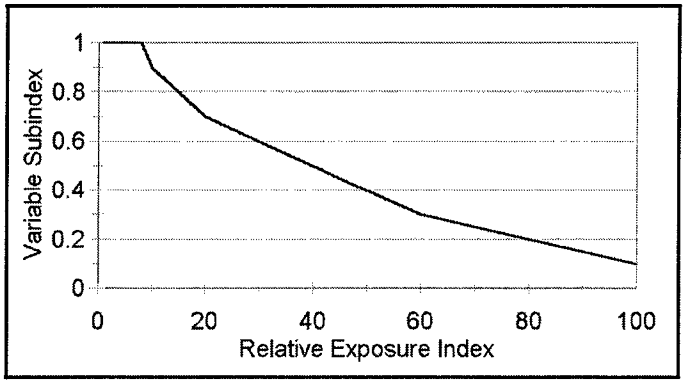


Figure C2. Relationship between relative exposure index and functional capacity

4. Surface roughness (V_{ROUGH})

Measure/Units: Manning's roughness coefficient (*n*) - unitless.

Method: (1) Use the following method modified from Arcement and Schneider (1989) and Gardiner and Dackombe (1983) to estimate Manning's *n* roughness based on a characterization of the various components that contribute to marsh surface roughness. These include: n_{BASE} = a base value of *n* for the marsh's bare soil surface, n_{TOPO} = a correction factor for the effect of topographic relief on the marsh surface, and n_{VEG} = a value for vegetation on the marsh surface.

(2) Using the descriptions in Table C3, assign an adjustment value for n_{BASE}, n_{TOPO}, and n_{VEG}.

(3) Sum the values of the roughness components.

(4) Report the value of Manning's *n* as a unitless number.

Variable Scaling:

Table C3
Adjustment Values for Roughness Components Contributing to Manning's Roughness Coefficient (*n*)

Roughness Component	Adjustment to *n* Value			Description of Terms
Sediment surface (n_{BASE})	0.025			Base value for bare marsh soil.
	0.03			More than 25% of sediment surface covered with gravel or broken shell.
Topographic relief (n_{TOPO})	0.001			Representative area is flat with essentially no microtopographic (i.e., hummocks) or macrotopographic relief (i.e., berms, tidal channels, ridges and swales, ponds).
	0.005			Microtopographic (i.e., hummocks) or macrotopographic relief (i.e., berms, tidal channels, ridges and swales, ponds) cover 5-25% of a representative area.
	0.010			Microtopographic (i.e., hummocks) or macrotopographic relief (i.e., berms, tidal channels, ridges and swales, ponds) cover 26-50% of a representative area.
	0.020			Microtopographic (i.e., hummocks) or macrotopographic relief (i.e., berms, tidal channels, ridges and swales, ponds) cover 50% of a representative area.
Roughness Component	**Percent Cover**			**Description of Conditions**
	<50	**50-75**	**76-100**	
Vegetation (n_{VEG})	0.025	0.030	0.035	Representative area predominantly short, flexible-stemmed grasses (i.e., short *Spartina alterniflora, S. patens, Distichlis spicata*).
	0.035	0.040	0.050	Vegetation in representative area predominantly short, with stiff, trailing stems (i.e., *Batis maritima, Salicornia virginica*).
	0.050	0.060	0.070	Representative area predominantly tall, flexible-stemmed grasses (i.e., tall *Spartina alterniflora, S. cynosuroides, Scirpus sp.*).
	0.070	0.100	0.160	Vegetation in representative area predominantly tall, with stiff leaves (i.e., *Juncus roemerianus*) or mixed woody shrubs (i.e., mangroves).

Note: Adapted from Arcement and Schneider (1989) and Gardiner and Dackombe (1983).

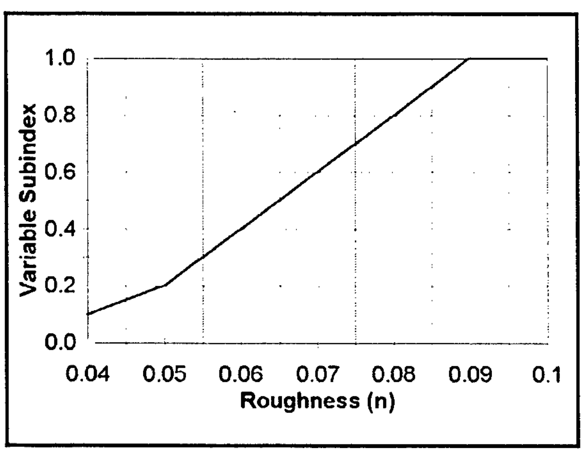


Figure C3. Relationship between surface roughness and functional capacity

5. Soil texture (V_{SOIL})

Measure/Units: The predominant soil texture of the WAA-unitless.

Method: (1) Using county soil survey maps or direct field sampling, determine the predominant soil texture of the WAA (Figure E1).

(2) Report the predominant soil texture as sandy, sandy loam, loam, clay loam, or clay.

Variable Scaling:

Table C4
Relationship Between Soil Texture and Functional Capacity

Predominant Soil Type	Subindex	Predominant Soil Type	Subindex
Clay	1.0	Sandy loam	0.4
Clay loam	0.8	Sandy	0.2
Loam	0.6	Impervious substrate	0.0

6. Hydrologic Regime (V_{HYDRO})

Measure/Units: A determination of the tidal flooding regime of the WAA - unitless.

Method: Estimate the degree of hydrological restriction/modification within the project area using the descriptions provided in the table below.

Variable Scaling:

Table C5
Relationship Between Hydrologic Regime and Functional Capacity

Condition	Site Description	Subindex
Unrestricted	Site is open to free exchange of tidal waters. No obvious hydrologic alteration or restrictions present.	1.0
Moderately restricted	Moderate hydrologic restriction present (i.e., presence of low-elevation berm, which is frequently overtopped by high tide events or has multiple breaches or large culverts).	0.6
Severely restricted	Severe hydrologic restriction present (i.e., presence of high-elevation berm, which is infrequently overtopped by high-tide events or has a single opening, breach, or small culvert).	0.3
Very severely restricted	Site receives tidal floodwaters only during extreme storm tide events.	0.1
No tidal exchange	Site is isolated from tidal exchange. The principal source of flooding is water sources other than tidal action (i.e., precipitation or groundwater). *Note: If this condition exists, another wetland assessment model should be strongly considered unless the site was formerly a tidal wetland prior to hydrologic modification.*	0.0

7. Marsh-water Interface (V_{EDGE})

Measure/Units: An estimate of the degree of marsh dissectedness, or the amount of marsh-water interface - m/ha.

Method: Using aerial photography, or digital imagery at a scale of 1 cm = 48 m (1 in. = 400 ft) and GIS mapping software, determine the amount of marsh-water interface using one of the following methods (in order of accuracy and preference).

Quantitative Approach.

(1) Measure the linear extent of all visible marsh-water edge in meters, including both banks of tidal creeks and channels.
(2) Determine the total size of the wetland tract in hectares.
(3) Express the result as a ratio of total edge per hectare.

Qualitative Approach.

(1) Using the site descriptions provided below and the pictorial key in Appendix D (Figures D2-D9), categorize the degree of marsh-water interface.

Variable Scaling:

Table C6
Relationship Between Edge:Area and Functional Capacity

Site Description	Qualitative	Quantitative (m/ha)	Subindex
1) Marsh shows signs of deterioration due to subsidence (i.e. highly fragmented with large amounts of open water. Vegetation occurs mainly in isolated hummocks or on natural levees along tidal creeks). Although edge:area is very high, this condition is not considered sustainable in the long term (Figure D2).	Very High	>800	0.8
1) Well-developed tidal drainage network present (Figure D3), OR **2)** Simple tidal drainage network (may consist of only a single channel) present with isolated ponds and depressions present in the marsh interior (Figures D4 and D5). **3)** Atypical geomorphic configuration with a large amount of shoreline relative to total area (i.e. small island or narrow peninsula) (Figure D7).	High	350-800	1.0
1) Simple tidal drainage network (may consist of only a single channel). Isolated ponds and depressions are few or lacking **2)** Narrow fringe marsh that lacks tidal creeks. One lengthwise shoreline is exposed to tidal waters. Area of marsh is small relative to shoreline length (Figures D6 and D8).	Moderate	200-350	0.7
Marsh lacks both tidal creeks and isolated ponds and depressions. Shoreline is generally linear or smooth curvilinear without embayments or convolutions. Area of marsh is large relative to shoreline length (Figure D9).	Low	<200	0.4

8. Opportunity for marsh access (V_{OMA})

Measure/Units: The proportion of the total marsh-water interface or "edge" that is tidally connected - ratio.

Method: Using aerial photography, or digital imagery at a scale of 1 cm = 48 m (1 in. = 400 ft), determine the amount of marsh-water interface using one of the following methods (in order of accuracy and preference).

Alternative 1.

(1) Using GIS mapping software, measure the linear extent of all visible marsh-water edge in meters.
(2) Determine the length of this edge (in meters) that is tidally connected.
(3) Express the result as a ratio of tidally connected edge to total edge.

Alternative 2.

(1) Using an English area grid overlay, count the number of points that intercept the tidally connected edge.
(2) Count the number of points that intercept all edge (marsh-water interface).
(3) Express the result as a ratio of tidally connected edge to total edge.

Variable Scaling:

Table C7
Relationship Between Opportunity for Marsh Access and Functional Capacity

Tidally Connected Edge: Total Edge	Subindex
50-100%	1.0
35-50%	0.7
25-35%	0.5
1-25%	0.2
No tidally connected edge present	0.0

9. Nekton habitat complexity (V_{NHC})

Measure/Units: A measure of the total number of different nekton habitat types present within or near the WAA - integer.

Method: (1) During field reconnaissance, record the number of potential habitat types identified in the list below that are present within a 30-m (100-ft) radius of project area perimeter - integer.

Coarse woody debris	Unvegetated flats	Algal mats
Subtidal creeks/channels	Oyster reef	Mangroves
Intertidal creeks/channels	Low marsh	High marsh
Ponds or depressions	Submerged aquatic vegetation	

Variable Scaling:

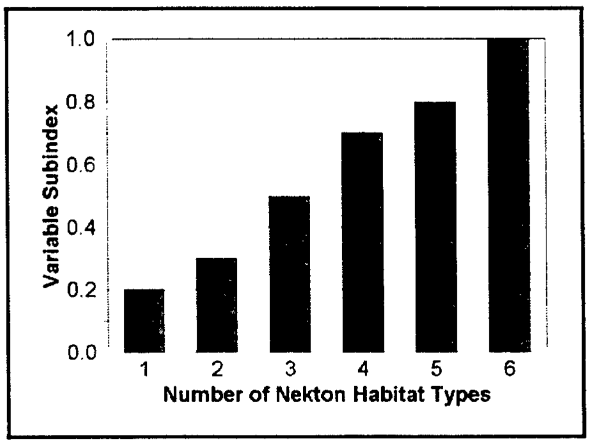


Figure C4. Relationship between nekton habitat complexity and functional capacity

10. Wildlife habitat complexity (V_{WHC})

Measure/Units: A measure of the total number of different wildlife habitat types present within or near the WAA - integer.

Method: (1) Count the number of habitat types in the list below that are present within a 2-km radius of the project area perimeter.

Supratidal habitats (hummocks, logs)	Forested uplands
Unvegetated beach	Grasslands
Scrub-shrub	

(2) Add to this number the total number of nekton habitat types observed (V_{NHC}).

(3) Report the sum as the total number of wildlife habitat types present.

Variable Scaling:

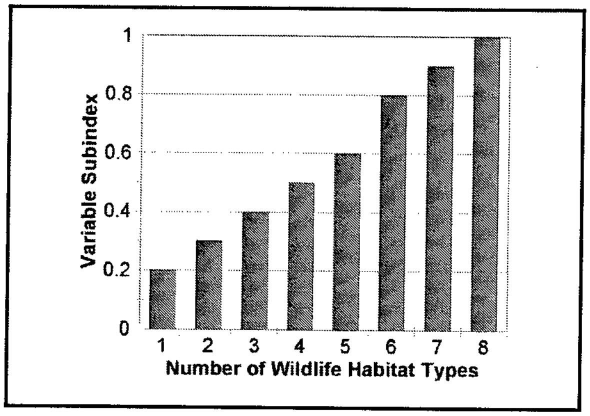


Figure C5. Relationship between wildlife habitat complexity and functional capacity

11. Mean vegetative percent cover (V_{COVER})

Measure/Units: The average percent cover by emergent macrophytic vegetation as determined in a series of 1-m^2 plots-percent.

Method: (1) Select one or more representative areas within the site for sampling. Beginning at the shoreward edge of the marsh, establish one or more 30-m transects perpendicular to the shoreline or along the hydrologic gradient (e.g., increasing elevation). If there are multiple vegetation community types within the WAA, the transect should intersect each vegetation community, in order to ensure a representative sample.

(2) Using a standard 1-m^2 frame, estimate total percent cover by **both live and dead** emergent macrophytic plant species at intervals along the transect, excluding any areas where water depths are too deep to support the growth of emergent vegetation. The number of transects and plots needed will depend on the size and heterogeneity of the site; **a minimum of 10 plots should be used**.

(3) Calculate the average of all total percent cover estimates.

Variable Scaling:

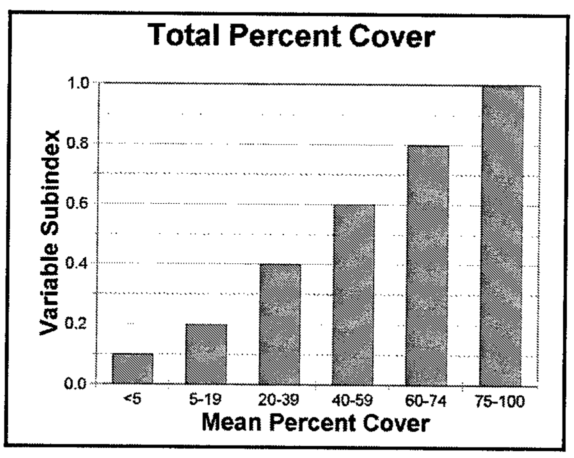


Figure C6. Relationship between mean total percent cover and functional capacity

12. Vegetative structure index (V_{VEGSTR})

Measure/Units: A weighted index of vegetative structural complexity-unitless.

Method: (1) Using the same 1-m^2 sample plots as above, estimate the proportion of the 1-m^2 plot area covered by each species, omitting any species that occupies <10% cover. If the total percent cover above was estimated at 80%, the sum of the percent cover of each individual species should be 80%. There may be cases where there are several species that individually account for <10% cover, but collectively amount to >10%. In these cases, estimate the cumulative percent cover for the species group.

(2) For each species identified, estimate the height in centimeters (rounded to the nearest 5 cm) at which the bulk of the biomass occurs (i.e., the most frequently occurring height) and record this value. For those species with trailing stems, the height should be measured in situ rather than extended vertically. Record an estimated height for the species group, if necessary.

(3) Calculate a vegetative structure index for each plot using the equation below.

$$V_{VEGSTR} = ((Hgt_1 \times Proportion_1) + (Hgt_2 \times Proportion_2) + \ldots\ldots (Hgt_x \times Proportion_x))$$

where x = # plant species per plot.

(4) Compute the sum of all the vegetative structure indices generated above.

(5) Divide by the total number of plots to determine the mean.

Variable Scaling:

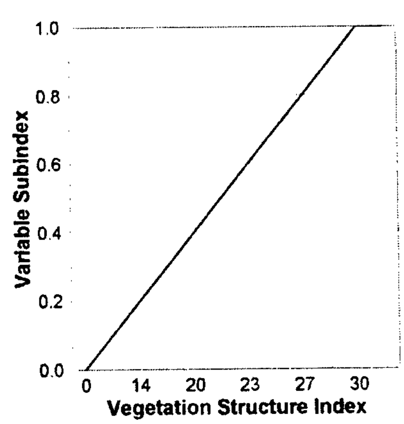


Figure C7. Relationship between vegetation structure index and functional capacity

13. Percent cover by typical vegetation species ($V_{TYPICAL}$)

Measure/Units: Percentage of the site that is covered with macrophytic vegetation typical of the regional subclass-percent.

Method: (1) If the WAA does not contain any exotic, nuisance, or otherwise undesirable plant species (see Table C8); assign a subindex value of 1.0.

(2) If the WAA contains any exotic, nuisance, or otherwise undesirable plant species, estimate the percentage of the WAA covered by these species using aerial photography and/or field reconnaissance. Table C8 lists some of these species and the salinity regime in which they occur.

(3) Subtract the percent cover estimate obtained above from 100 to estimate the percent cover of "typical" plant species.

Variable Scaling:

Table C8
Exotic, Nuisance, or Undesirable Plant Species

Scientific Name	Common Name	Salt/Brackish	Intermediate
Alternanthera philoxeroides	Alligator weed		L
Aster spinosus	Spiny aster		H
Baccharis halimifolia	Eastern false-willow	H	H
Phragmites australis	Common reed	H	L,H
Sesbania drummondii	Drummond's rattlebush		L,H
Typha spp.	Cattail	L	L

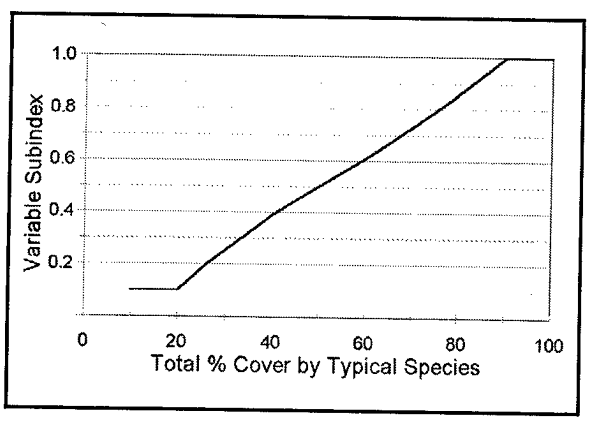


Figure C8. Relationship between percent cover by typical plant species and functional capacity

14. Total effective patch size (V_{SIZE})

Measure/Units: The sum of the sizes (in hectares) of all wetlands in the vicinity of the WAA weighted by their degree of corridor connectivity-hectares.

Methods: This variable is measured using the following procedure.

(1) If the core wetland size exceeds 200 ha, assign a variable subindex value of 1.0. The core wetland is defined as a contiguous patch of tidal fringe wetland that contains the WAA.

(2) If the core wetland size is <200 ha, identify other patches of wetlands in the surrounding area, and record the size of each patch in hectares. These wetlands may be in a wetland subclass other than tidal fringe. Then, using the descriptions provided in the table below, determine the degree of connectivity between each wetland patch and the core wetland.

(3) Multiply the size of each patch (hectares) identified in the previous step by the appropriate connectivity multiplier from the table below.

Table C9
Assessing the Degree of Corridor Connectivity

Corridor Type	Corridor Description	Multiplier
Contiguous corridor	**1)** Open water stretches <60 m (regardless of depth). **2)** Unvegetated stretches of shoreline or strips of other wetland subclasses <60 m in length that have an aquatic shelf at least 3 m wide and are <0.3 m deep at MSL. This discounts most tidal creeks and coves or unvegetated stretches of shoreline abutting uplands as barriers to wildlife that are traveling through their daily home range.	1.00
Partially impeded corridor	**1)** Open water stretches from 60-300 m (regardless of depth). **2)** Unvegetated shorelines or strips of other wetland subclasses from 60-500 m in length that have an aquatic shelf at least 3 m wide with water depths <0.3 m at MSL. Deeper stretches of water that interrupt the shelves are not considered impeding if they are <60 m wide. **3)** Stretches of undeveloped upland that are <30 m in width.	0.75
Impeded corridor	**1)** Shoreline shelves or wetland strips 500-1200 m long. **2)** Stretches of undeveloped upland 30-300 m in width.	0.50
Corridor absent or barrier present	**1)** Open water stretches or undeveloped upland >300 m in width. **2)** Shorelines >300 m long that contain no shelf with waters 0.3 m deep (i.e., long stretches of bulkheading). **3)** Roadways with >100 vehicle crossings per day that are unbridged or have a bridge opening <3 m wide. **4)** Highly developed urban, residential, or industrial areas.	0.0

(4) The sum of all the products above (in hectares) is the Total Effective Patch Size.

Variable Scaling:

Table C10
Relationship Between Total Effective Patch Size and Functional Capacity

Total Effective Patch Size	Subindex
>200 ha	1.00
5-200 ha	0.75
1-5 ha	0.50
0.2-1 ha	0.25
<0.2 ha	0.10

Table C11
Summary of Variables by Function

Variable	Function
1. Shoreline slope (V_{SLOPE})	Shoreline stabilization
2. Mean marsh width (V_{WIDTH})	Shoreline stabilization
3. Wave exposure (V_{EXPOSE})	Shoreline stabilization
4. Surface roughness (V_{ROUGH})	Shoreline stabilization Sediment deposition
5. Soil texture (V_{SOIL})	Shoreline stabilization
6. Hydroperiod (V_{HYDRO})	Sediment deposition Resident nekton utilization Nonresident nekton utilization Maintain invertebrate prey pool Nutrient and organic carbon exchange
7. Marsh-water interface (V_{EDGE})	Resident nekton utilization Nonresident nekton utilization Maintain invertebrate prey pool
8. Nekton habitat complexity (V_{NHC})	Resident nekton utilization Nonresident nekton utilization
9. Opportunity for marsh access (V_{OMA})	Nonresident nekton utilization
10. Mean vegetative percent cover (V_{COVER})	Maintain invertebrate prey pool Maintain characteristic plant community composition Provide wildlife habitat
11. Vegetative structure (V_{VEGSTR})	Nutrient and organic carbon exchange Plant biomass production
12. Percent cover by typical vegetation species ($V_{TYPICAL}$)	Maintain characteristic plant community composition Provide wildlife habitat
13. Total effective patch size (V_{SIZE})	Provide wildlife habitat
14. Wildlife habitat complexity (V_{WHC})	Provide wildlife habitat

Appendix D
Reference Data

Table D1
Locations of Reference Sites Sampled in the Mid-Coast Region

Site	Latitude	Longitude	Year Planted
Natural Reference Marshes			
Bayland Marina	29° 42.70	-94° 59.70	NA
Dickinson Bayou (mid)	29° 27.75	-94° 57.63	NA
Dickinson Bayou (lower)	29° 28.55	-94° 57.53	NA
Bolivar W	29° 27.89	-94° 41.26	NA
Bolivar E	29° 28.55	-94° 40.33	NA
East Chocolate Bay*	29° 11.14	-95° 06.48	NA
Matagorda Bay	28° 45.61	-95° 40.55	NA
Dickinson Bayou (upper)	29° 27.15	-94° 59.00	NA
Swan Lake	29° 20.59	-94° 53.79	NA
Wharton Bayou*	29° 10.24	-95° 10.00	NA
Created Marshes			
Alligator Point (C)	29° 10.69	-95° 06.75	1983
Armand Bayou (C)	29° 35.10	-95° 04.15	1995
Bayland Marina (C)	29° 42.30	-94° 59.40	1996
Bolivar E (C)	29° 25.17	-94° 43.95	1976
Bolivar W (C)	29° 25.44	-94° 44.21	1984-85
Kemah (C)	29° 33.33	-95° 01.45	1990
Matagorda Bay (C)	28° 45.45	-95° 40.43	1984
Swan Lake (C)	29° 21.20	-94° 53.73	1992
Webster Power Plant (C)	29° 31.58	-95° 05.78	1998

Note: (C) denotes created marshes planted on dredged material. * Reference Standard Sites.

Table D2
Exposure Indices for Each Site Presented in Rank Order

Site	Wind Station	Exposure Index
Bayland Marina	Morgans Point	<1
Wharton Bayou	Freeport	<1
Armand Bayou (C)	Eagle Point	2
Webster Power Plant (C)	Eagle Point	2
Matagorda Bay	Port O'Connor	2
Kemah (C)	Eagle Point	3
Dickinson Bayou (mid)	Eagle Point	3
Swan Lake	Galveston Pleasure Pier	4
Dickinson Bayou (lower)	Eagle Point	4
Dickinson Bayou (upper)	Eagle Point	4
Swan Lake (C)	Galveston Pleasure Pier	5
Bayland Marina (C)	Morgans Point	5
East Chocolate Bay	Galveston Pleasure Pier	6
Alligator Point (C)	Galveston Pleasure Pier	9
Matagorda Bay (C)	Port O'Connor	13
Bolivar W	Port Bolivar	63
Bolivar E	Port Bolivar	76
Bolivar E (C)	Port Bolivar	95
Bolivar W (C)	Port Bolivar	95

Table D3
Average Marsh Width for Sites in the Mid-Coast Region

Location	Average Width (m)
Dickinson Bayou (upper)	5
Bolivar W (C)	17
Dickinson Bayou (lower)	33
Dickinson Bayou (mid)	34
Bolivar E (C)	37
Dickinson Bayou (upper)	41
Swan Lake (C)	48
Swan Lake	131
Alligator Point (C)	226
Halls Bayou	283
East Chocolate Bay	416
Wharton Bayou	640
Elmgrove Point	835

Table D4
Mean Percent Cover and Vegetation Structure Indices for Mid-Coast Region Reference Sites

Site	Mean % Cover	Vegetation Structure Index
East Chocolate Bay	41	23.90
Alligator Point (C)	51	20.52
Armand Bayou (C)	38	32.66
Bayland Marina	61	34.41
Bayland Marina (C)	31	19.77
Bolivar W	72	37.03
Bolivar E	78	35.38
Bolivar E (C)	70	23.30
Bolivar W (C)	67	31.06
Kemah (C)	68	25.70
Dickinson Bayou (upper)	95	39.28
Dickinson Bayou (mid)	47	34.58
Dickinson Bayou (lower)	37	20.37
Swan Lake	22	8.26
Swan Lake (C)	39	27.36
Wharton Bayou	83	57.60

Table D5
Edge Characteristics for Selected Sites

Site	Total Edge (m/ha)	Tidally Connected: Total Edge
Natural Reference Marshes		
Matagorda Bay	339 M	40%
East Chocolate Bay	379 M	50%
Dickinson Bayou (mid)	380 M	100%
Bolivar W	444 H	25%
Bayland Marina	532 H	100%
Dickinson Bayou (lower)	585 H	100%
Swan Lake	812 VH	100%
Dickinson Bayou (upper)	844 VH	100%
Bolivar E	1134 VH	80%
Wharton Bayou	1200 VH	80%
Created Marshes		
Alligator Point (C)	123 L	1.00
Matagorda Bay (C)	189 L	1.00
Bayland Marina (C)	355 M	1.00
Bolivar E (C)	397 M	1.00
Bolivar W (C)	444 H	1.00
Kemah (C)	736 H	1.00
Armand Bayou (C)	453 H	1.00
Swan Lake (C)	467 H	0.98

Note: L = low (<200 m/ha), M = moderate (200-350 m/ha), H = high (350-800 m/ha), and VH = very high (>800 m/ha)

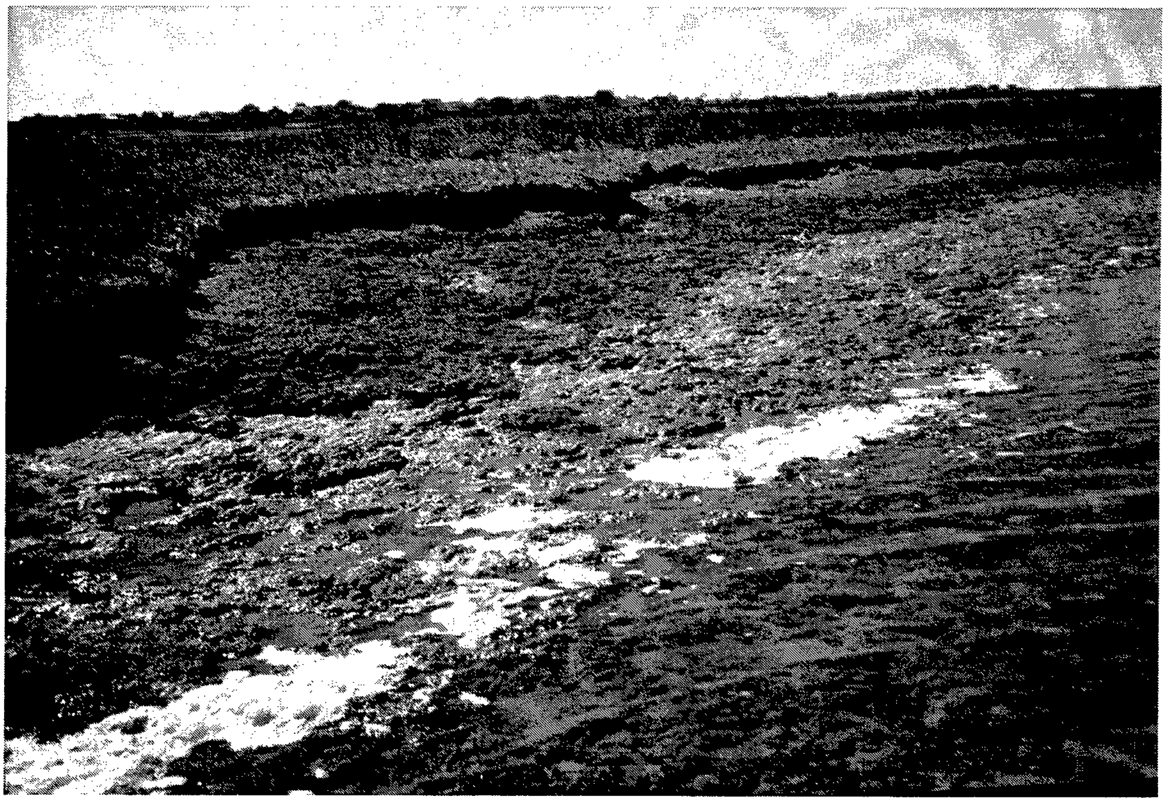


Figure D1. Marsh shoreline exhibiting evidence of erosion (e.g., vertical bank, exposed roots, slumping bank)

Pictoral key for estimating edge
1 in. = 400 ft

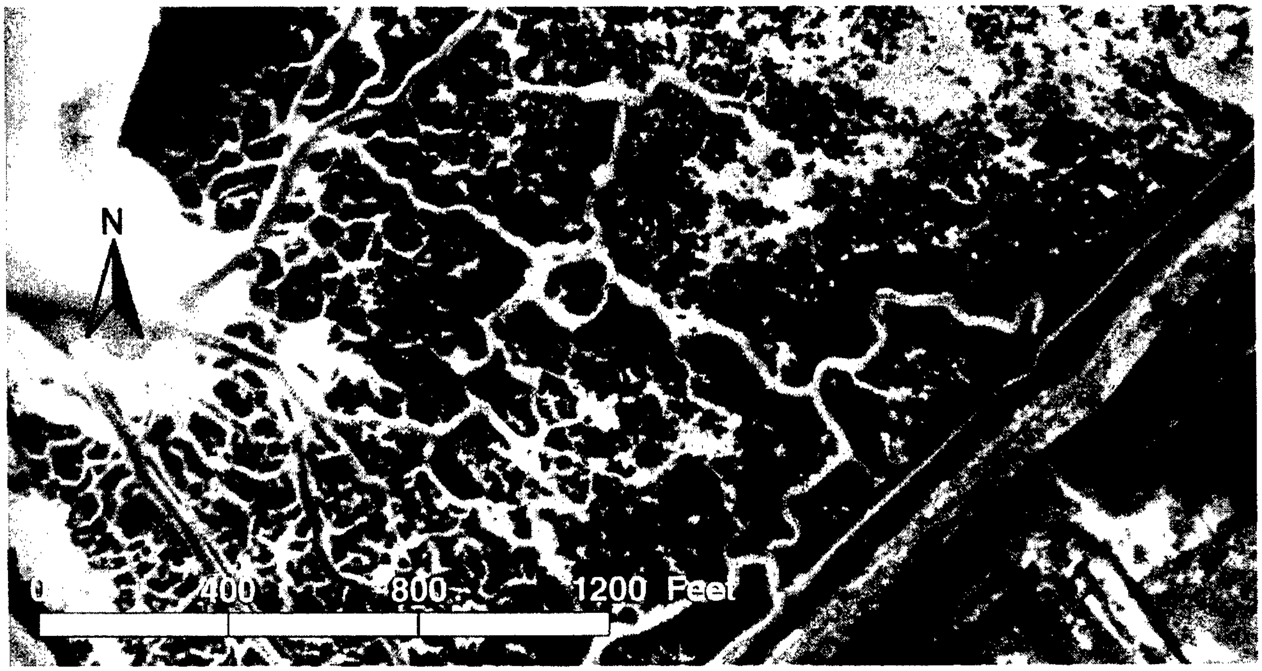


Figure D2. Example of natural marsh with very high edge:area ratio

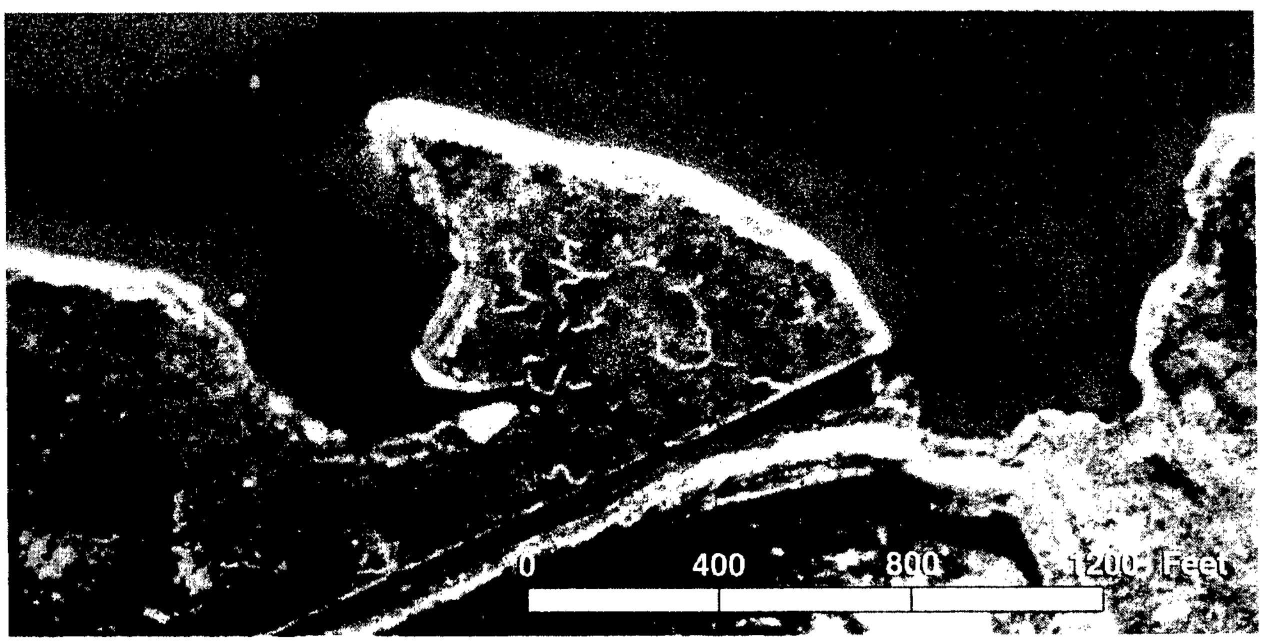


Figure D3. Example of natural marsh with well-developed tidal drainage network and convoluted shoreline (high edge:area ratio)

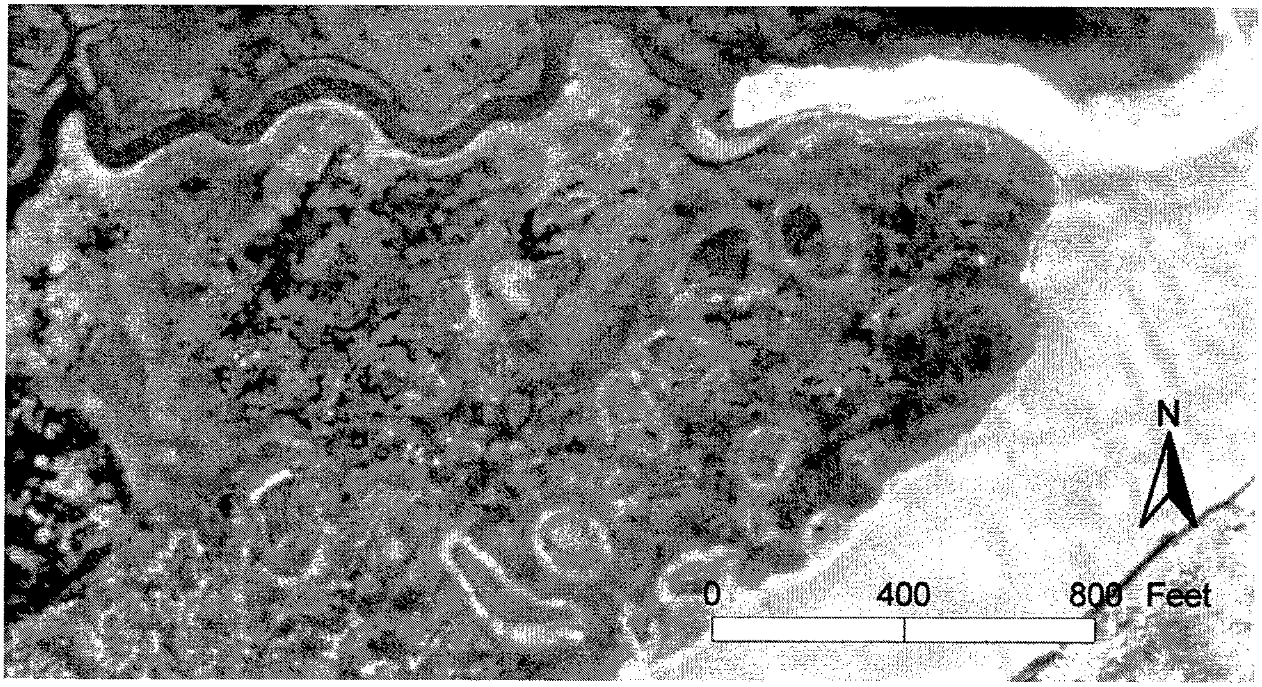


Figure D4. Example of natural marsh with simple tidal drainage network and isolated ponds and depressions (high edge:area ratio)

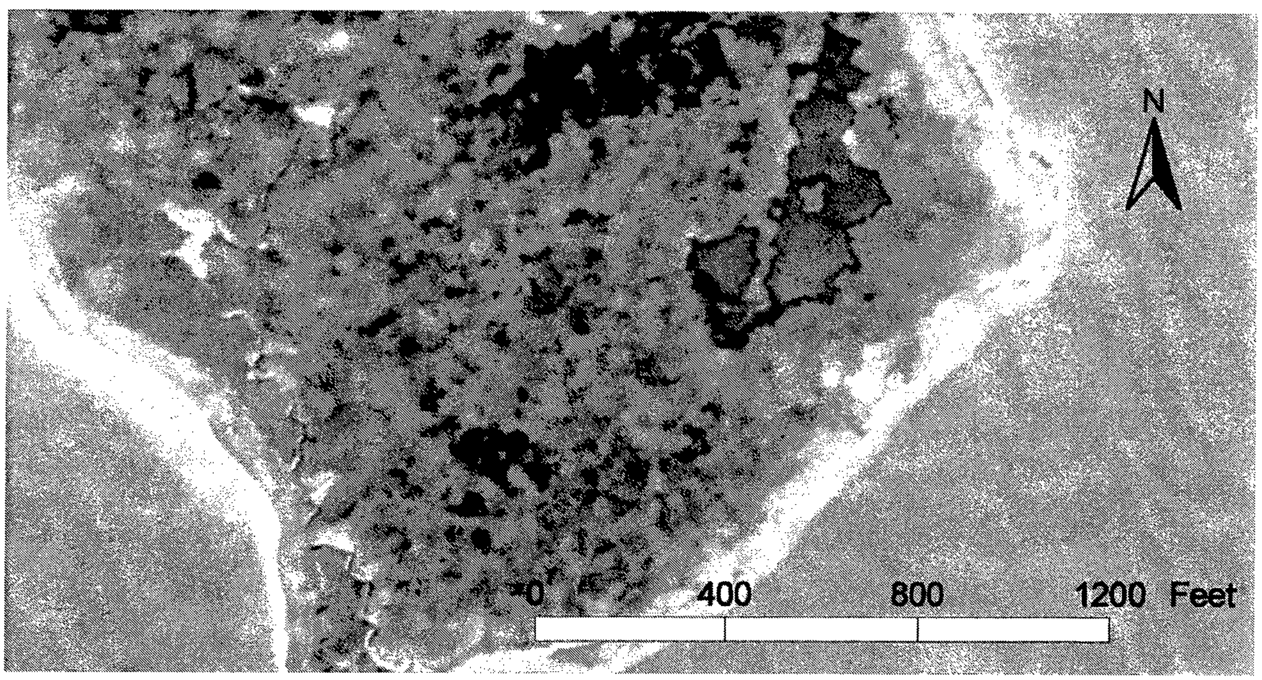


Figure D5. Example of natural marsh with simple tidal drainage network and isolated ponds and depressions (high edge:area ratio)

Figure D6. Example of small narrow fringe marsh that lacks tidal creeks (moderate edge:area ratio)

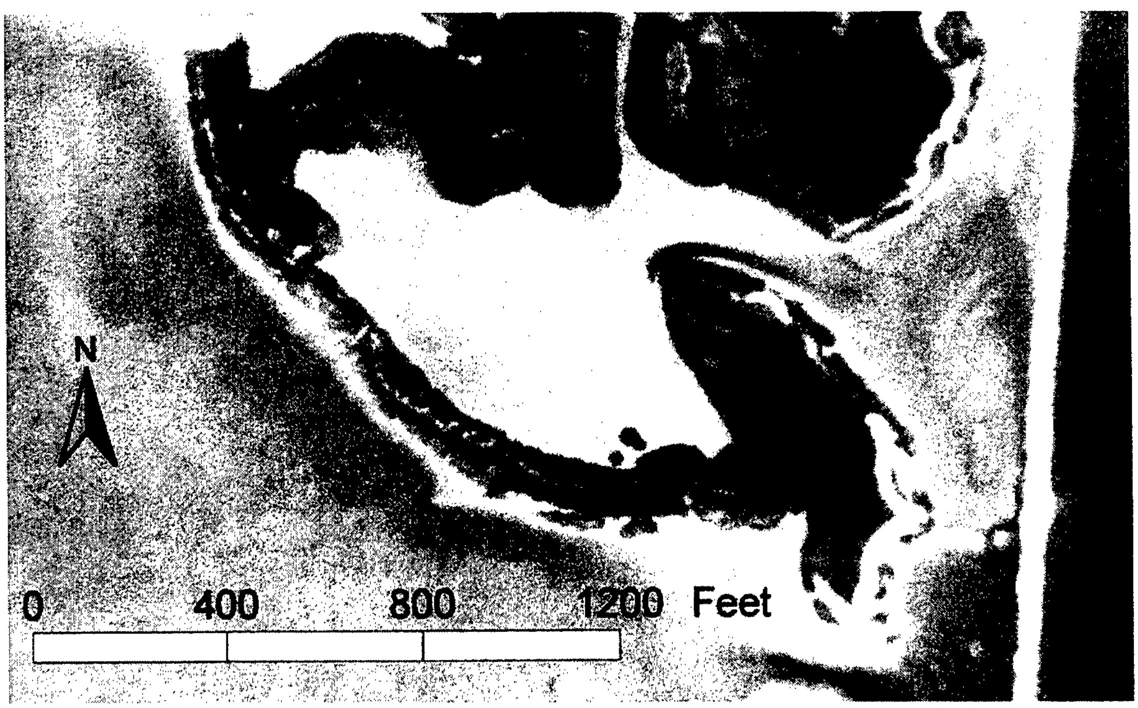


Figure D7. Example of created marsh with atypical geomorphic configuration (high edge:area ratio)

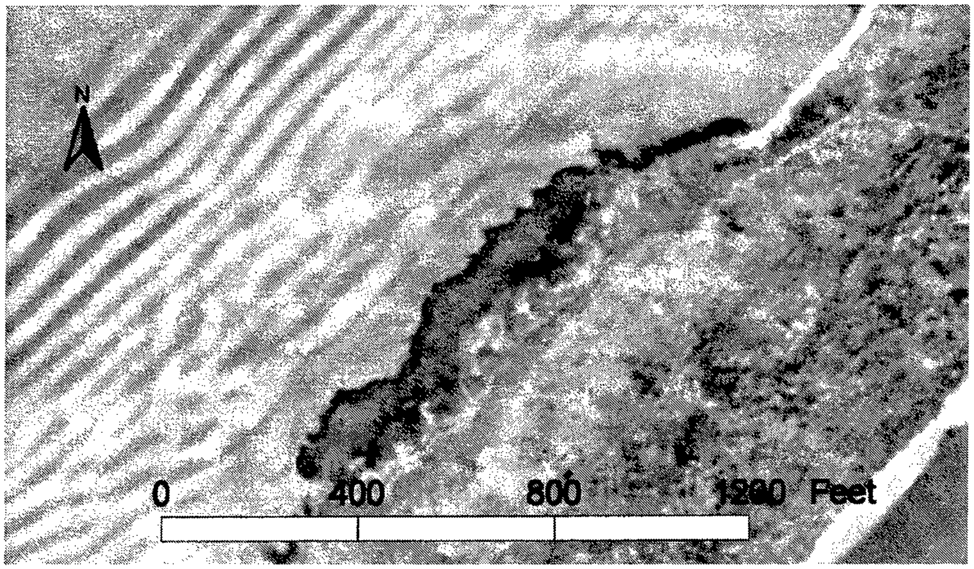


Figure D8. Example of created marsh with moderate edge:area ratio

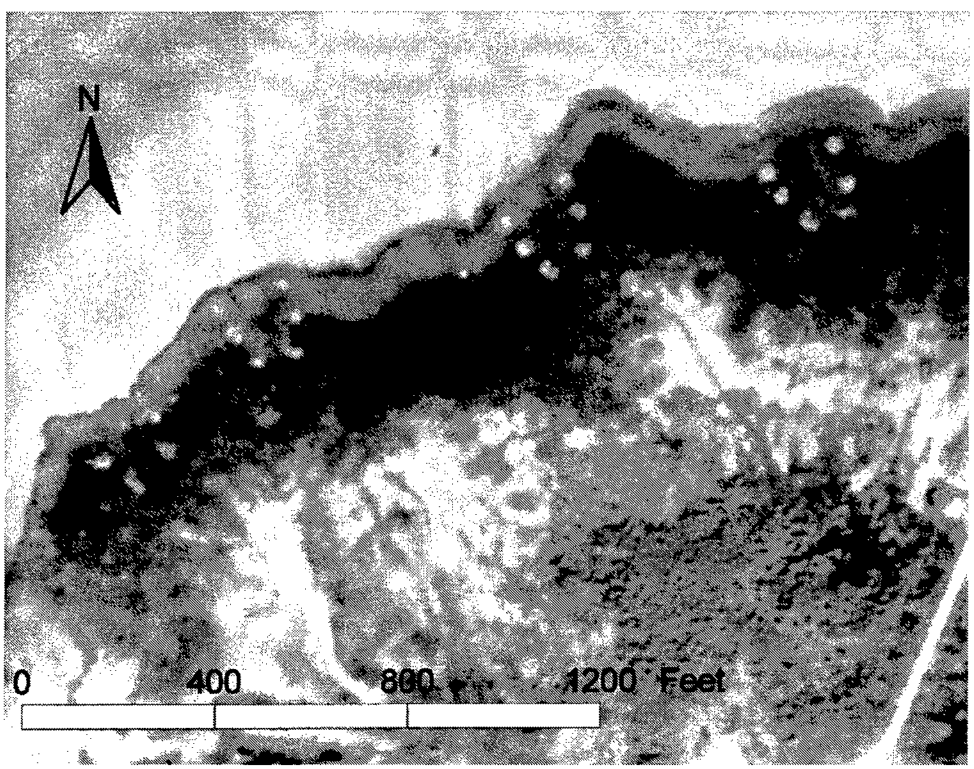


Figure D9. Example of created marsh that lacks tidal creeks (low edge:area ratio)

Table D6
Plants Characteristic of Northwest Gulf of Mexico Tidal Fringe Wetlands

Scientific Name	Common Name	Salinity Regime and Region		
		Salt/Brackish	Intermediate	Region[1]
Avicennia germinans	Black mangrove	L[2]		CB,LM
Spartina alterniflora	Saltmarsh cordgrass	L		MC,CB
Juncus roemerianus	Needlegrass rush	L		MC
Batis maritima	Saltwort	L,H		All
Lycium carolinianum	Carolina wolfberry	L,H		All
Monanthochloe littoralis	Keygrass	L,H		All
Salicornia virginica	Virginia glasswort	L,H		All
Scirpus maritimus	Saltmarsh bulrush	L,H		All
Distichlis spicata	Seashore saltgrass	L,H		All
Aster tenuifolius	Perrenial saltmarsh aster	L,H		All
Sporobolus virginicus	Seashore dropseed	L,H		All
Sesuvium portulacastrum	Sea-purslane	H		All
Heliotropium curassavicum	Seaside heliotrope	H		All
Suaeda spp.	Seepweed	H		All
Salicornia bigelovii	Dwarf glasswort	H		All
Limonium carolinianum	Sea-lavender	H		All
Iva frutescens	Bigleaf sumpweed	H		All
Rumex chrysocarpus	Amamastla	H		All
Borrichia frutescens	Sea oxeye	H	H	All
Spartina patens	Saltmeadow cordgrass	H	H	All
Spartina spartinae	Gulf cordgrass	H	H	All
Scirpus pungens	Three-square bulrush	L	L	All
Scirpus americanus	Olney's bulrush	L	L	MC
Bacopa monnieri	Coastal water-hyssop	L	L	All
*Typha domingensis**	Southern cattail	L	L	All
Fimbristylis castanea	Marsh fimbry	L,H	L,H	All
Spartina cynosuroides	Big cordgrass	L,H	L,H	MC(T-SJ)
Paspalum vaginatum	Seashore paspalum	L,H	L,H	All
Aster subulatus	Annual saltmarsh aster	L,H	L,H	All
*Phragmites australis**	Common reed	H	L,H	All
Hydrocotyle bonariensis	Coastal plain pennywort	H	L,H	All

(Continued)

[1] MC = Mid-Coast = Trinity – San Jacinto, Brazos – San Bernard, Lavaca – Colorado, and Guadalupe – San Antonio estuaries. CB = Coastal Bend = Mission – Aransas, and Nueces estuaries LM = Laguna Madre estuary.
[2] Water regime: L = low (proximal, regularly flooded); H = high (distal, irregularly flooded).
* Species that are invasive or may indicate eutrophication or other degenerative process (after Chabreck and Condrey 1979; Correll and Johnston 1979; Longley 1994; Kartesz 1994; Moulton 1998; Pulich 1990; Reed 1988; White et al. 1993; White et al. 1983 et seq.).

Table D6 (Concluded)

Scientific Name	Common Name	Salinity Regime and Region		
		Salt/Brackish	Intermediate	Region[1]
Solidago sempervirens	Seaside goldenrod	H	H	MC
Leptochloa spp.	Sprangle-top	H	H	All
Panicum virgatum	Switchgrass	H	H	All
*Baccharis halimifolia**	Eastern false-willow	H	H	All
Typha spp.*	Cattail		L	All
Cladium jamaicense	Jamaica sawgrass		L	MC,CB
*Alternanthera philoxeroides**	Alligator weed		L	MC
Sagittaria lancifolia	Bulltongue arrowhead		L	All
Eleocharis spp.	Spikerush		L	All
Scirpus californicus	California bulrush		L	All
Echinochloa spp.	Barnyard grass		L,H	All
Cyperus spp.	Flatsedge		L,H	All
*Sesbania drummondii**	Drummond's rattlebush		L,H	All
Centella asiatica	Asian coinleaf		H	All
*Aster spinosus**	Spiny aster		H	All
Polygonum spp.	Smartweed		H	All

* Species that are invasive or may indicate eutrophication or other degenerative process (after Chabreck and Condrey 1979; Correll and Johnston 1979; Longley 1994; Kartesz 1994; Moulton 1998; Pulich 1990; Reed 1988; White et al. 1993; White et al. 1983 et seq.).

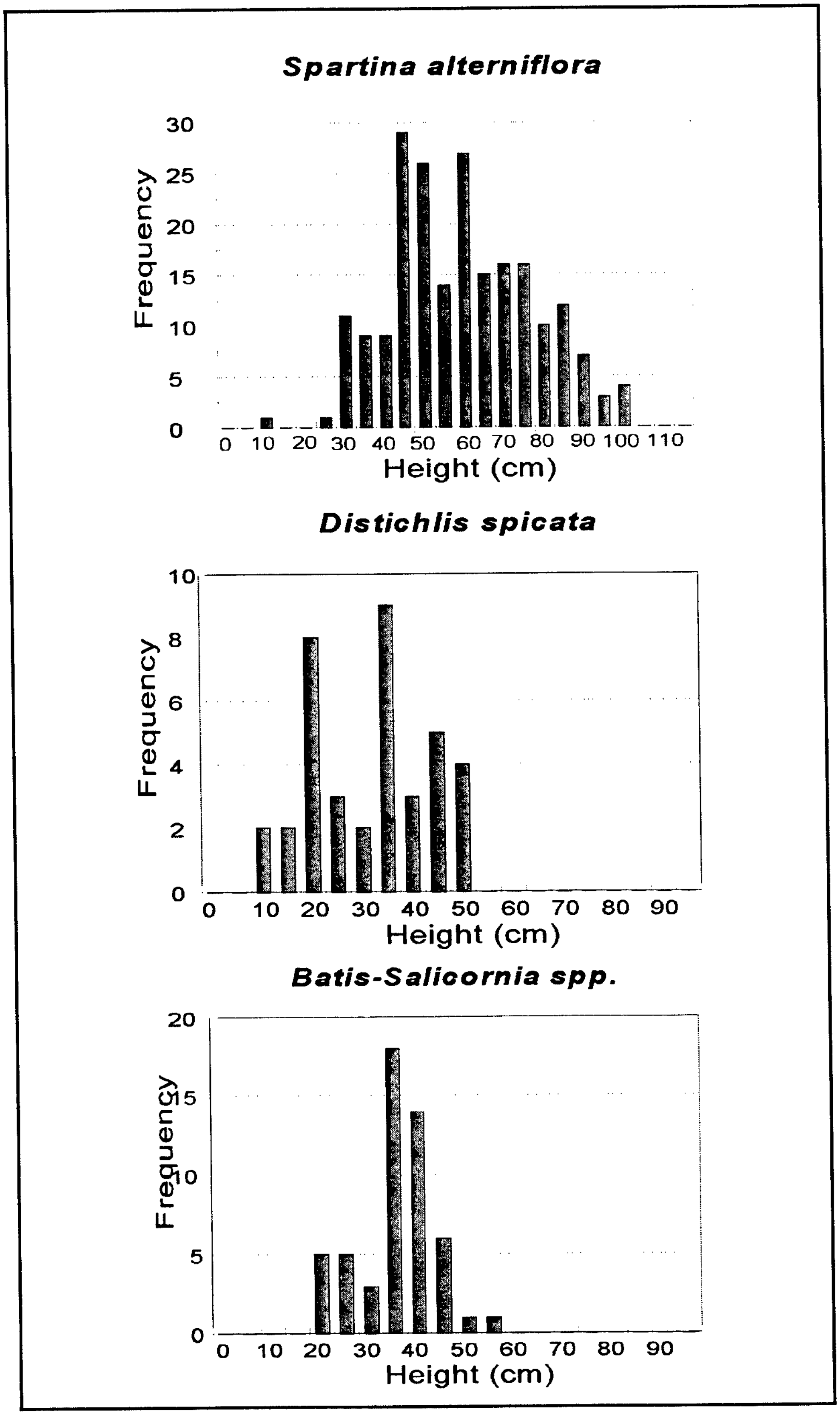


Figure D10. Frequency distributions of vegetation height for selected dominant saltmarsh species

Appendix E
Supplementary Information on Model Variables

This appendix contains the following summaries:

a. Mean annual wind speed by direction for selected sites along the Texas coast.

b. Soil texture by feel.

Table E1
Mean Annual Wind Speed (km/hr) and Proportion of Time Wind Blew from 16 Directions for Selected Sites (1997 data[1])

	Mid-Coast Region Sites							
	Morgans Point		Eagle Point		Port Bolivar		Galveston Pleasure Pier	
Wind Direction	Mean wsd (km/hr)	% Frequency	Mean wsd (km/hr)	% Frequency	Mean wsd (km/hr)	% Frequency	Mean wsd (km/hr)	% Frequency
N	15.86	0.04	14.26	0.06	21.09	0.04	19.72	0.08
NNE	13.35	0.05	16.49	0.09	19.84	0.07	20.20	0.05
NE	12.20	0.05	18.79	0.08	13.11	0.08	23.55	0.06
ENE	13.96	0.05	17.30	0.08	13.34	0.06	21.20	0.07
E	13.98	0.05	17.49	0.06	11.10	0.10	20.90	0.09
ESE	14.52	0.05	17.73	0.08	10.42	0.09	18.94	0.08
SE	14.78	0.09	16.46	0.10	14.66	0.11	19.63	0.10
SSE	12.83	0.09	20.03	0.09	18.76	0.10	19.74	0.10
S	13.41	0.07	19.68	0.06	17.45	0.10	20.24	0.09
SSW	10.26	0.05	20.47	0.07	15.77	0.04	19.49	0.04
SW	6.59	0.04	10.66	0.06	12.17	0.02	13.23	0.02
WSW	6.67	0.03	5.33	0.04	12.71	0.01	12.96	0.02
W	8.44	0.02	4.34	0.04	16.24	0.03	13.43	0.03
WNW	12.28	0.03	4.84	0.03	15.11	0.03	13.33	0.02
NW	16.03	0.03	7.05	0.02	20.57	0.05	14.70	0.04
NNW	12.48	0.02	10.25	0.03	22.22	0.02	17.65	0.05
NA	0	0.24	0.00	0.02	0.00	0.05	0.00	0.07

	Coastal Bend Region Sites							
	Freeport		Port O'Connor		Port Aransas		Ingleside Naval Air Station	
Wind Direction	Mean wsd (km/hr)	% Frequency	Mean wsd (km/hr)	% Frequency	Mean wsd (km/hr)	% Frequency	Mean wsd (km/hr)	% Frequency
N	18.44	0.05	31.04	0.06	21.59	0.06	24.58	0.06
NNE	16.90	0.08	25.19	0.05	21.47	0.08	22.86	0.08
NE	15.43	0.06	19.68	0.04	19.01	0.06	20.07	0.10
ENE	17.33	0.04	16.76	0.03	16.04	0.06	18.47	0.11
E	15.28	0.06	17.52	0.05	14.00	0.07	19.01	0.18
ESE	15.46	0.08	20.14	0.12	12.06	0.11	19.19	0.19
SE	16.08	0.09	22.69	0.23	13.45	0.19	16.25	0.07
SSE	15.28	0.11	21.49	0.17	11.05	0.13	8.58	0.03
S	18.65	0.13	17.00	0.08	13.06	0.07	13.01	0.02
SSW	16.30	0.10	13.88	0.04	6.15	0.02	18.48	0.02
SW	14.95	0.06	10.17	0.01	8.06	0.01	21.31	0.02
WSW	11.69	0.01	7.10	0.01	10.03	0.01	23.03	0.02
W	9.69	0.02	7.76	0.01	10.39	0.01	21.61	0.02
WNW	10.79	0.01	16.47	0.01	17.03	0.02	25.19	0.01
NW	17.95	0.02	23.37	0.02	18.23	0.03	26.65	0.00
NNW	19.03	0.06	33.11	0.04	21.60	0.04	28.31	0.01
NA	0.00	0.02	0.00	0.02	0.00	0.03	0.00	0.07

[1] Percent frequency was calculated from the hourly records for each of the 16 compass directions and was also computed for hourly records where no wind direction was given (NA).

Soil Texture by Feel

Clay content in soils can be measured in a laboratory by conducting a particle size analysis. However, this is often impractical in a rapid assessment scenario. Clay content can be estimated in the field using the soil texture by feel to determine the texture class (Figure E1), and the soil texture triangle to estimate percent clay (Figure E2).

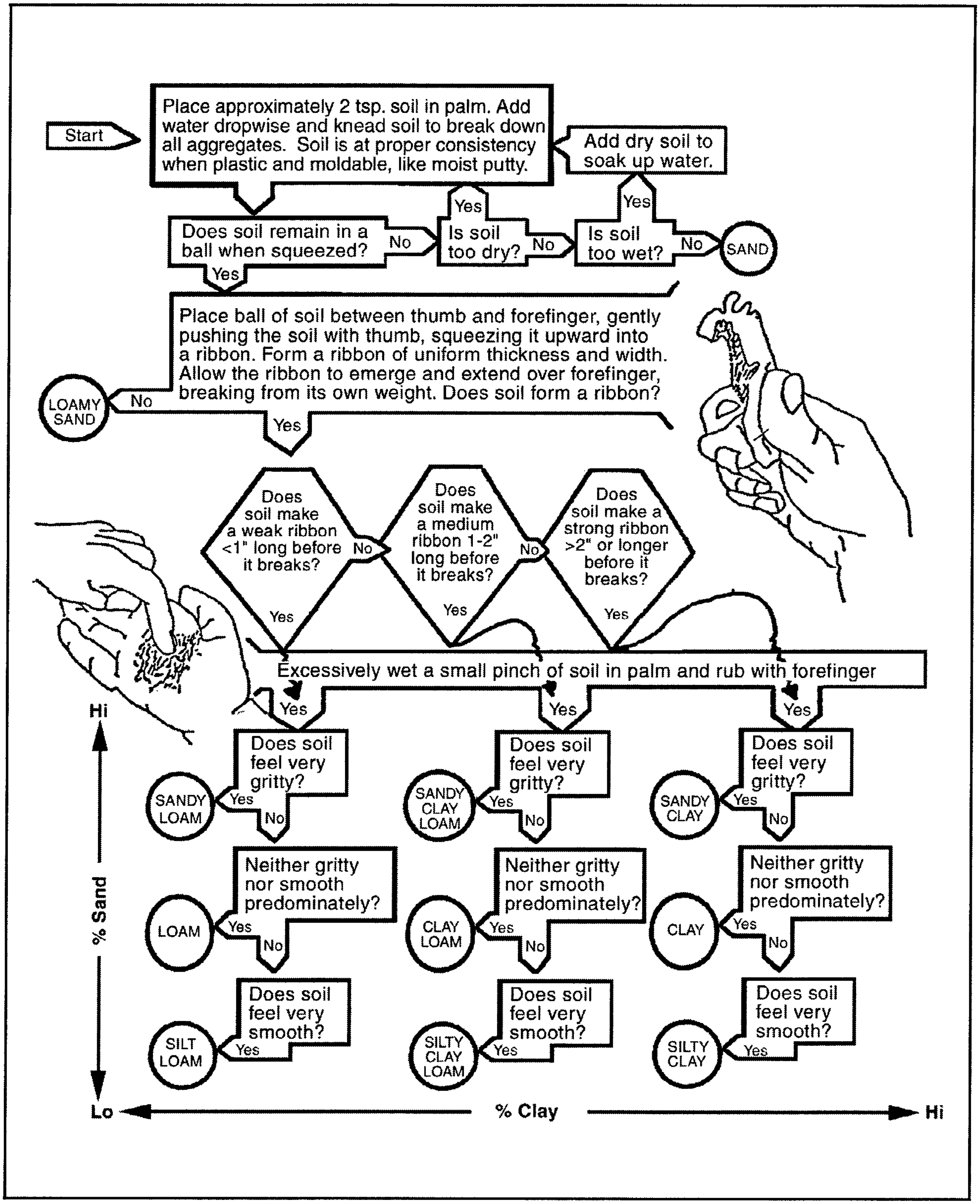


Figure E1. Estimating soil texture by "feel" (adapted from Thein (1979))

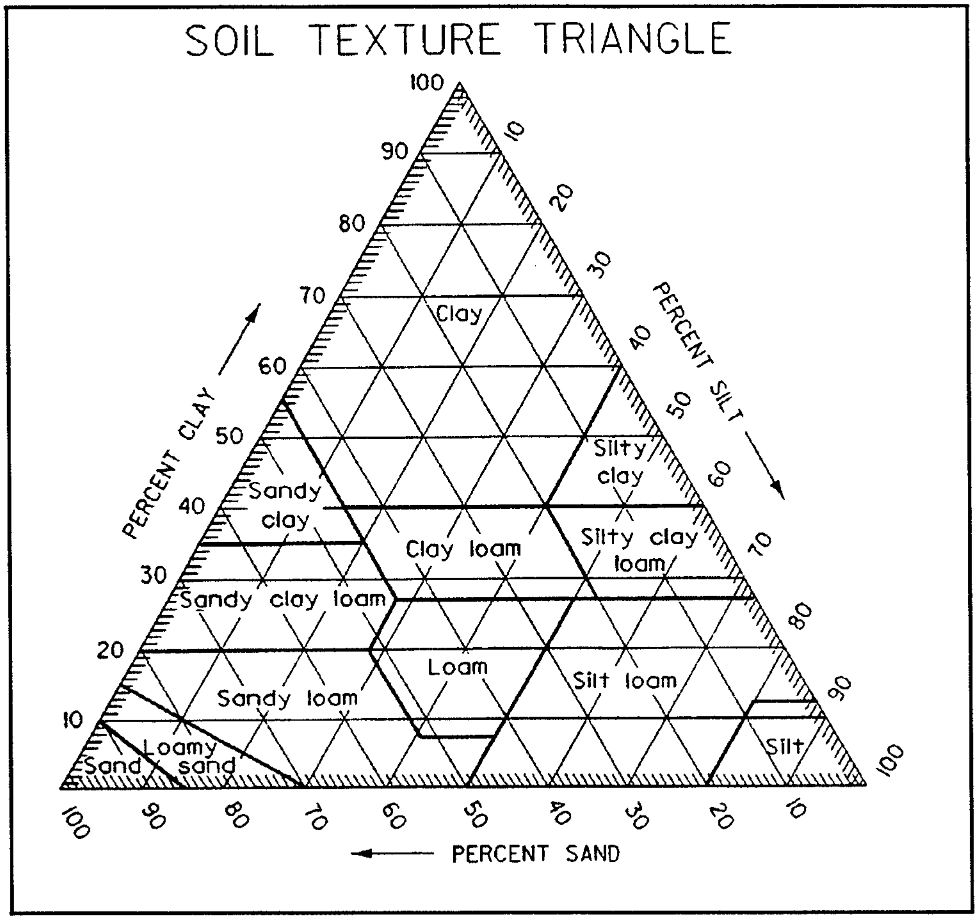


Figure E2. Soil texture triangle

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14. ABSTRACT

The Hydrogeomorphic (HGM) Approach is a collection of concepts and methods for developing functional indices and subsequently using them to assess the capacity of a wetland to perform functions relative to similar wetlands in a region. The approach was initially designed to be used in the context of the Clean Water Act Section 404 Regulatory Program permit review sequence to consider alternatives, minimize impacts, assess unavoidable project impacts, determine mitigation requirements, and monitor the success of mitigation projects. However, a variety of other potential applications for the approach have been identified including: determining minimal effects under the Food Security Act, designing mitigation projects, and managing wetlands.

This report uses the HMG Approach to develop a Regional Guidebook for assessing the functions of tidal fringe wetlands along the Texas Gulf coast, then discusses (a) the rationale used to select functions, (b) the rationale used to select model variables and metrics, (c) the rationale used to develop assessment models, and (d) the data from reference wetlands used to calibrate model variables and assessment models. Finally, it outlines an assessment protocol for using the model variables and functional indices to assess tidal fringe wetlands in the northwestern Gulf.

15. SUBJECT TERMS
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