

Memorandum



Date: February 1, 2022

To: Honorable Chairman Jose 'Pepe' Diaz
and Members, Board of County Commissioners

Agenda Item No. 5(G)

From: Lourdes M. Gomez, Director 
Department of Regulatory and Economic Resources

Subject: Class I Permit Application by John Dubois

Resolution No. R-93-22

Recommendation

I have reviewed the attached application for a Class I permit by John Dubois and based upon the applicable evaluation factors set forth in Section 24-48.3 of the Code of Miami-Dade County (Code), I recommend that the Board of County Commissioners (Board) deny the issuance of a Class I permit for the reasons set forth below.

Pursuant to Ordinance No. 16-73, this quasi-judicial matter may be submitted directly for placement on the Board's meeting agenda by the Director of the Department of Regulatory and Economic Resources.

Scope

The project site is located at 17575 Old Cutler Road, Palmetto Bay, in Commission District 8, which is represented by Commissioner Danielle Cohen Higgins.

Delegation of Authority

This item has no delegation of authority. Upon approval of this item, Section 24-48.2(C)(2) of the Code authorizes the Department to issue the permit, provided that other Code requirements are satisfied, such as submittal of final plans and bonds, and to require additional conditions, limitations, and restrictions if consistent with this Board's approval.

Fiscal Impact/Funding Source

This resolution relates to a regulatory application and does not have a fiscal impact.

Track Record/Monitor

The Coastal Resources Section Manager, McKee Gray, in the Department of Regulatory and Economic Resources, Division of Environmental Resources Management (DERM), will be responsible for monitoring the proposed project.

Background

The subject Class I permit application requests authorization to fill approximately 10,500 square feet of tidal waters in Biscayne Bay at 17575 Old Cutler Road, Palmetto Bay, Miami-Dade County, Florida. The scope of work also involves the installation of a concrete seawall, concrete king piles and concrete batter piles and limerock riprap boulders. The proposed project is required to be reviewed by the Board at a public hearing because filling within tidal waters is specifically referenced in Section 24-48.2 of the Code as work that shall be processed with a standard form application, including a public hearing.

After review of the proposed project based on the evaluation factors listed in Section 24-48.3(1) of the Code and the dredge and fill criteria outlined in Section 24-48.3(2)(a)-(f), as discussed herein, DERM is recommending denial of the subject application. The project as designed would not avoid and minimize impacts to tidal waters or coastal wetlands, and the proposed work is reasonably expected to result in adverse environmental impacts and cumulative adverse environmental impacts. The proposed work is not reasonably expected to provide additional shoreline protection; moreover, the proposed filling will have a detrimental impact to the habitat within the project area.

The subject 8-acre property is the site of a single-family residence which fronts Biscayne Bay in the Village of Palmetto Bay. The property abuts the Biscayne Bay Aquatic Preserve and is situated less than one hundred yards north of the boundary of Biscayne National Park. The shoreline of the subject property is primarily unconsolidated and stabilized by halophytic (i.e. salt tolerant) plant species, predominantly mangrove trees, consistent with the neighboring properties. The project area is not unlike other unconsolidated shorelines or tidal flat habitats throughout Biscayne Bay. A small portion of the shoreline located south of the single-family residence is stabilized by an existing concrete and limerock seawall (see Aerial of Subject Property, Attachment A). The proposed project area is shown on Attachment B.

On January 31, 2019, DERM staff biologists conducted a biological assessment (Attachment C) of the proposed project area. The project area is shallow with portions of the submerged lands exposed at low tide, which is typical of intertidal areas associated with mangrove fringes. The shoreline landward and to the south of the proposed fill footprint is predominantly vegetated with red mangroves (*Rhizophora mangle*), white mangroves (*Laguncularia racemosa*), and black mangroves (*Avicennia germinans*). Within the submerged portions of the proposed project areas, sparse shoal grass (*Halodule wrightii*) and macroalgae consisting of brown algae (*Sargassum spp.*) and green algae (*Batophora spp.*) were observed. The project area also contained organic detritus, which is decomposing organic matter produced by the natural breakdown of such things as marine vegetation.

The seagrass and macroalgae documented during the biological assessment represent important components of benthic habitat that are typical of the tidal system. Shallow tidal areas such as these are used by various wildlife including avian species. During the biological assessment at the subject property, DERM staff observed several avian species foraging in the project area including a green heron, a juvenile little blue heron (a State designated threatened species), and white ibis.

Mangroves and their root systems, including those at the subject property, are integral to shoreline stability in coastal areas and are an important component of the Biscayne Bay ecosystem. Mangroves and their root systems help minimize erosion through the stabilization of sediment, provide habitat for marine and avian fauna, increase water quality and clarity by filtering nutrients and other pollutants, and provide organic detritus to the marine trophic system as recognized in scientific literature (see three scientific articles related to mangroves, Attachment D). In addition, shorelines vegetated with mangroves, similar to those at the subject property and throughout Biscayne Bay, act as a form of natural protection to storm surge during damaging storm events, bearing the initial brunt against powerful winds and waves, as further supported by scientific literature (see scientific article related to storm surge and mangroves, Attachment E).

The organic detritus documented during the biological assessment is consistent with the natural accumulation of detritus within mangrove fringes and shallow tidal areas. The decomposition of organic detritus provides essential nutrients that form the foundation of the food web. These nutrients play a vital role in not only the ecology of the mangrove system itself but also adjacent

seagrass and coral reef communities and the overall health of Biscayne Bay (see scientific articles related to mangrove detritus, Attachment D). In addition, as the organic detritus breaks down, the resulting deposits settle to the substrate and create new habitat. This new habitat can support the growth of submerged aquatic vegetation such as seagrasses or emergent vegetation in the form of mangroves.

Proposed Work and Evaluation

The proposed scope of work for this Application involves the filling of approximately 10,500 square feet of tidal waters within Biscayne Bay to an elevation of +2.09 feet National Geodetic Vertical Datum (NGVD). The Applicant proposes to contain the fill by installing a concrete panel seawall along the east side of the project area (project area shown on Attachment B) and a three-foot-wide limerock riprap boulder revetment along the north, west, and south sides of the fill area, which would be situated between the proposed fill area and the existing mangrove fringe. The proposed seawall is proposed to have concrete king piles, concrete batter piles, and a row of limerock riprap boulders to be installed waterward of the new seawall. The northern portion of the new seawall is intended to connect to an existing seawall and the southern extent of the new seawall would terminate at the mangrove fringe. The Applicant has stated that the purpose of the request to fill tidal waters is to restore a segment of the property’s shoreline to what the Applicant asserts existed in 1938. The proposed project would result in filling approximately 0.25 acres of submerged bay bottom lands of Biscayne Bay. The Applicant further asserts that the proposed project will protect the property from hurricanes, eliminate stagnant water, and enhance the adjacent waters.

DERM staff completed an evaluation of the proposed work including a review of historic aerial photographs of the project location and previous Class I permit application records, along with the Applicant’s submittals. In addition, DERM considered both historical and existing site conditions when evaluating the project. Although the Applicant asserts that the purpose of the project is to restore the shoreline, a review of historic aerials (see historical aerials, Attachment F) revealed that the shoreline at the project location has been stabilized by the mangrove fringe and associated root systems in approximately the same configuration for over 50 years. Accordingly, the Applicant’s assertion regarding the configuration of the shoreline in 1938 is irrelevant and insufficient to overcome the Code criteria for the filling of tidal waters. Moreover, DERM staff notes that the 1938 aerial photograph that the Applicant relies on is inconclusive as to the location of the shoreline at that time. In addition, property records show that the Applicant purchased the property in the year 2000 with the shoreline in the current configuration.

Section 24-48.3(1) of the Code provides that DERM shall base its recommendation on this Application, and the Board shall make its decision, based on certain evaluation factors, when applicable. These evaluation factors are listed in Section 24-48.3(1) of the Code and include but are not limited to, the potential adverse environmental impact and cumulative adverse environmental impacts of the proposed project including but not limited to the effect upon hydrology, water quality, water supply, wellfields, aquifer recharge, aesthetics, navigation, public health, historic values, air quality, marine and wildlife habitats, archeological values, wetlands soils suitable for habitat, floral and faunal values, rare, threatened, and endangered species, natural flood damage protection, wetland values, land use classification, recreation, and any other environmental values affecting the public interest.

Based on a review of these evaluation factors, this Application should be denied, as explained in more detail herein. As proposed, adverse environmental impacts would likely occur as a result of the filling of the project area and construction of a new seawall and riprap revetment at the subject property. The proposed filling would alter a functioning intertidal system by eliminating the existence of seagrass and wetland habitat, thereby diminishing the wetland value of this area. It is likely that the proposed filling would also result in depleted floral and faunal values for various species by eliminating a high functioning intertidal system. The installation of the proposed seawall and the filling of the project area is not likely to provide additional protection during storm events or provide flood protection of the property as the elevation of the seawall cap would likely not be sufficient to prevent the continual erosion of fill as a result of tidal events (see excerpts of the Public Works Manual, Attachment G).

Although the construction of the project does not include the trimming or removal of mangroves, the project is expected, over time, to negatively impact the existing mangroves onsite for the following reasons. The project is likely to result in minimized tidal exchange to the mangroves growing along this section of the shoreline. This reduced tidal flushing is reasonably expected to have negative impacts to the mangroves and mangrove ecosystem. In addition, the filling of the project area, including the placement of riprap immediately adjacent to the mangroves, is reasonably expected to accelerate sedimentation within the mangrove fringe landward of the fill footprint and would likely result in the degradation of the existing mangrove habitat by minimizing the area for tidal exchange to the mangroves growing along this section of the shoreline.

In addition to the negative environmental impacts discussed above, there is also the potential for several secondary environmental impacts due to the proposed project design and construction methodology. The project as designed does not adequately address the containment or grading of the proposed fill. Based on the proposed seawall cap elevation, it is reasonably expected that tidal waters will wash over the seawall during high tides and storm events. As a result of these tidal events, it is reasonably expected that the proposed seawall configuration will result in the continual erosion of fill into tidal waters during an outgoing tide, or during extreme marine weather, thus negatively affecting water quality and disrupting the hydrology of the project area and possibly beyond. Long term erosion may affect the benthic resources documented waterward of the proposed seawall and the mangrove wetlands that border the proposed riprap revetment. Furthermore, the construction methodology submitted raises several concerns including the feasibility of conducting the proposed work by water, how the construction operations will avoid and minimize potential direct and secondary impacts to benthic resources, and whether adequate water depth exists for the barge and any associated vessels to navigate through the area. Although if approved, the Class I permit would require standard turbidity controls during all phases of construction, the above mentioned concerns relate to the long term and continued adverse environmental impacts associated with the project, beyond the construction phase of the project.

In addition, Section 24-48.3(2) of the Code provides that dredging and filling work proposed in Class I permit applications shall comply with at least one of the criteria listed below:

- a) Minimum dredging and spoiling for public navigation or public necessity.
- b) An alteration of physical conditions as may be necessary to enhance the quality or utility of adjacent waters
- c) Minimum dredging and filling for the creation and maintenance of marinas, piers, docks and attendant navigational channels.

- d) Minimum dredging and filling as is necessary for the elimination of conditions hazardous to the public health or for the elimination of stagnant waters.
- e) Minimum dredging and filling as is necessary to enhance the biological, chemical or physical characteristics of adjacent waters.
- f) A physical modification necessary to protect public or private property.

For the reasons discussed herein, this Application is not consistent with any of the dredge and fill criteria listed in Section 24-48.3(2)(a)-(f) of the Code as referenced above, and therefore this Application should be denied. Dredge and fill criteria (a) does not apply as the project does not involve work relating to public navigation or public necessity. The project is not consistent with dredge and fill criteria (b) because the filling and alteration of the shoreline would not enhance the quality or utility of adjacent waters. The proposed work would likely result in the discharge of fill material into the adjacent mangrove and benthic habitats, thereby decreasing overall water quality. Dredge and fill criteria (c) does not apply as the project does not involve the creation and maintenance of marinas, piers, docks and attendant navigational channels. The project is not consistent with dredge and fill criteria (d) because the proposed filling is not necessary for the elimination of conditions hazardous to the public health or for the elimination of stagnant waters. In fact, the project area does not contain hazardous conditions or stagnant waters, and is comprised of organic detritus that is consistent with the natural accumulation of detritus within mangrove fringes and shallow tidal areas. The nutrients produced from the decomposition of organic detritus provides essential nutrients that form the foundation of the food web and play a vital role in the ecology of the mangrove system, the adjacent seagrass and coral reef communities, and the overall health of Biscayne Bay. The project is not consistent with dredge and fill criteria (e) because the project will have a negative effect on the biological, chemical, and physical characteristics of the adjacent waters as the proposed fill material will eliminate the benthic habitat within the fill footprint and it is likely that the fill material will discharge into the adjacent mangrove and benthic habitats, decreasing water quality, habitat, and biodiversity. The project is not consistent with dredge and fill criteria (f) because the filling is not necessary to protect private property. As previously discussed in this memorandum, the shoreline of the property has been stable for over 50 years. In addition, based on the elevation of the proposed seawall it is reasonably expected that tidal waters will wash over the seawall during high tides or extreme marine weather events. The existing mangroves at the property serve as a form of natural protection to storm surge during damaging storm events and the proposed work is likely to have a negative effect on the mangroves as detailed above. Based on DERM's evaluation, there are less impacting options for the Applicant to protect the property, including the installation of a retaining wall landward of the existing mangrove fringe. DERM concludes that the project as proposed is not consistent with any of the dredge and fill criteria.

Pursuant to Section 24-48.4 of the Code, potential adverse environmental impacts and cumulative adverse environmental impacts for a proposed project must be avoided and minimized. Mitigation is required to compensate for the loss of habitat and for water quality impacts associated with the filling of tidal waters and is calculated based on the functional loss of the habitat. In addition, mitigation is required to compensate for water quality impacts associated with the installation of a vertical seawall. Section 24-48.4 of the Code further states that mitigation should not be used to make an otherwise non-permittable project permittable, and must maximize preservation of existing natural resources including avoiding the impact altogether by not taking certain action or parts of an action, as well as minimizing impacts by limiting the degree or magnitude of the action or its implementation. As proposed, the project has not been designed to avoid and minimize impacts and will result in adverse environmental impacts and cumulative adverse impacts to the existing benthic

and wetland habitat. To mitigate for the proposed project, the Applicant proposes to use credits purchased from the Florida Power & Light Everglades Mitigation Bank (EMB). Should the Board approve the Application as proposed, the amount of compensatory mitigation required would be determined based on the W.A.T.E.R. - Wetland Assessment Technique for Environmental Reviews and would therefore require the purchase of 0.17 EMB saltwater credits.

Conclusion

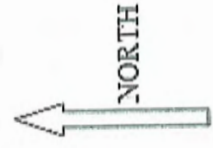
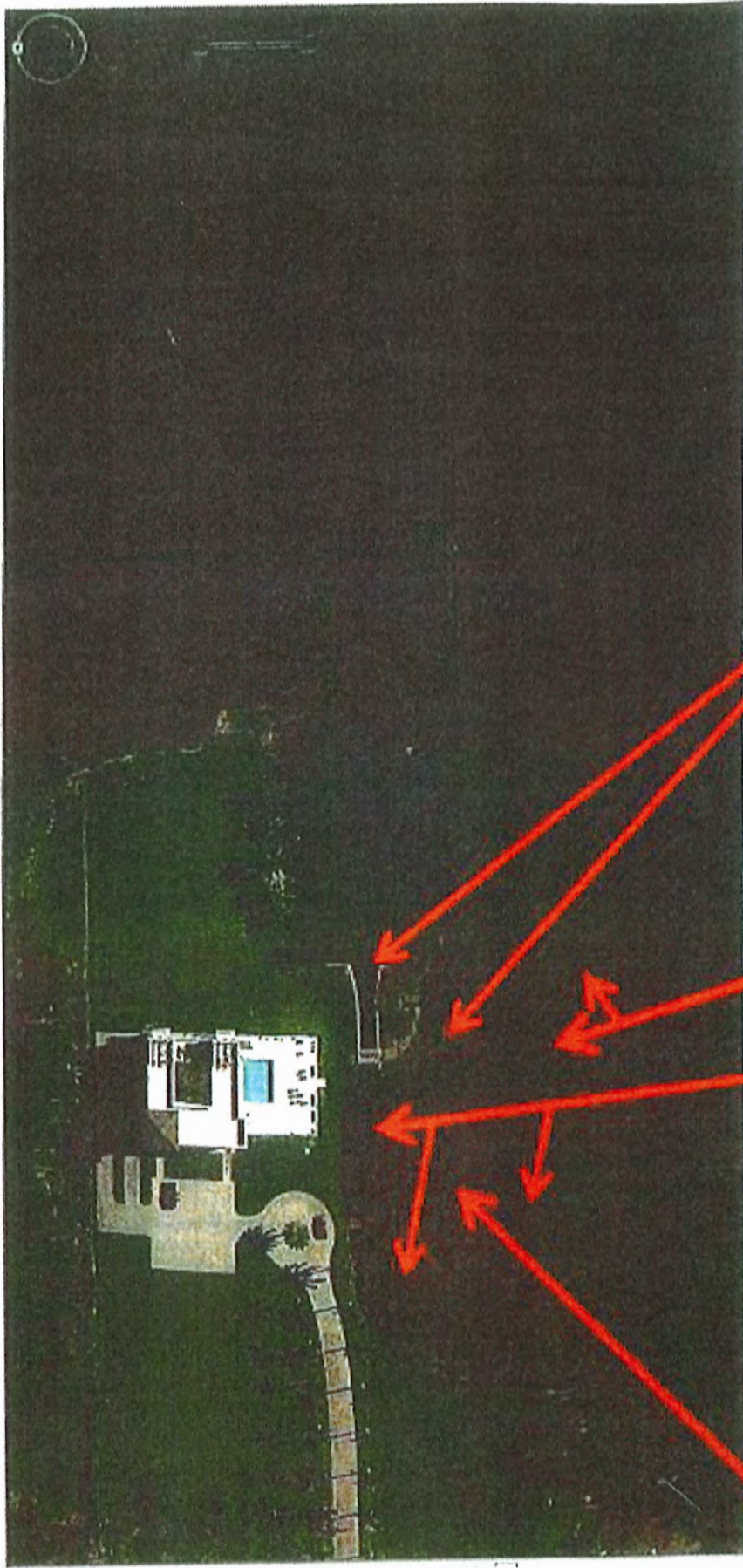
For all the reasons contained herein and in the incorporated project report, DERM recommends that the Application be denied and that the project be redesigned to avoid the filling of tidal waters.

Please find attached a DERM Project Report which sets forth additional reasons the proposed project is not recommended for approval by DERM pursuant to the applicable evaluation factors and criteria set forth in Section 24- 48.3 of the Code. The conditions, limitations, and restrictions set forth in the attached Project Report are incorporated herein by reference hereto.

Attachments

- Attachment A: Aerial of the Property
- Attachment B: Proposed Project Area
- Attachment C: DERM's Biological Assessment conducted on January 31, 2019
- Attachment D: The Habitat and Function of Mangroves for Terrestrial and Marine Fauna: A Review; Influence of Mangrove Detritus in an Estuarine Ecosystem; The Contribution of Mangrove Swamps to Florida Fisheries
- Attachment E: Storm Surge Reduction by Mangroves
- Attachment F: Historical Aerials
- Attachment G: Excerpts from the Miami-Dade County Public Works Manual
- Attachment H: Class I Permit Application
- Attachment I: Owner/Agent Letter, Engineer Letter, and Project Sketches
- Attachment J: Zoning Memorandum
- Attachment K: Names and Addresses of Owners of All Riparian or Wetland Property within Three Hundred (300) Feet of the Proposed Work
- Attachment L: DERM Project Report

Attachment A
Aerial of the Property



Existing Concrete and
Limerock Seawalls

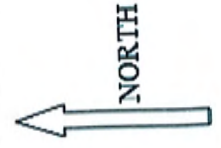
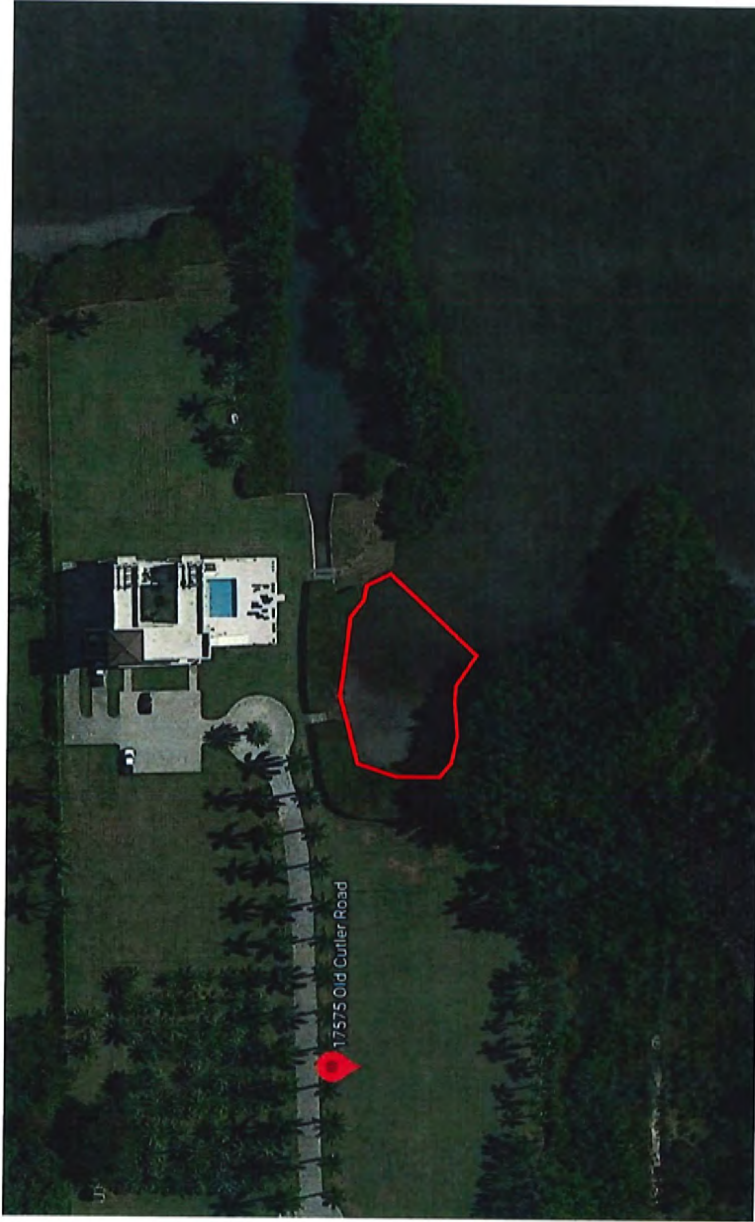
Shallow Areas Waterward
of Proposed Fill Footprint

Mangroves Landward of Proposed Fill
Footprint

Proposed
Fill Area

Attachment B
Proposed Project Area

Attachment B
Proposed Project Area



Attachment C

DERM's Biological Assessment conducted on January 31, 2019

Biological Assessment

File Number: CL-2018-0467-Dubois
(See corresponding aerial)

Address:
17575 Old Cutler Road
Palmetto Bay, Florida 33157

Date – Time: January 31, 2019 – 12:20pm-1:00pm

Biologists: Rockell Alhale/ Lourdes Barralli

Area A: Approximately 97 linear feet (measured from aerial) of red mangroves, white mangroves, black mangroves.

Area B: Approximately 169 linear feet (measured from aerial) of red mangroves, black mangroves, and white mangroves measuring 7 feet high.

Wood ramps (remain unchanged from August 26, 2015 assessment) previously documented measurements as follows:

- 16.7 feet long by 3.4 feet wide
- 17.3 feet long by 4.3 feet wide
- 16.1 feet long by 8.1 feet wide

Area C

1 foot wide concrete seawall on north and east sides (remains unchanged from August 26, 2015 assessment)
89.5 linear feet of rock wall on northwest, west, and south sides (remain unchanged from August 26, 2015 assessment)
Predominantly St. Augustine grass, sea purslane, crab holes also observed.

Area D (project area landward of the proposed seawall):

Green algae, Sargassum, and Shoal grass.
Red mangrove propagules rooted in substrate.

Area E

Red mangroves over 30 feet high, east of Area A

Tide Adjustment: 0.4 feet at 12:00pm / 0.3 feet at 12:30 pm

Water depths within project area adjusted to N.O.A.A. Mean Low Water datum are less than 1.5 feet

Flora

Red Mangroves (*Rhizophora mangle*)
White Mangroves (*Laguncularia racemosa*)
Black Mangroves (*Avicennia germinans*)
Smut grass (*Sporobolus indicus*)
St. Augustine (*Stenotaphrum secundatum*)
Hurricane grass (*Fimbristylis cymosa*)
Sea purslane (*Sesuvium portulacastrum*)
Green algae (*Barophora* sp.)
Sargassum (*Sargassum* sp.)
Shoal grass (*Halodule wrightii*)

Fauna

Green heron (*Butorides virescens*)
Juvenile little blue heron (*Egretta caerulea*)
White ibis (*Eudocimus albus*)





Area E - red mangroves / Location of proposed seawall.



South corner of Area A and north corner of Area E



East corner of Area B

Mangroves along the rock seawall south of Area C



Facing southeast from Area C



Area C





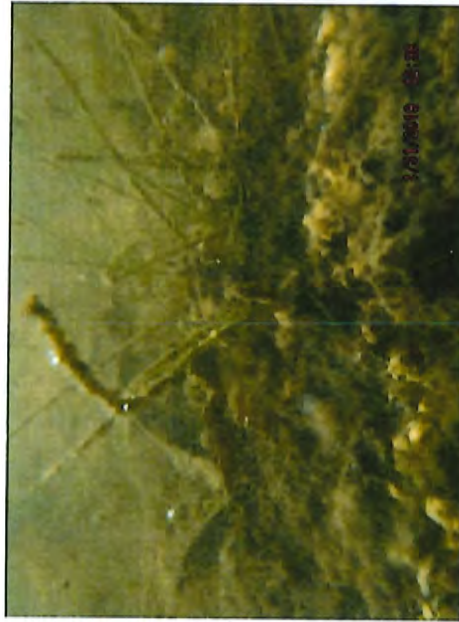
Area C



Area D



Area D



Area D





Ibis, juvenile little blue heron, and green heron



Area E - red mangroves



Location of proposed seawall

Area E - red mangroves



Area D / Adjacent to red mangroves from Area E





Area D - location of proposed seawall



Area B - mangroves



Area B - wood ramp

Attachment D

The Habitat and Function of Mangroves for Terrestrial and Marine Fauna: A Review; Influence of Mangrove Detritus in an Estuarine Ecosystem; The Contribution of Mangrove Swamps to Florida Fisheries

Review

The habitat function of mangroves for terrestrial and marine fauna: A review

I. Nagelkerken^{a,*}, S.J.M. Blaber^b, S. Bouillon^{c,d}, P. Green^e, M. Haywood^f,
L.G. Kirton^g, J.-O. Meynecke^h, J. Pawlikⁱ, H.M. Penrose^j,
A. Sasekumar^k, P.J. Somerfield^l

^a Department of Animal Ecology and Ecophysiology, Institute for Wetland and Water Research, Faculty of Science,
Radboud University, Toernooiveld 1, 6525 ED Nijmegen, The Netherlands

^b CSIRO Marine & Atmospheric Research, P.O. Box 120, Cleveland, Queensland 4130, Australia

^c Department of Analytical and Environmental Chemistry, Vrije Universiteit Brussel, Pleinlaan 2, B-1050 Brussels, Belgium

^d Netherlands Institute of Ecology, Centre for Estuarine and Marine Ecology (NIOO-KNAW), Yerseke, The Netherlands

^e Tucson Audubon, 300 East University Boulevard, Ste. 120, Tucson, AZ 85705, USA

^f CSIRO Division of Marine and Atmospheric Research, P.O. Box 120, Cleveland, Queensland 4160, Australia

^g Tropical Forest Biodiversity Centre, Forest Research Institute Malaysia, 52109 Kepong, Selangor, Malaysia

^h Australian Rivers Institute, and School of Environment, Griffith University, MB 50 GCMC, Queensland 9726, Australia

ⁱ Department of Biology and Marine Biology, Center for Marine Science, University of North Carolina Wilmington, Wilmington, NC 28409, USA

^j Centre for Marine Studies & School of Integrative Biology, The University of Queensland, St. Lucia, Queensland 4072, Australia

^k Institute of Biological Sciences, University of Malaya, 50603 Kuala Lumpur, Malaysia

^l Plymouth Marine Laboratory, Prospect Place, Plymouth PL1 3DH, UK

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Available online 8 December 2007

Abstract

Mangroves are defined by the presence of trees that mainly occur in the intertidal zone, between land and sea, in the (sub) tropics. The intertidal zone is characterised by highly variable environmental factors, such as temperature, sedimentation and tidal currents. The aerial roots of mangroves partly stabilise this environment and provide a substratum on which many species of plants and animals live. Above the water, the mangrove trees and canopy provide important habitat for a wide range of species. These include birds, insects, mammals and reptiles. Below the water, the mangrove roots are overgrown by epibionts such as tunicates, sponges, algae, and bivalves. The soft substratum in the mangroves forms habitat for various infaunal and epifaunal species, while the space between roots provides shelter and food for motile fauna such as prawns, crabs and fishes. Mangrove litter is transformed into detritus, which partly supports the mangrove food web. Plankton, epiphytic algae and microphytobenthos also form an important basis for the mangrove food web. Due to the high abundance of food and shelter, and low predation pressure, mangroves form an ideal habitat for a variety of animal species, during part or all of their life cycles. As such, mangroves may function as nursery habitats for (commercially important) crab, prawn and fish species, and support offshore fish populations and fisheries. Evidence for linkages between mangroves and offshore habitats by animal migrations is still scarce, but highly needed for management and conservation purposes. Here, we firstly reviewed the habitat function of mangroves by common taxa of terrestrial and marine animals. Secondly, we reviewed the literature with regard to the degree of interlinkage between mangroves and adjacent habitats, a research area which has received increasing attention in the last decade. Finally, we reviewed current insights into the degree to which mangrove litter fuels the mangrove food web, since this has been the subject of long-standing debate.

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Keywords: Mangrove; Habitat use; Fauna; Ecosystem interaction; Ecology; Biodiversity; Food web

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* Corresponding author. Tel.: +31 24 3652471; fax: +31 24 3652409.

E-mail address: i.nagelkerken@science.ru.nl (I. Nagelkerken).

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1. Introduction

Mangroves are predominantly intertidal habitats that occur worldwide in the (sub) tropics along sheltered and shallow-water coastlines. The prop-roots and pneumatophores of mangrove trees extend into the intertidal and subtidal where they become a rare feature: hard substrata in an otherwise soft sediment environment (Ellison and Farnsworth, 1992). As such, mangrove roots become home to terrestrial as well as marine plants, algae, invertebrates and vertebrates. Mangroves form a habitat for a wide variety of species, some occurring in high densities. They are productive habitats and may support coastal fisheries for prawns and fishes (Manson

et al., 2005). Mangroves are also important to humans for a variety of reasons, including aquaculture, agriculture, forestry, protection against shoreline erosion, as a source of fire-wood and building material, and other local subsistence use (Hogarth, 1999; Walters et al., 2008). Worldwide, loss of mangroves has been significant in recent decades, although in some regions of the world mangroves still occur as very extensive forests (Spalding, 1998; Alongi, 2002). They suffer from direct impacts such as cutting and pollution, as well as from hidden impacts such as changes in inland freshwater management (Dahdouh-Guebas et al., 2005), and are often regarded as unpleasant environments with little intrinsic value.

Animals found within mangrove environments include a variety of taxa, many of which are vulnerable or threatened as a result of human activities in the coastal zone. Determining the value of mangroves and other estuarine habitats for these animals requires knowledge of their life history, physiology and ecology as they interact across the dynamic mosaic of available habitats. Evidence suggests that mangroves are important to these species, but a lack of research is a major impediment to an evaluation of their mangrove dependency. A challenge for future research is separating the roles of mangroves from those of estuaries and other shallow-water habitats, to help determine the appropriate temporal and spatial scales for habitat protection (see Manson et al., 2005). Estuarine habitats have been recognised as important drivers of nearshore fish productivity. Worldwide, about 30% of all commercial fish species are mangrove-dependent (Naylor et al., 2000), producing an annual catch of almost 30 million tonnes in 2002 (FAO, 2004). Of all ecosystems, estuaries have the highest value per hectare (Costanza et al., 1997), making it significant for subsistence in many coastal communities. In Bragança (N-Brazil), for example, 68% of the cash income is primarily derived from mangrove crabs and fish (Glaser, 2003).

Recent and extensive reviews on mangroves as habitats for terrestrial and marine fauna include Hogarth (1999), Kathiresan and Bingham (2001), and Qasim and Kathiresan (2005). Studies related to the linkages between mangroves and coastal fish populations and fisheries, and new insights relating to the debate on the degree to which mangrove litter fuels the mangrove food web, form an important body of work published since these reviews; hence there is the need for a more up-to-date review. The current review summarises the available data on mangroves as a habitat for terrestrial and marine fauna, with special reference to the interlinkages with adjacent habitats and the importance of litter in the mangrove food web. We focus on the main groups of animals found in the mangrove habitat: sponges, various groups of meiofauna and macrofauna (epifauna and infauna), prawns, insects, fishes (bony fishes and elasmobranchs), amphibians, reptiles, and birds, accepting that a review of the complete fauna would be too far-reaching for this special issue, and that some mangrove fauna are not discussed here. These include less-well studied taxa like zooplankton (e.g., Mohan and Sreenivas, 1998; Ferrari et al., 2003; Krumme and Liang, 2004), tunicates (e.g., Carballo, 2000; Goodbody, 2003; Rocha et al., 2005), and mammals such as bats (Bordignon, 2006), buffalo (Dahdouh-Guebas et al., 2006), deer (Baird and Stirling, 2006), dolphins (Smith et al., 2006), flying foxes (Moore, 2002), manatees (Spiegelberger and Ganslosser, 2005), marsupials (Fernandes et al., 2006), otters (Angelici et al., 2005), primates (Nijman, 2000), rabbits (Forys and Humphrey, 1996), raccoons (Cuaron et al., 2004), and tigers (Gopal and Chauhan, 2006).

2. Mangroves as habitats for sponges

2.1. Diversity and distribution of sponges

Sponges occurring on mangrove roots are conspicuous because they often have large sizes (± 50 cm in diameter or

more) and brilliant colours (e.g., Rützler and Feller, 1996; Diaz et al., 2004). Although some encrusting mangrove sponges can survive above the water line for many hours during a tidal cycle (Barnes, 1999), the aquiferous system of larger sponges will collapse when emerged for periods longer than 4 h (Rützler, 1995). Considering the typical zonation of mangrove habitats (Nybakken, 1997), the mangrove area available to sponge communities is very small, especially when compared to the much larger adjacent subtidal habitats afforded by seagrass beds, hard-bottom areas, and coral reefs. In addition, while only prop-roots that extend below lowest low water (LLW) will support most sponge growth (Ellison and Farnsworth, 1992; Rützler, 1995), the vast majority of these roots are in water that is either too shallow or too stagnant. Most mangrove sponge assemblages are restricted to prop-roots that hang over tidal channels that cut through soft sediment habitats (e.g., Engel and Pawlik, 2005) or raised rock, rubble or peat banks that drop off abruptly to depths greater than LLW (e.g., Farnsworth and Ellison, 1996). These two types of shoreline are also accessible by boat and snorkelling, while the vastness of the remaining intertidal mangrove is not. Therefore, to infer that sponge-covered prop-roots are a common feature of mangrove habitats as a whole would be false. Nevertheless, where they occur, mangrove sponges form distinctive, high-biomass communities with associated fish and invertebrate faunas (Kathiresan and Bingham, 2001). For example, at Punta del Este, Cuba, Alcolado (1991) estimated 50–80 individual sponges per meter of shoreline, while at study sites in the Florida Keys, U.S.A., Engel and Pawlik (2005) counted 1195 sponges comprising ten species that occupied 73.5% of available mangrove root space.

The great majority of published information on mangrove sponge communities comes from the Caribbean rather than the Indo-Pacific (Barnes and Bell, 2002). There are probably several reasons for this disparity, related both to differences in the mangrove habitats and the sponge assemblages of the two regions. In the lower intertidal zone, Caribbean mangroves are dominated by *Rhizophora mangle*, which has long prop-roots that extend into deeper water and support a great diversity of epibiotic algae and invertebrates below the LLW line (Farnsworth and Ellison, 1996), while in most Indo-Pacific mangroves the equivalent zone is dominated by *Avicennia* and *Sonneratia* spp. which lack prop-roots (Nybakken, 1997). Unlike the large sponges found in Caribbean mangroves, Barnes (1999) found 91.9% of mangrove sponges from Mozambique were encrusting and the remainder were buried. The taxonomy of Caribbean sponges is much better described than sponges of the Indo-Pacific, and this has limited ecological studies of the latter. Caribbean sponge communities are remarkably similar over the breadth of the biogeographic region (see below), while those in the Indo-Pacific are more diverse and variable from location to location (van Soest, 1994).

There tends to be lower species diversity of sponges in mangroves than adjacent subtidal habitats (seagrass beds, coral reefs, hard-bottom, etc.) in both the Caribbean and Indo-Pacific (Barnes, 1999; Barnes and Bell, 2002). Numbers of sponge species can be high, ranging from 3 to 147 for Caribbean mangroves (Diaz et al., 2004), although these communities are

usually made up of less than ten dominant species on the basis of biomass, and generally the same dominant species are found throughout the biogeographic area. Among the most common Caribbean species are *Tedania ignis*, *Lissodendoryx isodictyalis*, *Chondrilla nucula*, *Geodia gibberosa*, *Halichondria melanodocia*, *Haliclona manglaris*, *Dysidea etheria*, *Hyrtios proteus*, *Mycale microsigmatosa* and *Spongia tubulifera* (cf. Sutherland, 1980; Alcolado, 1991; Engel and Pawlik, 2005; Diaz et al., 2004). Too few studies exist to generate a similar list for Indo-Pacific mangrove sponges, but it is interesting that some of the same genera were represented in a survey of the Quirimba Archipelago of Mozambique (Barnes, 1999), where *Tedania digitata*, *Haliclona* sp. and *Biemna* sp. were found in mangrove habitats.

2.2. Influence of abiotic factors on sponge communities

As primarily nearshore, estuarine habitats, mangroves are strongly influenced by abiotic factors such as freshwater runoff, sedimentation, and rapid temperature fluctuations from the influence of sun and wind on tidally driven shallow water. After many years of studying communities around Kingston Harbour, Jamaica, Goodbody (1961) concluded that mangrove root communities seldom reach a climax condition because of mortality after intense rain events. Studies of mangrove sponge communities in the Florida Keys, U.S.A., have also documented yearly cycles of mortality associated with physical disturbance, temperature extremes and rain events (Bingham and Young, 1995; Pawlik et al., 2007). Quite the opposite was found for mangrove sponge communities at Bahía de Búche, Venezuela, which, despite having the same community structure as other Caribbean sites, was remarkably stable over time (Sutherland, 1980). This unusual situation can be attributed to a combination of little or no rainfall runoff or storm events at this site as well as constant annual temperatures (Sutherland, 1980).

The cline in abiotic effects as a function of proximity to the shore has best been demonstrated by Ellison and Farnsworth (1992), who documented the prop-root faunal assemblages at six sites in Belize, ranging from nearshore to offshore sites. Species richness of all epibionts increased with distance offshore, with only two sponge species present in the most nearshore site increasing to nine in the most offshore site. In a subsequent study of four mangrove islands in Belize, Farnsworth and Ellison (1996) found that sponge diversity and abundance was greatest on the leeward rather than the windward side of islands, which they attributed to a combination of abiotic (physical disturbance) and biotic (larval supply) factors acting at different spatial and temporal scales. Compared with sponges growing in other tropical subtidal habitats (i.e., seagrass beds and coral reefs), species that comprise the typical Caribbean mangrove sponge community are specifically adapted to survive extremes in salinity, temperature and sedimentation, either through tolerance or rapid recovery after catastrophic loss (Engel and Pawlik, 2005; Pawlik et al., 2007). Adaptations to abiotic extremes do not preclude mangrove sponges from living in habitats where

physical conditions are better, such as reef habitats, but biotic factors, particularly predation, limit their distribution in those habitats (Pawlik, 1997; and see below).

2.3. Influence of biotic factors on sponge communities

While abiotic factors control the large-scale distribution of sponge assemblages in mangrove habitats, biotic factors may have important effects at smaller scales. Seastars of the genus *Echinaster* may be locally abundant in some Caribbean mangrove habitats, where they consume sponges on prop-roots that become accessible to them when they grow into the subtidal sediment (Waddell and Pawlik, 2000). Parrotfishes and angelfishes make excursions from reef habitats to some mangrove sites to feed on sponges (Dunlap and Pawlik, 1998). When the most common mangrove sponge species were transplanted to reef sites, they were quickly consumed by angelfishes, yet many of these same mangrove species can be found in interstices in the reef framework where predatory fishes cannot eat them (Dunlap and Pawlik, 1996; Pawlik, 1998). Therefore, mangrove habitats serve as a refuge from fish predation for sponges that are able to survive the abiotic conditions found there.

Competition for available stilt-root space within Caribbean mangrove sponge communities appears to be intense, with a clear dominance-hierarchy based on growth rate and the production of putative allelochemical agents (Engel and Pawlik, 2005). Interestingly, some mangrove sponges appear to use chemical cues to foster the growth of other sponge species on their surfaces, with the overgrowing species providing an anti-predatory chemical defence to the undefended species under them (Engel and Pawlik, 2000, 2005; Wilcox et al., 2002). While allelochemicals may be important in sponge-sponge interactions, Bingham and Young (1991) could find no allelochemical effect of existing mangrove sponges on other epifaunal invertebrate species in settlement experiments. Wulff (2005) recently suggested that the competitive superiority of mangrove sponges prevented the colonisation of mangrove sponge habitats by sponge species usually found in reef habitats. This conclusion, that biotic factors may be more important than abiotic factors in affecting mangrove sponge ecology, was based on transplantation experiments conducted in offshore mangrove habitats in Belize (Wulff, 2005), where Ellison and Farnsworth (1992) had previously found abiotic conditions were least stressful, and species richness was highest. In subsequent experiments at three coastal mangrove sites in the Florida Keys, U.S.A., and one offshore site on Grand Bahama island, Bahamas, Pawlik et al. (2007) observed that reef sponges declined in health and died within 60 days of being transplanted to mangrove sites, a result that was attributed to abiotic conditions of high temperature, rainfall events and sedimentation in mangrove habitats. While some mangrove sponge communities have been documented to be less influenced by abiotic factors, particularly freshwater runoff (Sutherland, 1980; Wulff, 2005), these are the exception rather than the rule (Goodbody, 1961; Ellison and Farnsworth, 1992; Bingham and Young, 1995; Farnsworth and

Ellison, 1996; Kathiresan and Bingham, 2001; Pawlik et al., 2007).

In addition to mutualisms between sponge species in mangrove habitats (Wilcox et al., 2002), sponges also form mutualisms with the mangrove plants themselves. Ellison and Farnsworth (1990, 1992) reported that epifaunal sponges and ascidians reduce damage to prop-roots of *R. mangle* by wood-boring isopods: roots without epifaunal cover exhibited damage and 55% lower growth relative to roots with epibiont cover. In addition, Ellison et al. (1996) discovered that transplantation of sponges onto prop-roots induced, within 4 weeks, the formation of fine rootlets that pervade sponge tissue.

Aside from the hard substratum provided by prop-roots, mangroves may also offer an enhanced food source for sponges. In general, sponges feed primarily on particles the size of bacteria. The rich microbial community that results from the productivity and nutrient cycling in mangroves (Kathiresan and Bingham, 2001) may promote faster sponge growth than in adjacent oligotrophic habitats, such as coral reefs.

3. Mangroves as habitats for meiofauna

3.1. Diversity and distribution of meiofauna

In mangroves a distinct 'phytal' meiofauna (generally defined as animals passing through a 1.0 or 0.5 mm sieve but retained on a 63- μ m-mesh), often dominated by acari, inhabits hard substrata such as prop-roots and pneumatophores (Proches et al., 2001; Bartsch, 2003). The focus of the majority of studies of meiofauna in mangroves, however, is on communities living in sediment or on decomposing leaves. Within mangrove sediments, as in most estuarine habitats, meiofauna are the numerically dominant metazoans. For practical purposes meiofauna may be split into hard-bodied and soft-bodied components (Somerfield et al., 2005). The former are organisms that preserve well in formalin and so can be identified in preserved samples. They are easier to study than the soft-bodied forms for which specialised preservation and examination techniques are required. Nematodes and harpacticoid copepods usually constitute over 90% of the hard-bodied component of the meiofauna and are the subject of most studies. Soft-bodied forms, such as Turbellaria, are generally ignored although they may be equally abundant (Alongi, 1987b).

Although a number of studies of meiofauna in mangrove habitats document the abundances of organisms identified to high taxonomic levels, such as phyla or classes, there are relatively few studies which provide information on the species composition and structure of meiofaunal communities. Species-level information for copepods is available from a geographically widespread set of mangrove systems with very different environmental characteristics, including a lagoonal system in Brazil (Gerlach, 1958; Por, 1984; Por et al., 1984), fringe mangrove forests in Florida (Hopper et al., 1973; Fell et al., 1975), hard-bottom mangroves in Sinai (Por, 1977; Reice et al., 1984), and soft-sediment tropical mangroves in India (Kondalarao, 1984; Kondalarao and Raman-Murty, 1988) and

southern Malaysia (Sasekumar, 1994; Gee and Somerfield, 1997; Somerfield et al., 1998). The heterogeneity of systems studied makes it difficult to draw general conclusions about benthic copepods in mangroves. For example, reports of the number of species found in mangroves vary widely. Por et al. (1984) recorded 14 species from the mangrove system of Cananea in Brazil. Kondalarao (1984) and Kondalarao and Raman-Murty (1988) found 32 and 22 species of harpacticoid copepods, respectively, from 2 estuarine mangrove systems in India. Sasekumar (1994) found 25 species in mangrove systems in Selangor, Southwest Malaysia. Between 60 and 70 putative species of Copepoda were recorded from the Merbok estuary in Northwest Malaysia (Somerfield et al., 1998). One reason for the relatively high number of species recorded in the latter study was that, as well as documenting sediment-dwelling species, it examined leaf-litter assemblages in detail. Particularly, a high number of species of the family Darcythompsoniidae, especially species of *Leptocaris*, were recorded. This family of copepods were only found on leaf litter, and appear to be adapted to feeding directly on litter, or for grazing epiflora off flat surfaces (Somerfield et al., 1998). Although there is little general evidence for a specialised mangrove copepod fauna, Por (1984) speculated that amongst the Darcythompsoniidae some specialised species might have evolved. The findings of Somerfield et al. (1998) support that speculation.

Species-level information on nematodes also comes from a heterogeneous set of mangrove environments which fall into two main groups: (1) seasonally arid, subtropical or temperate mangroves in Australia (Decraemer and Coomans, 1978; Hodda and Nicholas, 1986; Nicholas et al., 1991; Gwyther, 2003) and Brazil (Netto and Gallucci, 2003), and (2) tropical mangroves in Australia (Alongi, 1987a,b, 1990), India (Krishnamurthy et al., 1984), Malaysia (Gee and Somerfield, 1997; Somerfield et al., 1998) and eastern Africa (Ólafsson, 1995; Ólafsson et al., 2000). Neither Ólafsson (1995) nor Somerfield et al. (1998) found any evidence for a specialised nematode assemblage in mangrove sediments, as most of the genera found by them are typical of intertidal sediments worldwide. Estimates of the number of species inhabiting mangrove sediments vary widely. The number of nematode taxa found in the Merbok mangrove forest in Malaysia (107 putative species: Somerfield et al., 1998) compares well with numbers of nematode taxa from mangroves in Zanzibar (94 genera: Ólafsson, 1995) and Brazil (94 putative species, 86 genera: Netto and Gallucci, 2003), but is more than three times as many as were recorded from *Rhizophora* mangrove forests in Selangor, Malaysia (29 species: Sasekumar, 1994) or southern Australia (21 putative species: Gwyther, 2003). There are no nematode taxa which have been found exclusively on mangrove litter, but species in the genera most abundant on litter, such as *Diplolaimeloides*, *Diplolaimella*, *Atrochromadora* and *Theristus*, have been recorded in association with mangrove detritus from as far afield as Australia (Alongi, 1990; Nicholas et al., 1991), Southeast Asia (Gee and Somerfield, 1997; Somerfield et al., 1998; Zhou, 2001), Africa (Ólafsson, 1995) and Florida (Hopper et al., 1973).

3.2. Influence of abiotic and biotic factors on meiofauna distribution

Studies of meiofaunal distributions within mangrove sediments tend to highlight the importance of generalised intertidal and estuarine environmental gradients known to influence meiofaunal distributions everywhere, such as tidal height, salinity, oxygen availability, and sediment properties such as organic content and granulometry (Somerfield et al., 1998; Coull, 1999). Meiofaunal distributions are patchy, and vary seasonally (Alongi and Christoffersen, 1992), leading to widely varying estimates of abundance. The majority of individuals are concentrated near to the sediment surface, especially in muds (Somerfield et al., 1998), although in relatively oxic sandier sediments more specimens are found below the sediment surface (Vanhove et al., 1992). Siphonolaimid nematodes from genera such as *Parastomena*, which contain chemosynthetic symbionts and are typically found in anoxic environments rich in methane, are known to occur in mangrove sediments (Somerfield et al., 1998; Kito and Aryuthaka, 2006). Ansari et al. (1993), working on an Indian mangrove mudflat, showed that meiofauna depth distributions correlated with vertical gradients in redox potential, but also with interstitial water content, organic matter content, adenosine triphosphate concentrations, phytobenthic abundance and bacterial counts.

The fact that different mangrove plants have different environmental niches, and affect their surroundings differently, makes it difficult to assess the extent to which tree diversity influences meiofaunal diversity. Gee and Somerfield (1997) found no strong differences in meiofaunal communities between sediments underlying stands of *Rhizophora apiculata* and *Bruguiera parviflora* in otherwise similar areas of the Merbok mangrove forest, Malaysia. Such differences as were detected were attributed to differences in tree root structure, the degree of disturbance by macrofauna, and the physical and chemical nature of the leaves falling to the sediment surface. It appears that the presence or absence of mangrove vegetation is relatively unimportant in determining the composition of meiofaunal communities.

It has been suggested that high levels of sediment-associated tannins in mangrove sediments reduce the abundance of meiofauna (Alongi, 1987c) but mangrove leaves that fall onto the sediment are rapidly colonised, and there is a succession of species associated with their decomposition (Schrijvers et al., 1995; Gee and Somerfield, 1997; Somerfield et al., 1998; Zhou, 2001). Such observations support the view that meiofauna play an important role in litter degradation. Another important ecological role of meiofaunal organisms in mangroves is that they are an important component of the diet of commercially important fish (Coull et al., 1995) and crustaceans (Dittel et al., 1997). Despite their abundance and ubiquity, however, detailed knowledge of the taxonomy, biology and interactions of these animals, and their role in the functioning of mangrove ecosystems, is lacking.

4. Mangroves as habitats for macrofauna

Mangroves are inhabited by a variety of benthic invertebrates, such as brachyuran crabs, gastropods, bivalves, hermit crabs, barnacles, sponges, tunicates, polychaetes and sipunculids. Mangrove invertebrates often show marked zonation patterns, and colonise a variety of specific micro-environments. While some species dwell on the sediment surface or reside in burrows, others live on pneumatophores and lower tree trunks or prop-roots, burrow in decaying wood, or can even be found in the tree canopies (Sasekumar, 1974; Ashton, 1999). The burrowing activities of certain benthic invertebrates have a pronounced effect on sediment properties and biochemical processes, by enhancing the porosity and water flow through the sediment, assisting in flushing toxic substances. In addition, their feeding on the sediment surface (deposit feeding) and plant matter (detritivory) promotes nutrient recycling (see also Kristensen et al., 2008). In turn, benthic invertebrates are a source of food for vertebrate predators including shallow-water fishes that enter the mangroves at high tide (Sheaves and Molony, 2000).

Macrobenthos may be operationally separated in two groups, i.e., epifauna and infauna. Epifauna refers to those invertebrates that live on various substrates such as lower tree trunks and the sediment surface, but which do not burrow in it. A range of gastropods, crabs, and bivalve species are typical representatives of epifauna. Infauna refers to burrowing invertebrates which live within the sediment, and includes crabs, pistol prawns, polychaetes, and sipunculids. The distinction between infauna and epifauna is not always straightforward, however, and not always related to the organisms' functional role. For example, while many sesarmid crabs create extensive burrow systems, others appear to find refuge in crevices from decaying wood or root structures, or their burrowing status is unknown (see Gillikin and Kamanu, 2005).

4.1. Diversity and distribution of macrofauna

Macrofaunal communities in high and low intertidal mangroves are often distinctly different, and this relates in part to highly different environmental conditions. They appear to be influenced by hydroperiod, availability of organic matter and sediment characteristics (Lee, 2008). Lower intertidal mangrove sediments (typically silt- or clay-dominated) provide substratum for growth of benthic microalgae and macroalgae (Dor and Levy, 1984; King and Wheeler, 1985; Tanaka and Chihara, 1988; Aikanathan and Sasekumar, 1994; Sarpedonti and Sasekumar, 1996). In high intertidal mangroves, the substratum is often more sandy, and the reduced frequency of tidal inundation results in a drier, more saline environment where more leaf litter accumulates and which is less suitable for growth of micro- and macroalgae. Frequent inundation in the low intertidal zone also favours the presence of filter feeders and deposit feeders, whereas fauna in the high intertidal zone does not have frequent direct access to such food sources and other trophic groups therefore predominate there.

Gastropods are typically one of the dominant and most conspicuous macrofauna in mangrove systems, and occupy a wide range of ecological niches (Cantera et al., 1983; Plaziat, 1984). The distribution of gastropod species within a mangrove forest is influenced by a variety of factors such as light (as a major factor determining algal growth and as a factor influencing humidity), tidal exposure, salinity, and substrate type. The trophic position of gastropods is equally varied (see also Section 9): sediment dwellers feed – selectively or not – on sediment organic matter and/or microphytobenthos, *Littoraria* spp. feed on epibenthic crusts on stems and roots, and some species have been reported to feed on mangrove litter and/or propagules (such as *Melampus coffeus* and adult *Terebralia palustris*). Predatory and scavenging species such as *Thais* spp. and *Nassarius* spp. are much less abundant. Gastropods can attain very high species diversity in some mangrove ecosystems: Camilleri (1992) mentions 39 species of gastropods in an Australian mangrove, Jiang and Li (1995) found 28 species in a Chinese mangrove, and Wells (1990) reports 23 mollusc species from a mangrove forest in Hong Kong. On the other hand, species diversity differs strongly in different parts of the world, e.g., *M. coffeus* is the only gastropod present in the mangroves of Guadeloupe (Plaziat, 1984). The numerical abundance and biomass of molluscs can be equally impressive (e.g., Sasekumar, 1974), and they can even reach higher densities and biomass than brachyuran crabs in some cases (e.g., Wells, 1984), although the number of comparative studies is limited. A number of gastropod genera (e.g., *Ellobium*, *Enigmonia*) and species (e.g., *Littoraria scabra*, *T. palustris*) appear to occur exclusively in mangrove systems (Plaziat, 1984). The global pattern in species richness of mangrove gastropods closely follows that of mangrove trees (Ellison et al., 1999).

Bivalves are often considered to be confined to a narrow seaward zone, due to feeding and larval settlement restrictions (Plaziat, 1984). In Southeast Asia, however, *Polymesoda erosa* is adapted for a semi-terrestrial existence by living on the high shore where only occasional high tides inundate the habitat (Morton, 1976). A number of bivalves with chemo-symbiotic associations have also been reported from mangroves (e.g., Leбата and Primavera, 2001). Wood-boring bivalves are also common in the mangrove forest, and Singh and Sasekumar (1994), for example, reported 10 species of teredinids and 1 pholadid in several mangroves along the west coast of Peninsular Malaysia. These wood-boring bivalves are ecologically significant as they stimulate the decomposition of wood and live in symbiosis with nitrogen-fixing bacteria (Waterbury et al., 1983). It has been suggested that the latter process may represent a very significant yet overlooked source of nitrogen fixation in mangrove ecosystems in view of the abundance of dead wood and Teredinidae (Boto and Robertson, 1990). Although mangrove-associated bivalves are only rarely studied, their diversity can be surprisingly high: Alvarez-Leon (1983) reported 29 species of bivalves from the mangrove root systems on the Atlantic coast of Colombia, and Jiang and Li (1995) mention 24 bivalve species from a mangrove system in Hong Kong.

Together with molluscs, brachyuran crabs are the dominant macrofauna in most intertidal mangrove ecosystems. Early reports on the species diversity of mangrove-associated crabs in the Indo-Pacific (Jones, 1984) now appear to be outdated (see Lee, 1998), and as taxonomical difficulties are still a major restriction, the diversity and distribution of mangrove-associated crabs is likely to be far from understood. Ocypodid crabs (*Uca* spp. and *Macrophthalmus* spp., or *Ucides cordatus* in Central and South American mangroves) and grapsid crabs (Sesarminae, *Metopograpsus* spp., *Metaplex* spp.) usually dominate the crab fauna and species often exhibit marked horizontal and vertical zonation patterns (e.g., Frith et al., 1979; Jones, 1984; Frusher et al., 1994; Sivasothi, 2000). Whether these distribution patterns are related to physico-chemical characteristics of the environment (e.g., Frusher et al., 1994), or to the presence of specific tree species or tree diversity, remains to be determined (see Lee, 1997; Dahdouh-Guebas et al., 2002). Similar to what is observed for gastropods (Ellison et al., 1999), species richness of sesarmid crabs appears to follow global patterns in mangrove tree species richness (Lee, 1998), although the number of detailed surveys is relatively limited and taxonomical problems still exist. Sesarmids are most diverse in Southeast Asia and decrease to low numbers in Central America. Only five species of Grapsidae have been found in the mangroves of Florida and Central America (Abele, 1992). However, Alvarez-Leon (1983) recorded an impressive array of Grapsidae (16 species) on the Caribbean coast of Colombia.

Other relatively well represented groups of macrofauna such as polychaetes and hermit crabs have been much less frequently studied, and little is known on their overall diversity, abundance and functional role in mangroves. Worms can attain a high diversity in the soft, unconsolidated substrates on the seaward sides of mangroves, with polychaetes predominating in diversity as well as abundance (Metcalfe and Glasby, in press).

4.2. Functional role of macrobenthos

The mangrove macrobenthos is intimately associated with the bottom substratum. Crabs and gastropods ingest sediment and food such as bacteria, microalgae, meiofauna and detritus adhering to it, they burrow in it and move through it, and modify it in many physical and chemical ways (e.g., Warren and Underwood, 1986; Smith et al., 1991). Crab burrows provide an efficient mechanism for exchanging water between the anoxic substrate and the overlying tidal water (Ridd, 1996). This observation was confirmed by Stieglitz et al. (2000) who demonstrated that a burrow inhabited by a sesarmid crab and a pistol prawn was completely flushed within 1 h by the activities of the crustaceans during a single tidal event.

Crabs and gastropods are the two major seed predators in mangrove forests, and thus play an important role in determining plant community structure (Smith et al., 1989). An inverse relationship between the dominance of a given tree species in the canopy and the amount of seed predation was found for species of *Avicennia*, *Rhizophora* and *Bruguiera*. It is apparent there is a mutual relationship between sesarmid crabs

and mangroves, whereby mangroves provide a suitable habitat for the crabs, and the crabs reduces competition between mangrove plant species through selective predation on seedlings (Bosire et al., 2005). The selective effects of seed predation are not limited to sesarmid crabs, but can include land crabs and hermit crabs (Lindquist and Carroll, 2004). High seed predation by crabs can sometimes have a negative influence on regeneration of mangrove stands (Dahdouh-Guebas et al., 1997, 1998). Grapsid crabs dominate in Australia, Malaysia and Panama, while the gastropods *Cerithidea scalariformis* and *Melampus coffeus* are the most important seed predators in Florida mangroves.

Detritus-feeding invertebrates dominate the mangrove fauna. *Ucides cordatus*, a semi-terrestrial ocyropodid crab in Brazilian mangroves, feeds almost exclusively on plant material. Large male crabs consumed 3.3 g dry weight daily corresponding to 6% of their dry body weight (Nordhaus, 2004). Deposit feeders like *Uca* spp. scoop the surface layers of the sediment and derive nutrition from microalgae, bacteria and detritus. Some large sesarmid crabs are tree climbing and feed on fresh leaves (Sivasothi, 2000). Competition for mangrove litter has been observed in East African mangroves where many *Terebralia palustris* (potamidid gastropod) feed on the same leaf to prevent crabs from removing the leaf (Fratini et al., 2001).

The dominant role of grapsid crabs in the mangrove community structure and function has been investigated in Australia, Asia and East Africa (Giddens et al., 1986; Robertson and Daniel, 1989; Micheli, 1993; Lee, 1997; Ashton, 2002; Cannicci et al., 2008). The role of grapsid crabs as an agent affecting mangrove litter turnover in the Indo-Pacific is indisputable, but the exact trophic link remains unknown (Lee, 1997; see Section 9).

Numerous studies in Australia and East Africa indicate that grapsid crabs are major consumers of mangrove leaf litter and as a consequence produce large quantities of faecal material rich in nutrients and energy (Leh and Sasekumar, 1985; Micheli et al., 1991; Emmerson and McGwynne, 1992; Lee, 1997). These crabs also spend considerable time grazing and picking organic material off the surface of the substrate (Skov and Hartnoll, 2002), suggesting that they are using microbial resources for their nitrogen needs.

Fish predation on mangrove invertebrates occurs at high tide when the mangroves are inundated (Sasekumar et al., 1984; Wilson, 1989; Sheaves and Molony, 2000). For example, the mangrove crabs *Chironomantes* spp. and *Metapla* spp., and the sipuncula *Phascolosoma arcuatum* were found in the gut of fishes that were netted within the mangroves at high tide (Sasekumar et al., 1984). To what extent this form of feeding contributes to the food of shallow-water fish community and controls the structure of the mangrove benthic community awaits further studies.

In summary, benthic invertebrates in mangrove forests play an important ecological role by their activities of burrowing in the sediment where they assist in flushing toxic substances, and modifying the oxidation status of the surrounding sediment. Feeding on plant matter (detritivory) assists in recycling organic matter and produces animal biomass which is a source

of food for vertebrate predators (e.g., reptiles, birds, and otters) and inshore fishes that come in with the high tide.

5. Mangroves as habitats for prawns

Dall et al. (1990) classified penaeid prawn life cycles into four different types according to the environments in which the adults spawned and the postlarvae settled. Adults of two of these types both spawn offshore, and their larvae move inshore where the postlarvae settle in their preferred nursery grounds, either estuarine or nearshore habitats. The postlarvae develop into juveniles which spend between 6 and 20 weeks in the nursery ground before emigrating offshore (Haywood and Staples, 1993). The nursery grounds are generally dominated by some form of vegetation: either seagrasses (Coles and Lee Long, 1985), algae, mangroves (Staples et al., 1985), or saltmarshes (Webb and Kneib, 2002; Minello et al., 2003).

In one of the few comprehensive studies on epibenthic communities of mangroves, Daniel and Robertson (1990) found that, along with small fish, penaeids dominated the epibenthos. While a range of juvenile prawn species have been caught either within, or (more often) in the river or creek channels adjacent to, mangroves, only a few species have been found to be almost exclusively associated with mangroves as juveniles, e.g., *Penaeus¹ merguensis* (Staples et al., 1985), *P. indicus* (Rönnbäck et al., 2002; Kenyon et al., 2004) and *P. penicillatus* (Chong et al., 1990). Other species are less specific in their choice of nursery habitat, e.g., *Metapenaeus ensis* (Staples et al., 1985), *M. monoceros* (Rönnbäck et al., 2002), *M. brevicornis* and *M. affinis* (Chong et al., 1990). In addition to being associated with mangroves they are also found in other habitats such as mud flats and seagrass beds.

Prawns can only gain access to intertidal mangrove forests for 10–12 h each day, when they are inundated by the tide. When the tide recedes the prawns move out, and in the case of *Penaeus merguensis* and *P. indicus*, aggregate close to the water's edge (Staples, 1980; Kenyon et al., 2004). Because of the difficulty of sampling amongst the mangrove trunks, prop-roots and pneumatophores, most sampling for juvenile prawns (and other epibenthos) in mangrove habitats has been done using small beam trawls (Staples and Vance, 1979) or seines (Robertson, 1988) in the creek or river channels adjacent to the mangroves after the water has receded, or by setting trap nets on an ebbing tide across the small channels which drain the mangroves (Robertson, 1988; Robertson and Duke, 1990b). It is only relatively recently that researchers have examined the distribution of prawns and fishes within the mangrove forest using either drop samplers (Sheridan, 1992) or some form of stake- or lift-net to enclose an area of mangrove at high tide and collect the fish and epibenthos once the water has receded (Vance et al., 1996, 2002; Rönnbäck et al., 1999, 2002; Meager

¹ The subgenera of *Penaeus* were elevated to genera by Pérez-Farfante and Kensley (1997). However, as there is some controversy over this revision we have chosen to use the old names in this paper (Lavery et al., 2004; W. Dall, CSIRO Marine and Atmospheric Research, personal communication).

et al., 2003). Prawns seem to make extensive use of the mangrove forest during high tide; in northern Australia, Vance et al. (1996) sampled mangroves at the creek fringe and at sites up to 59 m inland from the creek banks. Juvenile *P. merguensis* appeared to distribute themselves throughout the forest as the tide rose, sometimes moving as far as 200 m into the mangroves (Vance et al., 2002). Similarly, working in the Philippines, Rönnbäck et al. (1999) found *P. indicus* and *Metapenaeus ensis* moved between 55 and 93 m from the mangrove fringe into the forest. In contrast, other studies have indicated that while *P. merguensis*, *P. indicus* and *M. bennettiae* move into the forest as the tide rises, densities are higher closer to the creek-mangrove interface (Rönnbäck et al., 2002; Vance et al., 2002; Meager et al., 2003). It is possible that these differences are due to variation in local topography between the study sites.

5.1. Influence of abiotic factors on prawn distribution

Many mangrove systems are located in estuaries and so are characterised by being subjected to a wide range of salinities. Juvenile prawns are euryhaline and so are able to cope with these fluctuations. Large postlarvae and juveniles of many species of penaeids are more tolerant of low salinities than early-stage postlarvae or adults (Dall, 1981). In fact, some authors have suggested that the postlarvae of certain species may use low salinity as a means of locating potential nursery grounds (Williams and Deubler, 1968; Hughes, 1969; Young and Carpenter, 1977), although high levels of rainfall (and presumably very low salinity) may inhibit postlarval immigration (Vance et al., 1998). Salinity fluctuations do not appear to significantly affect abundance, growth or mortality of penaeids (Staples, 1980; Haywood and Staples, 1993; Ahmad Adnan et al., 2002; Meager et al., 2003).

One of the main factors attributed to eliciting emigration of penaeids from mangroves out to sea is rainfall (Staples, 1980; Staples and Vance, 1986). Presumably the penaeids are responding to some other factor resulting from the increased rainfall, such as decreased salinity, increased current velocity, or changes in the levels of nutrients or other chemicals (Staples, 1980). Rainfall alone explained 70% of the observed variation in numbers of *Penaeus merguensis* emigrating from the Norman River in northern Queensland, Australia (Staples and Vance, 1986). The amount of rainfall also determined the size at which the prawns emigrated; in very wet years a wide size range of prawns emigrated, whereas in relatively dry years only the larger prawns emigrated (Staples, 1980; Staples and Vance, 1986). In contrast to these results, studies on the closely related species *P. indicus* in South Africa indicated that emigration was related to temperature, although only temperature and salinity were recorded in this study and not rainfall (Benfield et al., 1990). Garcia and Le Reste (1981) noted that the relative importance of rainfall and temperature in determining emigration varies geographically. They postulated that in areas with a small temperature range, but seasonal rainfall, rain is the most important stimulus, whereas in areas that have rainfall that is extended throughout the year, change in temperature is more important.

Temperature has been shown to be positively related to the growth rate and negatively related to the mortality rate of juvenile *Penaeus merguensis* in a tropical mangrove system in northern Australia (Haywood and Staples, 1993). Temperature has also been shown to influence penaeid catches in a temperate mangrove forest in Southeast Queensland, Australia. Meager et al. (2003) found that temperature was positively correlated there with catches of postlarval and juvenile *P. merguensis*, although a long-term (6 years) study in tropical Queensland indicated that temperature explained very little of the variation in numbers of postlarval or juvenile *P. merguensis* (Vance et al., 1998). Southeast Queensland is close to the southern limit of the distribution of *P. merguensis*, and it is likely that the relationship with temperature found in the Meager et al. (2003) was because temperatures here ranged from 14 to 30 °C compared to 23.5 to 31.9 °C in the study by Vance et al. (1998).

Several studies have found higher densities of juvenile prawns, other crustaceans, and fishes in mangroves compared to adjacent nearshore habitats, and the hypotheses offered to explain this can be grouped into three general categories (Robertson and Duke, 1987; Robertson and Blaber, 1992; Chong, 1995; Manson et al., 2005): (1) that mangrove forests are more productive than alternative inshore habitats and so provide more food, (2) that the structural complexity provided by mangrove trunks, roots and debris, high turbidity and soft sediment afford greater protection from predators (Robertson and Duke, 1987), and (3) that the mangrove forest acts as a larval-retention mechanism, preventing planktonic prawn larvae from being distributed further by local currents (Chong et al., 1996). The remainder of this section will discuss the existing evidence for these hypotheses.

5.2. Feeding

There appears to be little doubt that juvenile prawns are foraging while they are inside the mangrove forests. Wassenberg and Hill (1993) collected samples of juvenile *Penaeus merguensis* at various intervals during the tidal cycle and found that prawns had full guts as they left the mangrove forest on a receding tide, and inferred that the prawns had been feeding amongst the mangroves. An hour before low water, their guts were almost empty, indicating they had not been feeding after leaving the mangrove forest. Dietary studies have suggested that prawns eat a wide variety of food items including crustaceans, bivalves, gastropods, polychaetes, mangrove detritus, fish, insects, foraminiferans and diatoms (Chong and Sasekumar, 1981; Moriarty and Barclay, 1981; Leh and Sasekumar, 1984; Robertson, 1988; Wassenberg and Hill, 1993).

Because mangroves produce large amounts of detritus (Odum and Heald, 1975) it has been assumed that juvenile prawns and other mangrove-associated epibenthos have been able to utilise this production. However, although mangrove detritus has been identified in prawn guts (Chong and Sasekumar, 1981; Leh and Sasekumar, 1984; Robertson, 1988), cellulose is not digestible by penaeids (Omondi and Stark, 1996; Gonzalez-Pena et al., 2002) and so detritus,

although ingested by penaeids, may not be assimilated (see Section 9).

5.3. Protection from predation

Predation is generally considered to be one of the key causes of mortality in prawns (Minello and Zimmerman, 1983; Robertson, 1988; Salini et al., 1998) and several studies have indicated that predatory fish consume significant numbers of juvenile prawns (Davis, 1985; Robertson and Duke, 1990b; Salini et al., 1990). Mangrove systems provide shallow water, structural complexity and in many cases high turbidity and fine sediment, suitable for burrowing prawns. These factors, in combination with the behaviour of prawns, acts to protect prawns by reducing their visibility to, and lowering their encounter rate with potential predators (Minello and Zimmerman, 1983; Laprise and Blaber, 1992; Kenyon et al., 1995).

Large predatory fish tend not to move far into the mangroves at high tide, whereas juvenile *Penaeus merguensis* move long distances from the mangrove margin, making them less vulnerable to fish predation (Vance et al., 1996). The structural complexity provided by mangrove roots, trunks, detritus and fallen timber is thought to provide protection to juvenile prawns and other prey from fish predation (Robertson, 1988; Robertson and Blaber, 1992; Primavera, 1997; Meager et al., 2005). Protection from predation by mangrove structures is not as simple as might be expected; instead, it appears to be a complex interaction between predator species, prey behaviour and habitat complexity (Primavera, 1997; Meager et al., 2005). Primavera (1997) examined predation rates on two species of prawns, *P. merguensis* and *P. monodon*, in laboratory experiments conducted on bare sand, coconut leaf bracts (to simulate mangrove detritus) and a range of different densities of pneumatophores. Two different species of predator (the barramundi *Lates calcarifer* and the mangrove jack *Lutjanus argentimaculatus*) were tested. Overall, significantly more prawns were eaten on the bare sand (48%) and leaf bracts (44%) than among the pneumatophores (30%). The presence of the structure (leaf bracts and pneumatophores) did not appear to affect the hunting abilities of the barramundi which chased prawns underneath the leaf bracts and between the pneumatophores, whereas the mangrove jack restricted themselves primarily to the clear water, away from the structure. Predation rates on *P. monodon* were lower on medium density pneumatophores compared to those on bare sand whereas there was no significant difference between predation rates on *P. merguensis* on bare, medium or high-density pneumatophores. Primavera (1997) speculated that this may be due to behavioural differences between the two prawn species. The behaviour of prawns in relation to shelter may be modified by the presence of particular predators. Meager et al. (2005) noted that while juvenile *P. merguensis* selected habitats providing vertical structure (pneumatophores or mangrove debris) over those with no vertical structure (bare substrate or leaf litter), once *L. calcarifer* was introduced to the tank, the prawns selected the mangrove debris over the pneumatophores. However, the prawns did not react to the presence of an

alternative predator, the blue catfish *Arius graeffei*. Previous experiments had demonstrated that *L. calcarifer* was a more effective predator of juvenile *P. merguensis* than *A. graeffei* (Meager, 2003).

High turbidity is a characteristic of many, but not all mangrove forests and turbidity within the mangrove forests may change dramatically depending upon the strength of tidal currents. While high turbidity may decrease the effectiveness of some prawn predators, others, particularly those using chemosensory mechanisms for prey detection, may be more successful in capturing prawns in turbid water (Minello et al., 1987; Macia et al., 2003).

5.4. Larval retention mechanism

The mouths of many mangrove-lined creeks are shallow and are characterised by extensive intertidal mud flats. The resulting high friction between the water and the mudflats means that, rather than forming tidal jets, the ebbing water spreads out in a fan-like manner at the mouths of the creeks (Wolanski et al., 1992). During periods of little or no longshore current most of the water from the swamp is returned during the following flood tide. In this way, mangrove swamps trap the lateral movement of coastal water for periods of between 2 and 8 weeks (Wolanski and Ridd, 1986; Wolanski et al., 1990) and so planktonic prawn and other larvae may be retained within the mangrove forest rather than being dispersed by currents (Chong, 1995). Even during the wet season water can be retained for weeks in the mangroves and side creeks (Robertson and Blaber, 1992). Lateral trapping is more effective in the upper reaches of a mangrove estuary than at the mouth (Wolanski and Ridd, 1986). This phenomenon may be responsible for influencing the along-river distribution of postlarval and juvenile *Penaeus merguensis* observed by Vance et al. (1990). They found densities were almost five times higher in the upstream reaches of a small creek compared to those in the main river channel, near the mouth of the creek.

In summary, the fact that some species of juvenile prawns are found in relatively high densities within mangrove forests has been well documented, although the precise nature of the prawn-mangrove relationship is still unclear. There is a possibility that mangroves may be acting as a physical trap for water bodies containing the prawn larvae. While this seems plausible it does not explain why other species of prawns which have different habitat preferences (e.g., for seagrass) are not retained within mangroves as well. Based on the evidence available, mangroves seem to provide both food and protection from predation. Juvenile penaeids appear to be gaining their nutrients from mangrove-associated bacteria, epi- and infauna rather than directly from mangrove leaves or detritus.

6. Mangroves as habitats for insects

The global distribution of mangroves has been divided into two biogeographical hemispheres, the Indo-West Pacific and the Atlantic-East Pacific (Duke, 1992). The former ranges from the east coast of Africa to Asia, Australia and the western

Pacific islands, while the latter includes the eastern Pacific islands, the coasts of the American continent and the African west coast. Insect diversity in the mangroves of the Indo-West Pacific is thought to be higher than in the Atlantic-East Pacific as a result of higher plant diversity in the former although, to some extent, the dearth of insect species in the latter reflects gaps in our knowledge rather than low species diversity (Macintosh and Ashton, 2002). In general, the mangrove insect fauna of the Indo West Pacific has been better studied.

Mangroves provide a habitat that supports a large number of insects at different trophic levels. The primary trophic groups are (1) herbivorous insects that feed on leaves and other plant parts, (2) saproxylic and saprophagous insects that feed on dead and decaying organic matter, and (3) parasitic and predatory insects that feed or prey on other animals.

6.1. Herbivorous insects

Although the herbivorous insects of mangrove trees have sometimes been portrayed as being poor in diversity and lacking in specialists, more recent studies have shown that there is greater host specialisation and herbivory levels than previously thought (Burrows, 2003). In a study on herbivorous insects of mangroves in Singapore, Murphy (1990) found 102 herbivore species feeding on 21 species of mangrove plants from nine principle taxa. A wide range of mangrove plant parts were fed on, including leaves, shoots, flowers, fruits and stems. Stenophagy (feeding on narrow host range) appeared to be more developed among insects in *Avicennia* and *Sonneratia*, and was thought to be reflective of the relative taxonomic isolation of these plant taxa. In Queensland, Australia, little overlap was found between the insect herbivore fauna of *A. marina* and *Rhizophora stylosa*, and the diversity of herbivores on these tree species was similar to the diversity of nearby non-mangrove trees, and to the diversity levels reported for other tree species elsewhere in the tropics (Burrows, 2003). Mangrove habitat specificity has also been noted in a study on moths of the superfamily Geometroidea in Peninsular Malaysia, although species diversity in the mangrove habitat was low in comparison to that in other lowland forest types (Intachat et al., 2005). At the habitat level, however, mangroves can be expected to have lower herbivore diversity than other lowland forest types as a result of their lower plant diversity.

In butterflies, for which habitat and host associations are better known, few species are entirely restricted to mangroves (Corbet and Pendlebury, 1992). However, some butterfly species such as *Junonia* spp. (Nymphalidae) that occur in landward edge zones of mangroves and upper reaches of estuaries have host plants that are primary colonisers. These plants may be rare or absent under the closed multi-storey canopies of inland forests, but are able to persist in more open coastal habitats. The significance of these populations in natural coastal habitats is rarely recognised because many of the species are also able to colonise disturbed and man-made habitats such as parks and gardens. The few mangrove-inhabiting butterflies that occur across a range of natural habitats are oligophagous or polyphagous species such as

Hypolycaena erylus (Lycaenidae) and *Polyura schreiber* (Nymphalidae) that have been recorded feeding on one or more mangrove plant species (Murphy, 1990), but feed on other non-mangrove host plants as well (Corbet and Pendlebury, 1992). A notable example of a monophagous mangrove specialist butterfly is *Danaus affinis* (Nymphalidae). The larvae feed on the vine *Ischnostemma selangorica* (Asclepiadaceae) that grows in the landward edge zones of mangroves, and the adults feed on nectar only from the flowers of the coastal shrub *Wedelia biflora* (Kirton and Azmi, 1996, 2000, 2004).

Special adaptations to the mangrove environment occur in a number of taxa of herbivorous moths. Moths of the pyralid subfamily Nymphulinae have aquatic larvae that feed selectively on the algae *Dictyota* (Dictyotaceae) and *Murayella* (Rhodomelaceae) in the intertidal zones of the mangrove floor (Yoshiyasu, 1984; Murphy, 1989). *Eristena mangalis*, a moth of the family Pyralidae, has aquatic larvae that feed on brown algae. The larvae have breathing filaments on the body, and build tube-like shelters in drainage channels under rotting leaves (Ng and Sivasothi, 2002). Other moths, such as *Cleora injectaria* (Geometridae), *Aucha velans* and *A. velliana* (Noctuidae), feed on the leaves of mangrove trees but pupate in the intertidal zone under rotting vegetation or algal mats at the base of their host plants (Murphy, 1990). A leaf-tying caterpillar of the moth genus *Argyroplote* (Tortricidae) has also been reported feeding on the tidally inundated leaves of young *Sonneratia-griffithii* in Peninsular Malaysia, and was able to survive submersion in seawater for 4.5 h (Lever, 1952).

6.2. Saproxylic and saprophagous insects

Insects that feed on dead trees or wood (saproxylic insects) or decaying organic material (saprophagous insects) play an important role in nutrient cycling in forests. Termites and wood-borers (usually the larvae of beetles or moths) form the majority of saproxylic insects, and a relatively characteristic assemblage occurs in mangroves. The relative abundance of a limited number of tree species provides an abundant and stable food source for this group of insects.

In the intertidal zone, periodic or continuous flooding makes mangroves uninhabitable for many termite species that forage from the ground. However, species that nest above the ground thrive in this habitat in the absence of competing fauna and in the presence of abundant food resources. Among these are drywood termites such as *Glyptotermes*, *Neotermes* and *Cryptotermes*, which feed on dead branches that die in the canopy, or on dead standing trees (e.g., Miller and Paton, 1983; Salick and Tho, 1984). Their ability to feed on relatively dry wood is thought to be a result of an ability to obtain metabolic water through physiological processes in their bodies, as well as a result of structural adaptations that enable them to conserve water (Collins, 1969). The uniform, single-tiered canopy of mangrove forests results in a relatively low humidity in the canopy. Trees that die or branches that die in the canopy dry faster than in the multi-storied canopies of inland forests. Other termite groups that readily colonise the mangrove habitat are *Prorhinotermes* (Tho, 1992) and some species of *Coptotermes*

(Kirton, 1995), which are able to nest in moist wood with no ground contact, and species that build arboreal carton nests on tree trunks and branches, such as *Microcerotermes* spp. and some members of the subfamily Nasutitermitinae (L.G. Kirton, personal observations).

Scolytids and platypodids are the primary wood-boring beetles in mangroves. The majority feed on dead branches that have yet to dry completely. They burrow under the bark or into the wood and culture fungi on which their brood feeds. However, some are seed or prop-root feeders (Ng and Sivasothi, 2002), and others may cause the death of branches and trees through girdling and hollowing of stems and twigs. Their effects on some mangrove communities are known to exceed that of herbivores (Feller and Mathis, 1997; Feller, 2002).

A large number of ground-dwelling saprophagous insects can also be found in the mangrove habitat, and many have specialised adaptations for survival in the intertidal zone. Collembola (springtails) are diverse among the roots of mangrove plants and in the leaf litter that accumulates on the ground (Murphy, 1965; Roque, 2007), where they feed on a range of organic material including detritus and fungi. *Pseudanurida billitonensis* (Neauridae) is a springtail in the mangroves of Singapore that feeds at low tide and returns to underwater shelters during high tide (Ng and Sivasothi, 2002). A pyralid moth, *Hymenoptychis sordida*, is known to feed on fallen fruit of *Avicennia*, dead leaves, roots and rotting timber on the mangrove floor in Singapore (Ng and Sivasothi, 2002). The larvae shelter in silken tunnels when the tide rises.

6.3. Parasitic and predatory insects

A wide range of predatory and parasitic insects with a great diversity of host and habit occur in mangrove habitats. These include predatory larvae and adult insects that prey on other organisms, parasitoids that feed within a single host and eventually kill it, hyperparasitoids that parasitize parasitoids, and blood-sucking parasites of vertebrates. They occur throughout the mangroves, from the soil to the water surface and on mangrove plants, where they exert a restraining influence on populations of herbivorous and saprophagous organisms.

Ants (Formicidae) are important predators in mangroves. One species of weaver ant, *Oecophylla smaragdina*, that is common in mangroves in Australasia, nests by drawing the leaves of mangrove plants together with silk threads spun by their larvae. The ant has been shown to significantly reduce herbivory levels on the mangrove tree *Rhizophora mucronata* in Thailand (Offenberg et al., 2004a), and the presence of this ant's pheromones on leaves of *R. mucronata* have been shown to deter feeding by the leaf beetle, *Rhyarida wallacei* (Chrysomelidae) (Offenberg et al., 2004b).

The best-known predatory insects that characterise the mangrove habitat are, arguably, fireflies (Coleoptera: Lampyridae). Although fireflies occur in a range of habitats, some are associated with mangroves and mangrove estuaries, where their larvae prey on snails in the intertidal zones. These include species of *Pteroptix*, which range from South and Southeast

Asia to New Guinea (Ballantyne and McLean, 1970; Ballantyne, 1987), and are well known for their impressive group displays on certain trees (Buck and Buck, 1976; Nallakumar, 1999; Zaidi and Yong, 2004). In Southeast Asia, members of groups of *Pteroptix tener* or *P. malacca* flash in close synchrony (Buck and Buck, 1976; Hanson, 1978). The synchronous flashes have made sites with large congregations of fireflies ecotourism destinations in some countries such as Malaysia (Nallakumar, 1999). *Pteroptix tener* is the species for which the biology and ecology has been better studied (e.g., Kumar, 1979; Motuyang, 1994; Rahmat, 1996; Nallakumar, 2002; Nada and Kirton, 2004), particularly in Kuala Selangor, Malaysia, where the adults display and mate on riverside trees, especially the mangrove tree *Sonneratia caseolaris*. The larvae prey primarily on the snail *Cyclotropis carinata* (Assimineidae) in tidally inundated areas of the riverbanks. They are able to enter the shells of the snails and inject a paralysing toxin and enzyme into the soft body tissues of the snails to aid in feeding.

Many other predatory insects live and feed on the ground, sheltering under plant debris during high tides and emerging to feed on springtails, copepods, protozoa and nematodes when the tides recede (Ng and Sivasothi, 2002). Among the more common are heimpertans and pselaphid beetles. On the water surface of mangrove tidal pools, water skaters (*Xenobates* sp., Veliidae) prey on smaller insects that fall or land on the water (Ng and Sivasothi, 2002).

Female mosquitoes (Culicidae) and other small-biting flies (Ceratopogonidae, Simuliidae and Phlebotominae) that inhabit mangroves take a blood meal from vertebrate hosts prior to reproduction. Biting midges breed in the mud in mangroves and mosquitoes breed in stagnant pools as well as rot holes in trees. One species of mosquito in East Africa, *Aedes pembaensis*, is known to lay its eggs on the claws of the crab *Neosarmatium meinerti*, and the larvae develop in the burrow of the crab (see Hogarth, 1999).

7. Mangroves as habitats for elasmobranchs

The number of elasmobranch species recorded from fresh or estuarine waters is at least 171 species, representing 68 genera and 34 families, with the greatest diversity occurring along the rapidly developing tropical coastlines of South America, West Africa, and Southeast Asia. Of the 171 elasmobranch species, more than 50% occur in estuaries (Martin, 2005).

Estuaries are used by various species of euryhaline and obligate freshwater elasmobranchs, but their life history and ecology, including dependency on various habitats such as mangroves, is virtually unknown (Martin, 2005). Similarly, no information exists that compares the species diversity of rays versus sharks in mangrove habitats. Habitat selection by elasmobranchs is influenced by a multitude of interacting variables such as temperature, salinity, depth, substrate type, benthic vegetation, prey distribution and variability, predator distribution, social organisation, and reproductive activity (Simpfendorfer and Heupel, 2004). It is well known that many species of elasmobranchs rely on nearshore habitats as nursery grounds (Montoya and Thorson, 1982; Simpfendorfer and

Milward, 1993; White and Potter, 2004), with several species such as the nervous shark *Carcharhinus caudatus* spending its entire life cycle in the estuaries of northern Australia (Lyle, 1987). However, separating the contribution of mangroves from that of the non-mangrove estuaries can be problematic as the functional roles of mangroves may be the same as those provided more generically by 'estuarine' and 'nearshore' environments (Loneragan et al., 2005). For example, estuaries without any mangroves can function as productive nursery habitats and thus the respective roles of estuary and mangroves may not be clear (Ruiz et al., 1993).

An acoustic telemetry study to determine habitat selection of juvenile lemon sharks (*Negaprion brevirostris*) at Bimini in the Bahamas, found that *Rhizophora* mangrove thickets were routinely selected according to their availability. The main abiotic factors influencing habitat selection were temperature, substrate type and depth (Morrissey and Gruber, 1993). It is well known that mangroves support large populations of prey on which elasmobranchs feed and also provide a complex habitat structure as a means of avoidance from large sharks and other marine predators (Robertson and Duke, 1987; Simpfendorfer and Milward, 1993). These biotic factors were considered to be the most likely to explain the use of mangrove habitat by *N. brevirostris* at Bimini (Morrissey and Gruber, 1993). In the large subtropical marine embayment of Shark Bay, Western Australia, mangrove tree density was found to influence the number of species and catch rates of elasmobranchs and their prey (White and Potter, 2004).

Within the shallow, nearshore waters of Shark Bay, the arid zone mangroves are considered particularly important feeding habitats for juvenile giant shovelnose rays (*Rhinobatos typus*) at high tide (White and Potter, 2004). In a comparable environmental setting in the remote, arid Exmouth Gulf of Western Australia, juvenile and neonate *R. typus* occur at a density of 3 m⁻² within the fringing mangroves and salt flat habitats (H.M. Penrose, unpublished data). Due to fishing-related pressures, *R. typus* is listed on the IUCN Red List as 'vulnerable' to extinction on a global scale (IUCN, 2006). As with many other species of elasmobranch, clarification of the value of mangrove as habitat during this critical life history stage is fundamental for their conservation and management.

8. Mangroves as habitats for bony fishes

8.1. Diversity and distribution of fishes

The Indo-West Pacific region, stretching from the east coast of Africa through South and Southeast Asia to Australia and the Central Pacific, has the highest diversity of fishes in the world (at least 600 species in mangrove systems) (Blaber, 2000). The high diversity decreases latitudinally away from the equatorial core area (sensu Blaber, 2000) in Southeast Asia, but larger subtropical mangrove systems still contain at least 100 species. Many species occur throughout this region, while others are restricted to particular regions. Examples of species that occur in most mangrove areas of this region include the sly bream *Acanthopagrus berda*, the glassfish *Ambassis gymnocephalus*,

the trevally *Caranx sexfasciatus*, the wolf herring *Chirocentrus dorab*, the tenpounder *Elops machnata*, the pursemouth *Gerres filamentosus*, the ponyfish *Leiognathus equulus*, the mangrove jack *Lutjanus argentimaculatus*, the flathead *Platycephalus indicus*, the flounder *Pseudorhombus arsius*, the whiting *Sillago sihama* and the thornfish *Terapon jarbua*. A special group of fish species found in mangroves are the mudskippers (family Periophthalmidae) which occupy a specialised niche in the intertidal zone. They are physiologically and morphologically adapted to an amphibious existence in this zone with highly variable environmental conditions (Clayton, 1993), and they are able to dwell on exposed mudflats when other fish species are forced to retreat to deeper waters with outgoing tides (Kruitwagen et al., 2007).

Mangroves occur in four tropical zoogeographic regions of the world (Indo-West Pacific, East Pacific, West Atlantic and East Atlantic) and the fish communities of all have many common characteristics. In almost all cases they are dominated by fishes of marine origin, with more than half the number of species as well as the number of individuals being contributed by either fully estuarine species or marine migrants. The mangroves of the tropical East Atlantic region along the west-African coast have somewhat fewer species than the Indo-West Pacific, but are still relatively rich, with larger estuaries such as the Senegal having more than 130 species, and smaller systems such as the Fataha in Guinea about 100 species. The tropical West Atlantic region from the Gulf of Mexico to northern South America has similar numbers of species, with most systems in the equatorial region containing at least 100 species.

The dominant taxa in each region are broadly similar, but there are some interesting contrasts. In all regions, except the Indo-West Pacific, Sciaenidae are one of the dominant families. In the Indo-West Pacific, sciaenids are important in the equatorial regions of Southeast Asia, but much less so elsewhere. This pattern may be connected with the amount of rainfall and the degree to which coastal waters approach estuarine conditions.

There are important differences in the relative proportions of freshwater species, both between and within regions. Freshwater species make up more of the fish fauna in tropical Atlantic mangrove systems than in the Indo-West Pacific or East Pacific, particularly in South America where many of the very diverse fauna of silurid catfishes are common in estuaries (Barletta et al., 2005). Similarly, in West Africa various silurids and cichlids make a significant contribution to mangrove fish communities (Baran et al., 1999). In East Africa and Australia, however, freshwater species are usually insignificant components of the mangrove fish fauna (Blaber, 2000). The equatorial regions of Southeast Asia have somewhat more freshwater species than other areas of the Indo-West Pacific, but despite the diversity of the freshwater fish faunas of Borneo and Sumatra relatively few live in estuaries. Throughout the islands of the Caribbean most mangroves are non-estuarine and small in size (fringing), and typically contain marine species dominated by the families Gerreidae, Haemulidae, Lutjanidae and Scaridae (Rooker and Dennis, 1991; Acosta, 1997; Nagelkerken et al., 2000b).

One of the key influences of mangroves on fishes is the physical structure they provide. Their pneumatophores, prop-roots, trunks, fallen branches and leaves make a complex habitat for a host of potential prey organisms, and the rich epiflora of algae and diatoms found on the mangroves and associated substrata are an important food source for many fish species (Blaber, 2000; Verweij et al., 2006a). Because mangroves usually occur in shallow intertidal areas of deposition, with quiet waters, muddy substrata, variable turbidities and a rich fauna and flora, their effects on fish are inextricably linked with these factors. The whole suite of mangrove-associated biotic and abiotic conditions makes them one of the core fish habitats of tropical estuaries and lagoons (Blaber, 2007).

8.2. Influence of abiotic factors on fish communities

Almost all fishes living in subtropical and tropical mangroves are euryhaline and able to cope with salinities from almost freshwater (<1 ppt) to at least 35 ppt, but their ability to do so varies from species to species and hence may influence their distribution. Salinities down to about 25 ppt apparently pose few osmoregulatory problems for most tropical marine fishes. Salinities over vast areas of mangrove-fringed coastal waters in the tropics may decline in the wet season to as little as 20–25 ppt. In some areas, such as the Bay-of-Bengal-and-parts-of-the-South-China-Sea, salinities seldom rise above 30 ppt and the fish communities of such areas are highly diverse (Pauly, 1985). Salinity is not only relevant to the distribution patterns and survival of fishes in estuaries, but may also affect metabolic processes. Peterson et al. (1999) have demonstrated that there is the potential for fluctuations and spatial variability in salinity to cause significant variability in the short-term growth rates of at least one species in nursery areas. Long-term salinity variations may also affect the distribution and occurrence of fish species in estuaries. In various studies, catch rates of the most abundant species were most strongly correlated with long-term (months and years) salinity patterns (Sheaves, 1998; Barletta et al., 2005; Lugendo et al., 2007a). In the coastal lake Nhlange, part of the Kosi system of northern KwaZulu-Natal, South Africa, the changes in the fish fauna in response to gradual changes in salinity were a reduction in the diversity of marine species and an increase in numbers of individuals of a few freshwater species (Blaber and Cyrus, 1981). Nevertheless, even at salinities of 1 ppt or less the system still contained 23 estuarine or marine species.

Turbidity is one of the major factors influencing the distribution of juvenile fishes in subtropical and tropical mangrove systems. In a study of the fishes of Moreton Bay, a large estuarine embayment on the Queensland coast, Blaber and Blaber (1980) showed that variations in turbidity correlated with different distribution patterns of fish which could be divided into three categories: those tolerant of turbidity, those indifferent to turbidity (comparatively few), and those intolerant of turbidity. Most fishes in mangroves are tolerant of high turbidities.

8.3. Factors determining the attractiveness of mangroves for fishes

Three hypotheses, namely (1) reduced predation, (2) increased food supply, and (3) increased living space or shelter, have been suggested to explain the attractiveness of usually turbid mangrove areas, as juvenile-fish nurseries (Blaber, 2000). Predation on juvenile fishes in mangroves may be less, because turbid waters reduce the effectiveness of large visual fish predators (Blaber and Blaber, 1980; Cyrus and Blaber, 1987a), shallow waters exclude large fishes (Shulman, 1985), and structure such as seagrass or mangroves enables small fishes to hide from predators (Laegdsgaard and Johnson, 2001). Evidence supporting the turbidity hypothesis comes from comparisons of fish densities across a variety of coastal habitats which show that the abundances of certain species respond positively to increases in turbidity. Further evidence comes from observations of greater abundances of piscivorous species in the creeks of the Dampier mangroves of Northwest Australia that receive no run-off from the land, and thus have much clearer waters (Blaber et al., 1985). Evidence supporting this hypothesis also comes from estuarine systems in which there is little or no mangrove habitat (e.g., the St. Lucia system in South Africa; Cyrus and Blaber, 1987b). However, if fish are responding solely to turbidity, it is possible that mangrove vegetation has little effect on the dependence of fish on estuaries.

Both the quantity and types of food may differ between mangrove areas and adjacent waters (Nagelkerken et al., 2000b). Many of the foods available in sheltered mangrove waters are rare or absent in offshore waters, particularly detritus and microfauna and flora, as well as aquatic macrophytes and their epifauna and flora. The presence of mangroves in tropical estuaries increases the diversity and quantity of food available to juvenile fishes (Blaber, 1980, 1987), and Robertson and Duke (1987, 1990a) have shown highly significant differences in the densities of juvenile fishes between mangrove and other nearshore habitats when they are immediately adjacent to each other. Furthermore, densities of zooplankton in mangrove habitats are greater (by an order of magnitude) during the late dry- to mid-wet season recruitment period of fishes, than in the middle of the dry season. Most newly recruited fish in estuarine or mangrove habitats are zooplanktivores. In intertidal mangroves, fishes typically enter the mangroves at high tide to feed (Vance et al., 1996; Sheaves and Molony, 2000; Lugendo et al., 2006). On Caribbean islands where mangroves are often permanently inundated, on the other hand, fishes mainly shelter in the mangroves during daytime and feed on adjacent seagrass beds at night (Nagelkerken et al., 2000a; Nagelkerken and van der Velde, 2004a). Nevertheless, opportunistic feeding takes place while they shelter in mangroves during daytime (Nagelkerken and van der Velde, 2004b; Verweij et al., 2006b).

The structural significance of mangroves for fishes is well demonstrated by the studies of Thayer et al. (1987) in Florida, Blaber and Milton (1990) in the Solomon Island, Vance et al. (1996) in Australia, and Rönnbäck et al. (1999) in the

Philippines. They showed that the prop-root habitat of mangroves is of major importance to a wide variety of fishes and that the species composition of fishes varies according to the species of mangrove tree, whether the channels are blocked or choked by fallen mangrove tree branches, and the type of substratum. Cocheret de la Morinière et al. (2004) showed in an experimental setup that different densities of prop-root mimics and different degrees of shading (mimicking the above-water mangrove canopy) affected the abundance of fish sheltering in them. It appears that mangrove-associated species inhabit mangrove prop-roots in higher densities than reef-associated species, because their juveniles occur more commonly in shallow-water mangrove areas and not because they possess a greater competitive advantage for mangroves compared the reef-associated species (Nagelkerken and Faunce, 2007). The importance of presence of mangrove-root structure is also shown by the positive effects on abundance and presence of fauna (e.g., crabs, fish, infauna) as a result of mangrove replanting (Bosire et al., 2004; Crona and Rönnbäck, 2005; Walton et al., 2007; Bosire et al., 2008).

The effects of differing structure are illustrated by reference to the physical characteristics of Solomon Islands estuaries, which are similar to those of other tropical Indo-West Pacific estuaries in terms of their salinity, temperature and turbidity regimes (Blaber, 1980; Blaber et al., 1989). There are two different types of mangrove estuaries in the Solomon Islands, based on substrata and mangrove tree species, each with a different fish fauna. Hard substratum estuaries with an abundance of mangrove tree debris are inhabited mainly by species that apparently need the cover or structure provided by the debris, such as Pomacentridae and some species of Apogonidae, together with juvenile Lutjanidae and Serranidae. These species are largely absent from the soft substratum estuaries that have little debris cover, where Gobiidae, including burrowing species are dominant.

The complexity of the interplay among various factors on the distribution of fishes in mangroves is exemplified by the study of Rönnbäck et al. (1999) who showed that the numbers and biomass of fish species were higher among the pneumatophores than in the prop-root areas in Philippine mangroves. This was thought to be because the limited height of pneumatophores allows a larger volume of water free of roots where fish can swim without encountering structural complexity. Also food items associated with mangroves are much more concentrated among pneumatophores compared to among the relatively less dense prop-roots, thereby facilitating feeding (Rönnbäck et al., 1999). For both root types, the densities of small fishes were higher and that of larger carnivores lower than in more seaward habitats. Verweij et al. (2006a) showed that some species mainly utilised prop-roots for shelter whereas other species mainly used them for feeding.

In summary, the three hypotheses of predator avoidance, food and shelter, are probably all important. However, their relative significance, in terms of dependency and utilisation of tropical and subtropical mangroves by fishes, will vary

depending upon the fishes and the nature of each system and its mangroves (Pittman et al., 2004; Sheridan and Hays, 2003; Lugendo et al., 2006).

9. The role of litter in the mangrove food web

9.1. Importance of mangrove litter

The idea that mangroves provide a trophic link with the abundant faunal communities within the mangrove ecosystem and in adjacent habitats has been a longstanding issue in the literature and was first proposed by the classical work of Odum and Heald (1972) in their 'outwelling hypothesis'. Odum and Heald suggested that the high productivity of mangroves is partially exported to the aquatic environment, providing an important food source for secondary consumers and thereby supporting adjacent fisheries. It has become increasingly apparent over the past two decades that this hypothesis needs to be revised, since a number of studies using natural tracer techniques (e.g., stable isotopes, fatty acids) have almost invariably found little solid evidence for a significant amount of mangrove-derived carbon in adjacent food webs, and that many of the earlier estimates may have been biased and should be revised (see Bouillon et al., 2008, for a recent overview). Relationships between fisheries or shrimp catch and the presence or extent of nearby mangroves are therefore unlikely to result from a direct trophic link, but rather from the effects of other factors such as the provision of a suitable nursery habitat, refuge from predators, or the provision of other food sources besides mangrove litter (see Sections 5, 8, and 10). The contribution of mangrove-derived organic matter in adjacent systems also appears to vary according to the environmental setting and geomorphology of the system, being more important in riverine/estuarine systems than in lagoon or island settings (Pineda, 2003).

Despite the evidence that mangroves are far less important as a food source for aquatic faunal communities than previously assumed, the assumption that they sustain intertidal food webs has endured much longer, and is often taken for granted. A number of prominent mangrove-dwelling species are indeed known to remove and consume leaf litter, in particular sesarmid crabs (e.g., Lee, 1998) and species such as the crab *Ucides cordatus* (Nordhaus et al., 2005) and the gastropod *Terebralia palustris* (Slim et al., 1997). Subsequently, there is a wealth of literature on the impact of mangrove invertebrates on litter dynamics, and on feeding experiments involving various species of mangrove tree leaves or their state of senescence (see Lee, 1998, for references). The amount of literature exploring the trophic importance of other primary producers, in contrast, is far more limited. The general view emerging from recent studies is that, although faunal communities may have a profound impact on litter dynamics (Lee, 1998; Kristensen et al., 2008), the role of mangrove litter in sustaining epibenthic communities is often fairly limited, and decreases when systems are more open with respect to material exchange with adjacent systems (Bouillon et al., 2004).

9.2. Importance of other carbon sources

When a variety of food sources are available, mangrove fauna show a remarkable diversity in their resource utilisation patterns of the different sources, with different faunal groups each occupying a different trophic niche (Bouillon et al., 2002). Sesamid crabs, considered to be key consumers of fallen litter, generally show the highest degree of dependency on mangrove carbon in comparison to other faunal taxa. However, in systems where other inputs are considerable, certain species, and in particular juveniles, show a high reliance on imported material and microphytobenthos (Bouillon et al., 2004). Moreover, several studies have indicated that mangrove litter alone would not suffice to meet the nitrogen requirements of sesamids (Skov and Hartnoll, 2002; Thongtham and Kristensen, 2005). In contrast to sesamids, fiddler crabs (*Uca* spp.) which are also typically highly abundant in mangrove systems rely mainly on microphytobenthos (France, 1998; Meziane et al., 2002). Other important groups of brachyuran crabs in mangrove systems are known to feed to an important extent on macroalgae (e.g., certain *Metapograpsus* spp.: Dahdouh-Guebas et al., 1999) or forage on other fauna (e.g., *Epixanthus* spp., *Scylla* spp.: Dahdouh-Guebas et al., 1999; Bouillon et al., 2004). Molluscs constitute the second major group of mangrove epifauna, and show equally diverse feeding preferences. This includes grazing on epiphytic algae on tree trunks or pneumatophores (e.g., *Onchidium* spp. and *Littoraria* spp.: Christensen et al., 2001; Lee et al., 2001; Bouillon et al., 2004), utilising surface organic matter and microphytobenthos (e.g., *Assiminea* spp. and *Cerithidea* spp.: Bouillon et al., 2004), selective or unselective filter-feeding on suspended matter (for a number of bivalves: Bouillon et al., 2004), and consuming mangrove leaf or propagule litter (*M. coffeus*: Proffitt and Devlin, 2005). In addition, some molluscs have particular adaptations, such as wood-boring shipworms (*Teredinidae*) which harbour symbiotic bacteria capable of N_2 fixation to supplement their N-poor diet (see Bouillon et al., 2002), and sacoglossan sea slugs, some of which retain functional chloroplasts from their food algae ('kleptoplasty': see Bouillon et al., 2008). Previous studies have likely been biased towards more conspicuous groups of fauna, with much less attention to infauna and meiofauna, where recent results suggest that a variety of chemo-symbiotic relationships exist in reduced mangrove sediments (see Bouillon et al., 2008). An integrated view of the importance of different primary sources for consumers, or of the quantitative impact of mangrove fauna on material processing, is still lacking, due to the absence of solid combined datasets on feeding preferences and secondary production or consumption rates. Moreover, there is likely no general 'mangrove food web' and the importance of various sources and the impact of fauna on organic matter turnover will significantly depend on the environmental settings and the faunal community composition.

A number of exclusion/inclusion experiments carried out in mangrove systems demonstrate that the presence and activity of macro-epifauna (e.g., gastropods, ocapodid crabs) cause a significant modification of the benthic habitat. The exclusion of fauna may increase the micro-epiphytic biomass due to the

absence of grazing (Branch and Branch, 1980; Kristensen and Alongi, 2006) and indirectly cause a modification of meiofaunal communities (Schrijvers and Vincx, 1997; Schrijvers et al., 1998; Carlén and Ólafsson, 2002). Such studies are scarce for mangrove systems, but offer a valuable alternative perspective on trophic interactions and on the mechanisms structuring mangrove faunal communities.

9.3. Linkages with adjacent systems

An important issue in the ecological and habitat function of mangroves is the importance of mangrove areas as feeding habitats for mobile or visiting fauna (Fry and Ewel, 2003). A number of studies suggest that mangroves can provide an important feeding habitat for foraging fish or prawns during high tide, based on observational evidence, gut content analyses, and stable isotope evidence.

For certain mangrove fish species, the importance of predation on mangrove food sources, such as sesamids and other invertebrates, has been indicated by a number of studies (e.g., Sasekumar et al., 1984; Thong and Sasekumar, 1984; Wilson, 1989; Sheaves and Molony, 2000; Lugendo et al., 2006). The accessibility of the intertidal zone and its potential role as a refuge or feeding habitat is constrained by the tidal regime and system characteristics (Sheaves, 2005; Lugendo et al., 2007b). Lugendo et al. (2007b) showed that feeding by fishes within mangroves is more common when they are permanently accessible than when they are only temporarily accessible due to the tidal regime. Nagelkerken and van der Velde (2004a) and Verweij et al. (2006b) demonstrated that the majority of fish species from a Caribbean mangrove-seagrass-reef system derived little or no food from the mangrove habitat, even though they were permanently inundated, which is consistent with the relatively low tidal amplitude in this region. A probable cause is that Caribbean island mangroves are narrow fringes which provide excellent shelter habitat but little food (Nagelkerken et al., 2000a). As a result, fish that shelter there during the daytime migrate to adjacent seagrass beds at night for most of their feeding activities, while the same species sheltering in seagrass beds derive their food entirely from within that habitat (Nagelkerken and van der Velde, 2004b; Nagelkerken et al., 2006).

9.4. Modelling mangrove food webs

The lack of basic data on the origin of organic matter sustaining different faunal communities is also propagated in modelling approaches to describe the food web structure and energy flows in mangrove systems. A number of studies (e.g., Manickchand-Heileman et al., 1998; Ray et al., 2000; Wolff et al., 2000; Vega-Cendejas and Arreguín-Sánchez, 2001) have used a network analysis approach, typically using Ecopath software (Christensen and Pauly, 1992) to construct an integrated view on energy flow in mangrove systems. While this approach has its own merits and, moreover, has the potential to be an excellent tool to estimate the potential effects of ecosystem changes on overall energy flow, the input data for

such models require solid knowledge on the feeding habits of critical ecosystem components. It is worth noting that, while recent studies stress the importance of microphytobenthos and allochthonous carbon sources to intertidal food webs, current models (cited earlier) have not included microphytobenthos in their analyses. The assumption that mangrove litter or mangrove-derived detritus represents the dominant food source for certain major groups of fauna (e.g., fiddler crabs; Wolff et al., 2000) is likely to significantly bias model output, since tracer studies indicate that mangrove-derived carbon contributes only marginally to these organisms' diets (e.g., France, 1998; Meziane et al., 2002; Bouillon et al., 2004). Such bias is inevitably transferred to higher trophic levels. In this context, there is clearly a need for more synergy between specific tracer studies, experimental field studies on trophic interactions, and trophic models in order for these modelling efforts to generate more realistic outputs, and for results from other approaches to generate more added value.

10. Mangroves as drivers of nearshore fishery production

10.1. The role of mangroves and estuaries in relation to fisheries

Over the last four decades, many studies have demonstrated a strong relationship between mangrove presence and fish catch (Turner, 1977; Yáñez-Arancibia, 1985; Pauly and Ingles, 1986; Lee, 2004; Manson et al., 2005; Meynecke et al., 2007), with fishery catch being influenced by the relative abundance of mangroves in a region. Correlations have also been found between the extent (area or linear extent) of mangroves and the catches of prawns (particularly banana prawns) in the fisheries adjacent to the mangroves (Turner, 1977; Staples et al., 1985; Pauly and Ingles, 1986; reviewed in Baran, 1999). Such studies provided important information on the fisheries–mangrove relationship and were the base for economic valuation of mangroves (e.g., Barbier and Strand, 1998; Grasso, 1998; Barbier, 2000). This observed relationship mainly derives from a group of economically important species classified as estuarine-dependent (Cappo et al., 1998) or (non-estuarine) bay-habitat-dependent (Nagelkerken and van der Velde, 2002). Mangroves, or similar environments, are the principal habitat for at least one part of their life cycle (Blaber et al., 1989; Nagelkerken et al., 2000b). Typically, the adults spawn offshore, producing eggs that disperse in the water column for varying lengths of time. The eggs then develop into planktonic larvae which move, or are carried by currents, into inshore and estuarine waters. The subadults or adults migrate out of the estuary or lagoon, and back towards the offshore areas or adjacent coral reefs. Therefore, mangroves could function as an important link in the chain of habitats that provide complementary resources and benefits, e.g., as nursery areas for fish, prawns and crabs (Sheridan and Hays, 2003; Crona and Rönnbäck, 2005), with spatial complexity at a scale that provides refuge to small prey, and abundant food for commercial species at certain stages in their life cycle (Chong et al., 1990).

10.2. Controversy in the literature: nursery ground and outwelling

Opinions vary as to the importance of mangrove habitats to fish and, by extension, to nearshore fisheries (Blaber et al., 1989; Thollot, 1992). Although, studies have documented greater abundances of juvenile species in mangroves than in other estuarine and inshore habitats (Robertson and Duke, 1990a; Nagelkerken and van der Velde, 2002), other studies found a significant contribution by saltmarshes (Connolly, 1999). Controversy also surrounds the measurement of nursery ground values (Sheridan and Hays, 2003). Beck et al. (2001) and Adams et al. (2006) gave a detailed discussion of the nursery habitat concept, where the value of nursery grounds is measured in terms of numbers contributed to adult populations: either the average number of individuals per unit area (Beck et al., 2001) or the total number of individuals per habitat (Dahlgren et al., 2006; see also Layman et al., 2006), in an attempt to simplify the measurement of nursery function and to identify core components (e.g., habitats) that are most important in maintaining overall ecosystem function. Sheaves et al. (2006) argued that this approach is oversimplistic and relates to the value of a nursery from a short-term, fisheries perspective. It is important to develop a better understanding of habitat connection and how these connections can be maintained (Meynecke et al., 2007).

The discussion about the importance of tidal wetlands for fisheries includes the debate on the passive transport of dissolved and particulate nutrients and detritus from productive tidal wetlands to coastal and offshore waters, termed 'outwelling'. Outwelling was considered critical in supporting the secondary production of many coastal fisheries (Nixon, 1980). The transport of nutrients between estuarine and coastal systems is now recognised as a complex and dynamic process (Ford et al., 2005; Webster et al., 2005), and the general model of outwelling appears too simplistic (Kneib, 1997; Ford et al., 2005; see also Section 9). Recent studies using stable isotope analysis and other trophic tracers indicated that offshore fisheries may not be driven through the outwelling of nutrients from estuaries (Loneragan et al., 1997; Connolly, 1999; Chong et al., 2001; Melville and Connolly, 2003) and that, for example, mangrove-derived nutrients only contribute directly to the food webs of some animals within highly restricted areas, e.g., mangrove-lined creeks (Loneragan et al., 1997). Separating the contribution from mangroves, seagrass and salt marshes using casually deployed tracers, such as natural abundance stable isotope analysis, is still problematic since benthic and pelagic organisms seem to rely on different mixes of organic matter sources.

10.3. Studies linking fish catches with mangroves and estuaries

The most direct approach to search for links between mangroves and fisheries is to use fish catch data in comparison with mangrove or estuarine parameters. Manson et al. (2005) found a significant influence of mangrove forest characteristics

(e.g., perimeter and area) on mangrove-related and estuarine fishery species, while latitude was the only variable influencing catch of offshore species along the north-eastern Australian coast. The earliest studies in this field were completed by Macnae (1974) who showed that inshore fish production in Malaysia was related to mangrove area. Turner (1977) found a positive correlation between penaeid shrimp catches and the vegetated surface area of estuaries in the Gulf of Mexico. Martosubroto and Naamin (1977), working in Indonesia, showed a positive correlation between annual fish landings and mangrove area. They implied that a certain minimum mangrove area is necessary for high production, a point also noted by Pauly and Ingles (1986) who suggested that the impact of destroying a mangrove area might be greater if the area is small and residual. A review of literature (Baran and Hambrey, 1998) demonstrated the dependence of fish harvests on estuarine environments, and showed the importance of these systems in terms of sustainable management of the coastal resource. Rönnbäck (1999) and Barbier (2000) identified and synthesised ecological and biophysical links of mangroves that sustain seafood production from an economic perspective, and Manson et al. (2005) tried to go beyond the correlative approach and developed a new framework on which evaluations can be based.

The review presented here found a total of 27 studies comparing commercial catch with estuarine habitats, in particular mangroves in tropical and subtropical zones, over the last four decades. Most of the analyses in these studies have resulted in positive correlations, assuming that the area of mangroves or tidal wetland habitat directly translates to the catch of commercial fisheries (Manson et al., 2005; Meynecke et al., 2007). Estimates of the amount of commercial catch explained by the presence of mangroves or estuaries range from 20 to 90% (Fig. 1). The most common variables used were mangrove area, followed by linear extent and intertidal area or estuarine size. Over 15 studies used mangroves as a proxy and seven studies used the extent of estuaries, coastal vegetation or shallow water. Most studies were undertaken in Australia, Asia

and the U.S.A., whereas West Africa and South America were under-represented (see also Faunce and Serafy, 2006, in their review on mangroves–fish studies). Estimates in studies covering worldwide tropical commercial catch range between 21% (Houde and Rutherford, 1993) and 30% (Naylor et al., 2000) for an estuarine–fish catch relationship, 38% for a mangrove–prawn catch relationship (Lee, 2004), 53% for a mangrove area–fish catch relationship (Pauly and Ingles, 1986), and 54% for an intertidal wetlands–prawn catch relationship (Turner, 1977).

Depending on species, location and time scale, the relationships between commercial catch and mangroves vary largely, indicating that the link is more complex than a linear function. The predictors used in the regression analyses are themselves strongly correlated, and catch statistics are often not well delineated. There is high variation within the data sets (mangrove forest distribution, commercial records, effect of stock size and fishing pressure) and difficulty in distinguishing links against a background of highly variable temperature, rainfall, ocean currents, and fishing effort.

On the other hand, estuarine or lagoonal habitats and the strong links between them have been neglected in past studies (Sheridan and Hays, 2003). A certain combination of habitats and their accessibility is likely to explain the importance of estuaries to nekton (Cappo et al., 1998). Evidence is mounting that permanently inundated fringing mangroves in the Caribbean primarily serve as daytime refugia for a major component of fishes occupying various habitats in lagoons or bays (Nagelkerken et al., 2000a; Valdés-Muñoz and Mochek, 2001), while fishes from adjacent habitats feed in large intertidal mangroves at high tide (Sheaves, 2005; Lugendo et al., 2007b). This suggests for some species that fish production attributed to mangroves may not necessarily be derived from this habitat alone. Evidence exists, for example, that fish abundance and species richness are higher when mangroves and seagrass beds occur together rather than in isolation (Robertson and Blaber, 1992; Nagelkerken et al.,

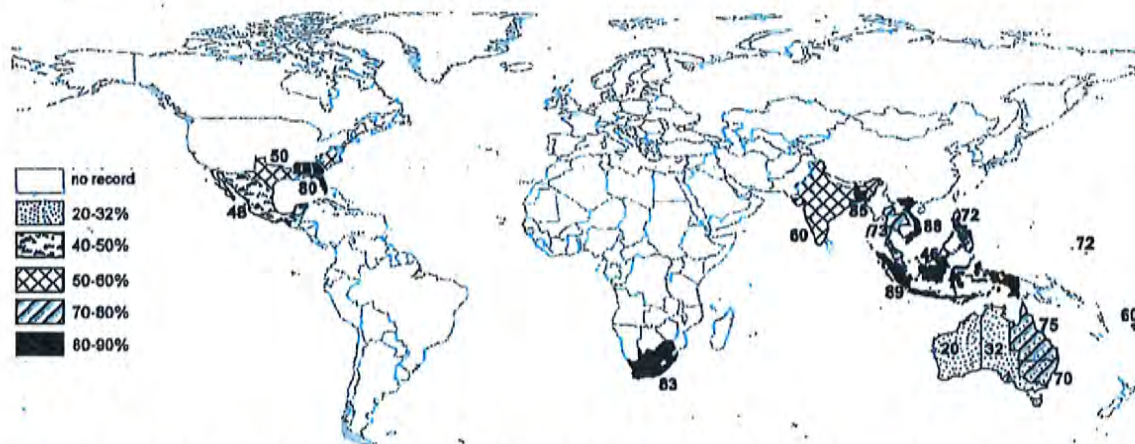


Fig. 1. World map showing the percentage of commercial fish catch explained by either mangroves or estuarine presence from 27 reviewed studies of which 22 provided quantitative data for this linkage. For countries where more than one study was conducted the conservative number has been chosen for display (for a detailed listing see Manson et al., 2005; Meynecke et al., 2007).

2001; Dorenbosch et al., 2006a,b; Jelbart et al., 2007). Many fish species occupying lagoons appear to show ontogenetic fish movements between seagrass beds, tidal channels and mangroves (Rooker and Dennis, 1991; Nagelkerken et al., 2000c; Nagelkerken and van der Velde, 2002, 2003; Eggleston et al., 2004; Lugendo et al., 2005), making the individual contribution of mangroves difficult to determine (Adams et al., 2006).

Worldwide, most studies on mangrove fish communities and their linkages with offshore fisheries have been done in estuarine mangrove systems (Nagelkerken, 2007). However, there are hundreds to thousands of small islands in the Caribbean and Indo-Pacific which only harbour non-estuarine mangroves located in marine embayments and lagoons. Although their surface area is mostly much smaller than that of large estuarine mangrove forests, they may be important on an island scale for coral-reef associated fisheries. Only in this millennium have studies started focusing in more detail on the connectivity between non-estuarine mangroves (and seagrass beds) and adjacent coral reefs with regard to fish movement (Nagelkerken, 2007), mostly based on multiple habitat density comparisons using a single census technique and distinguishing between fish size classes. This has resulted in the identification of several (commercial) reef fish species which appear to depend on mangroves while juvenile (e.g., Nagelkerken et al., 2000b,c; Cocheret-de-la-Morinière et al., 2002; Christensen et al., 2003; Serafy et al., 2003; Eggleston et al., 2004; Dorenbosch et al., 2007). Studies comparing reef fish communities near and far from mangrove habitats, and with the presence or absence of island mangroves, have shown that the dependence on mangroves is species-specific, but appears to be high for various reef species (Nagelkerken et al., 2000b, 2001, 2002; Mumby et al., 2004; Dorenbosch et al., 2004, 2005, 2006a,b, 2007). Otolith microchemistry studies have also suggested a linkage between mangroves and coral reefs (Chittaro et al., 2004). Mumby (2006) developed algorithms to describe various aspects of mangrove-reef connectivity that can be used for management purposes.

Future investigations of tidal wetlands-fisheries links, should be based on an understanding of connectivity and should use standardised data collection. More research is required documenting the natural temporal and spatial variability of assemblages in fish habitats. Although there is no direct evidence of the fish catch-mangrove dependence, studies so far clearly infer a strong link emphasising the need to reverse the loss of mangroves and tidal wetlands (FAO, 2003) from both natural and anthropogenic causes.

11. Mangroves as habitats for amphibians and reptiles

Several independent evolutionary lines of reptiles and amphibians have successfully colonised, and are variously dependent on, mangrove ecosystems. These include frogs, marine and freshwater turtles, crocodilians, lizards, and marine and terrestrial snakes.

11.1. Frogs

Globally, little is known of the amphibian fauna inhabiting mangroves (Kathiresan and Bingham, 2001). Amphibians are generally intolerant of saline conditions found within mangroves, although many species are associated with estuarine habitats such as frog species from the genus *Eleutherodactylus* (Hedges and Thomas, 1992). Notable exceptions include the crab-eating frog *Rana cancrivora* from Southeast Asia that is adapted to salt water during all stages of its life cycle (Dunson, 1977), and *E. caribe* which is unique among species in the genus by inhabiting the flooded, *Rhizophora mangle*-dominated mangrove habitat of the Tiburon Peninsula of Haiti in the Caribbean (Hedges and Thomas, 1992).

11.2. Turtles

Freshwater turtles are known to inhabit estuaries, but little is known of their specific habitat requirements. The mangrove terrapin *Batagur baska* (from Central and Southeast Asia), and the painted terrapin *Callagur borneoensis* (from the Sundarbans, Bangladesh) are large freshwater chelonians that are known to inhabit tidal creeks and rivers (Blanco et al., 1991). Both species are listed as critically endangered and rely on riparian vegetation, including mangrove fruit, as a food source (IUCN, 2006; UNEP-WCMC, 2007). *Callagur borneoensis* nests on ocean beaches and the hatchlings have to swim through seawater to reach the river mouths; however, they are not physiologically adapted to the high water salinity in estuaries for extended periods (Dunson and Moll, 1980).

Five species of marine turtles have global distributions in tropical and temperate waters ranging from the lower reaches of estuaries to oceanic/pelagic habitats, while two species have relatively restricted ranges: the flatback turtle *Natator depressus* is endemic to the Australian-New Guinea continental shelf (Limpus and Chatto, 2004), and the Kemp's ridley turtle *Lepidochelys kempi* is restricted to the warm temperate zone of the North Atlantic Ocean (Bowen et al., 1997). Marine turtles occupy different habitats at different stages of their life cycle: natal beaches, mating areas, inter-nesting habitat, feeding areas and pelagic waters (EA, 2003). Juvenile movements in neritic developmental habitats are little known compared to adult reproductive migrations. There is also a lack of knowledge of life history and habitat requirements away from nesting beaches. Habitat selection may be significantly influenced by biotic factors such as the availability and/or quality of food or the co-occurrence of predators. In nearshore waters, abiotic factors such as tidal flux, water temperature and depth, salinity and turbidity may influence the occurrence of marine turtles via direct (physiological tolerance) or indirect (effects on prey or predators) mechanisms (EA, 2003). Habitat quality for *L. kempi* in the nearshore waters of the north-western Gulf of Mexico was influenced by water temperature and depth, salinity, dissolved oxygen and turbidity, which influenced the distribution and abundance of predators and prey (Metz, 2004). In general, there is a lack of detailed studies of the habitat requirements of in-water life history stages of marine turtles (Metz, 2004).

Many animals alter their movements and home range in relation to the particular type and quality of the habitat occupied. Fidelity of marine turtles to foraging areas has been demonstrated in several studies (Limpus and Limpus, 2000; Godley et al., 2002). Green turtles (*Chelonia mydas*) have a pelagic existence for the first 3–5 years, after which they recruit to coastal waters and occupy a series of developmental habitats, including mangroves, until sexual maturity (Makowski et al., 2005). These movements coincide with an ontogenetic dietary shift from an omnivorous to a herbivorous diet including *Avicennia marina* fruit (Pendoley and Fitzpatrick, 1999), cotyledons and propagules (Limpus and Limpus, 2000; H.M. Penrose, unpublished data). Indirectly, mangroves can provide a food source for marine turtles by providing habitat for macroalgae that grow on roots, trunks (Cribb, 1996) and pneumatophores (H.M. Penrose, unpublished data). *Rhizophora mangle* is also known as a developmental foraging ground for Kemp's ridley turtles in the coastal waters of their range (Schmid, 2000). Despite a paucity of data, evidence suggests that mangroves are important to marine turtles and further research is needed to determine species-specific differences in mangrove use in a range of environmental settings.

11.3. Crocodilians

The 23 extant crocodilian species (crocodiles, alligators, caimans and gharials) are integral components of aquatic, wetland and marine/estuarine ecosystems in tropical and subtropical regions, with most species requiring large areas of undisturbed habitat to maintain viable populations (Hutton and Webb, 1990). The relative importance of mangroves as a habitat for crocodilians, compared with other habitats, is variable, due to the wide diversity in their biology and ecology. However, as nurseries for fish and other marine animals on which crocodilians feed, mangroves provide an abundance of food sources at all life history stages. The estuarine crocodile *Crocodylus porosus* is one of the most notable species associated with mangroves. Generally, *C. porosus* do not nest in mangroves but are found nesting in vegetation fringing mangrove areas (Webb et al., 1977; Magnusson, 1980; Webb et al., 1983). However, the indirect importance of mangroves for *C. porosus* has been demonstrated in Sri Lanka where the decline of this species is linked to an increase in coastal runoff due to clearing of mangrove trees, resulting in the destruction of crocodile nests and eggs (Santiapillai and de Silva, 2001). Prop-roots of *Rhizophora* spp. are also known to provide an important structural refuge for hatchlings (Santiapillai and de Silva, 2001).

11.4. Lizards

Many lizard species, ranging from geckos to iguanas, are known to inhabit intertidal mangrove forests. Some lizards are terrestrial species that enter mangroves on an opportunistic basis to access resources, whereas others live a semi-aquatic existence, with the monitor lizards *Varanus* spp. being the most notable. The rusty monitor *Varanus semiremex*, which is

restricted to the east coast of Queensland, Australia, are particularly dependent on *Avicennia marina*, utilising hollow limbs of living mature as well as dead trees (Wilson and Knowles, 1988).

11.5. Snakes

Pythons (*Python morulus*) and king cobras (*Ophiophagus hannah*) make opportunistic movements from terrestrial to intertidal mangrove habitats to feed (Macintosh and Ashton, 2002). Some sea snakes (family Hydrophiidae) access mangroves during high tide, whereas others, such as the 'primitive' *Ephalophis greyae* of Western Australia, which has retained its terrestrial mode of locomotion, undertake foraging migrations across the dry mangrove substrate during low tide in search of gobiid fish (Storr et al., 1986; H.M. Penrose, unpublished data). Other snake species rely on mangrove trees as a physical habitat structure, such as *Myron richardsonii* that is endemic to Northern Australia (Guinea et al., 2004), and *Boiga dendrophila* of Southeast Asia and Australia, both members of the family Colubridae (Macintosh and Ashton, 2002). The structure of snake communities in the mangroves of south-eastern Nigeria was described by Luiselli and Akani (2002). Eighteen snake species were recorded in mangroves, whereas 43 species inhabited neighbouring habitats (rainforest and forest-plantation mosaics). Of these 18 species, 50% were arboreal, 22% terrestrial, 11% terrestrial-arboreal, 11% semi-aquatic, and 6% described as very generalist. Species preyed on a wide range of organisms including lizards, birds, bird eggs, frogs, mammals and fish. The relative frequency of the African rock python *Python sebae* in the mangroves, compared with other habitats, suggest that mangroves may represent an important refuge or dispersal corridor for this threatened species. Luiselli and Akani (2002) highlight the need for further research on the rich diversity of the high trophic level arboreal snake species, and the preservation of the habitat mosaics within mangrove creeks, where the great majority of snakes are found.

12. Mangroves as habitats for birds

Mangrove habitats play host to a moderate number of bird species around the globe. Most diverse are the Queensland mangroves of Australia which host 186 bird species (Noske, 1996). Other counts are 135 in Peninsular Malaysia (Nisbet, 1968), 125 in Guinea-Bissau, West Africa (Altenburg and van Spanje, 1989), 104 in north-western Australia (Noske, 1996), 94 in Surinam (Haverschmidt, 1965), and 84 in Trinidad (Ffrench, 1966). The forests are strongly zoned with few tree species and a sparse understory producing a simple vertical profile.

When considering the bird faunas of these and other mangrove habitats around the globe, it is interesting to speculate on the sources of the species that make up these communities. To what extent have mangrove forests played the role of independent sites of species diversification, to what extent have they been recipient habitats, and to what extent have

they been the source of colonists for other habitats? What ecological forces may have driven these processes? This suggests a way to review the avifauna of mangroves, namely those that have a tolerance for a narrow range of habitats (stenotopic species), mangrove invaders, and mangrove species that move out into other habitats.

12.1. Mangrove specialists

The first interesting observation is that there are few true mangrove specialists. No African species has been found to be exclusively mangrove-dependent, the closest being the insectivorous sunbird *Anthreptes gabonicus* which is found in mangroves but also in riverine woods hundreds kilometres inland. Mangrove habitats in Surinam host 94 bird species, while in Trinidad these habitats support 84 bird species. Only one species, the rufous crab-hawk (*Buteogallus aequinoctialis*) is restricted there to mangrove habitats (Haverschmidt, 1965; Pfrench, 1966). In north-western Australia, 16 of 104 species are more or less confined to mangroves, in eastern Australia, 9 of 106. Just 11 species are mangrove-dependent there, namely great-billed heron *Ardea sumatrana*, striated heron *Butorides striata*, chestnut rail *Eulabeornis castaneoventris*, collared kingfisher *Todiramphus chloris*, mangrove robin *Peneoenanthe pulverulenta*, kimberley flycatcher *Microeca flavigaster tormenti*, white-breasted whistler *Pachycephala lanioides*, dusky-gerygone *Gerygone tenebrosa*, yellow silver-eye *Zosterops lutea*, red-headed myzomela *Myzomela erythrocephala*, and mangrove honeyeater *Lichenostomus fasciolaris*.

In Peninsular Malaysia, Wells (1999) cites nine species (brown-winged kingfisher *Pelargopsis amauroptera*, ruddy kingfisher *Halcyon coromanda*, greater goldenback *Chrysocolaptes lucidus*, mangrove pitta *Pitta megaryncha*, black-hooded oriole *Oriolus xanthornus*, great tit *Parus major*, dusky warbler *Phylloscopus fuscatus* (a non-breeding visitor), mangrove blue flycatcher *Cyornis rufigastrea*, and copper-throated sunbird *Leptocoma calcostetha*) that depend exclusively on mangrove forests, or mangrove forest out to an immediate fringe of Nipa palm swamp or strand or plantation woodland. Three others (great-billed heron *Ardea sumatrana*, great egret *Ardea alba*, and milky stork *Mycteria cinerea*) depend on it exclusively for nesting but feed elsewhere. Grey heron *Ardea cinerea*, striated heron, and black-crowned night heron *Nycticorax nycticorax* now also breed inland. Only the brown-winged kingfisher, mangrove pitta, and copper-throated sunbird are global mangrove specialists. In regions outside the peninsula some of these specialists use other habitats. Ruddy kingfishers and great tits inhabit inland forests in Thailand, and the laced green woodpecker enters inland forest north of latitude 60°N. The fact that none of these birds has invaded inland non-forest habitats suggests that the peninsular populations have become specialised for mangrove habitats. The abundant, mainly frugivorous, families of adjacent dipterocarp evergreen forest are conspicuously rare, namely babblers, barbets, bulbuls, leafbirds, hornbills, and pheasants.

12.2. Mangrove invaders

In some cases, species may be better adapted to non-mangrove habitats but use mangrove either because their preferred habitats are not available locally, or because some mangrove forests provide marginal habitat for some individuals whose principal populations occupy other adjacent forest habitats. Wells (1999) analysed the species origins of the birds in mangroves of Peninsular Malaysia, where inland forests once abutted with mangrove trees over long stretches of the coast until well into the 20th century. Around one-third of the avian mangrove community (mainly kingfishers, sunbirds, warblers and woodpeckers) is shared with its former inland forest habitats which may indicate how marginal mangroves were for at least some of the inland forest species that used their back zone. Leafbirds, broadbills, giant woodpeckers, and others formerly reported in the mangroves are now scarce or absent. Wells (1999) suggested that if more of the interface between mangroves and inland forests had been explored before the interface had been destroyed, more shared species may have been found in the mangroves. The same may be true in Western Australia, where the proximity of mangroves and so-called vine forests may maintain species in mangroves (Noske, 1996).

Given the nature of some current mangrove specialists, it seems likely that mangroves were attractive for some open-forest adapted species. The latter habitats would have been much drier during the Pleistocene-glacial periods, while the mangroves were probably more extensive providing an essential refuge for birds from shrinking open forests. Noske (1996) cites the existence of mangrove specialists such as mangrove fantail *Rhipidura phasiana*, broad-billed flycatcher *Myiagra ruficollis*, kimberley flycatcher, mangrove gerygone *Gerygone levigaster*, and mangrove honeyeater, species with relatives in these open forest regions, and even inland regions. However, given the huge linear continuity of continental mangrove forests, it is almost impossible to guess where any particular specialist originated.

12.3. Species that move out to non-mangrove habitats

Some scientists suggest that the mangrove assemblage is an exporter of bird species to non-forest habitats. This assumption is based on the birds having niches that developed in an environment of low plant species diversity, itself derived from instability of the habitat over various time scales. This, in turn, results in a relatively simple structure and reduced niche space. Bird species niches are broad enough to predispose mangrove birds to simplicity and instability elsewhere (Noske, 1995; Ward, 1968). Noske (1995) revealed that certain groups (woodpeckers, passerines) found both in mangroves and non-forest habitats have narrower niches than those species that still live exclusively within the mangroves, perhaps because of competition once outside the mangrove habitat, or because the simplicity of the mangroves precluded such specialisation.

Ward (1968) remarked how mangroves have contributed many bird species to rural and urban habitats in Peninsular Malaysia, and suggested that this may be due to the simple

floristic and habitat structure shared by mangroves and these anthropogenic habitats. Broad niches and tolerances by mangrove bird species may enable them to exploit these newer habitats. Noske (1995) related broad niches of mangrove bird species to fluctuations in food availability driven by tidal cycles. He interpreted the absence of colonisation of Australian urban and exurban habitats by mangrove bird species as being a consequence of the sharp contrast between the stable, evergreen, fire-protected mangroves, and the highly seasonal, deciduous, and fire-prone savannas that abut them. An alternative explanation could be that such savannas never were empty of species as the suddenly deforested sub-coastal lowlands of Peninsular Malaysia, which provided the opportunity for colonisation (D.R. Wells, personal communication).

As a rule, most bird species found in mangroves are also found in other habitats, but how would we tell in which direction colonisation took place: into or out of mangroves? We know that most of the West African mangrove bird species in Guinea-Bissau were more common in adjoining forests and rice fields. Some use the mangrove for roosting only (Altenburg and van Spanje, 1989), and in Peninsular Malaysia 46 residents and at least 6 non-breeding visitors to mangroves also occur in 1 or more of the original inland forest formations (Wells, 1999). Some kinds of birds, such as aerial insectivores (nightjars, tree swifts, rollers, swallows), may nest in mangroves but feed over a wider range of habitats while others (white-vented myna *Acridotheres javanicus*, java sparrow *Lonchura oryzivora*) may nest in mangroves but never feed there (Medway and Wells, 1976).

Almost as many bird species that we now find in mangroves can also be found in open coastal areas, agricultural, and urban areas that meet the truncated mangrove succession inland. In Australia, the floristically rich north-eastern mangroves have fewer mangrove specialist bird species than the north-western mangroves, though it is unclear why. Mangrove habitats may have remained widespread during the Pleistocene glacial periods when rainforests may have contracted to small patches, providing places for many of the rainforest species to survive, and then have been a source for re-populating re-expanding rain forests.

12.4. Synthesis

The most intriguing insights of the role of mangroves for bird communities come from a consideration of mangroves in their broader context, namely their relationships with other habitats, both current and historic. We should also ask why there are so few true mangrove specialists. It is interesting to speculate on what current species may tell us about past habitat relationships. What roles may mangroves have served as stable habitats during times of climatic change in the past? We do not have an accepted model of how mangrove forest might have behaved in response to sea-level changes. While one can envisage the mangrove system extending seaward as sea levels fall, observation shows that a simple change in local long-shore currents can eliminate the forest, while a rising sea level would probably reduce mangroves. So what was the distribution and

availability of mangroves in the past? How did changes in mangrove availability push species out into non-mangrove habitats? We can imagine a scenario where mangroves change sequentially from becoming refugia for drying forests to expellers of birds as they shrink with rising sea levels, and a resulting two-way flow of species into and out of these habitats over time.

It is not clear to what extent mangrove forests support relict populations. Noske (1995) reports that one of his transects in Malaysia still retained small areas of original back-mangrove forest, and several decades after the link with inland forest was broken this still contained a few 'inland forest' birds, such as velvet-fronted nuthatch *Sitta frontalis*. Bird species of adjacent inland habitats, and those habitats formerly adjacent before their destruction, use mangroves in different ways.

As one would predict, the greatest bird species diversity occurred in the more plant-species diverse back-mangrove zones, where colonisation of the mangrove from the inland species once occurred, and where it interfaces with habitats inland (Wells, 1999). Nisbet (1968) suggested that the current scarcity of overlapping forest bird species in the mangroves of Peninsular Malaysia is the result of the near-total clearance of the transition zone between mangrove and inland forests. Some formerly widespread species of this zone, such as the giant woodpeckers, have all but disappeared from their mangrove haunts. Noske (1996) suggested that the relatively small number of mangrove bird specialists in Peninsular Malaysia resulted from long historical and continuous contact between mangrove and other forest habitats and so no geographical block to gene flow was present. However, Peninsular Malaysia has more species than West Africa and Surinam. This could in turn be explained by the direct contact of its mangrove ecosystem with that of the rest of tropical Asia, acting as a corridor for immigrants from outside rather than by generation of species from within the habitat.

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INFLUENCE OF MANGROVE DETRITUS IN AN ESTUARINE ECOSYSTEM

Michael Fleming, Guanghui Lin and
Leonel da Silveira Lobo Sternberg

ABSTRACT

This study has determined the relationship and extent of mangrove reduced carbon flow and its contribution to the diet of higher consumers in a nearby seagrass bed using stable isotope techniques. In a seagrass bed approximately 80 m from a riverine mangrove community, detritus from seagrass or other marine sources of carbon such as epiphytes (*Stilophora rhizodes*) or phytoplankton the major contributor of reduced carbon. Mangroves make only a localized contribution to the food chain, thus contributing a relatively low proportion of reduced carbon to this seagrass community.

The major primary producers in southern Florida fringe coastal communities are seagrasses and mangroves. *Thalassia testudinum* dominates the seagrass beds offshore, with *Halodule wrightii* as a secondary species. *Rhizophora mangle* dominates and *Avicennia germinans* and *Laguncularia racemosa* contribute significantly to the riverine and fringe forests. The detrital material formed by the partial decomposition of the leaves of these vascular plants may be the basis of trophic food chains in coastal ecosystems of southern Florida. Mangrove forests are net exporters of particulate organic carbon (Golley et al., 1962; Heald, 1969; Lugo et al., 1976). Further, mangrove carbon supports food chains within riverine and basin communities along the southwest coast of Florida (Odum and Heald, 1975; Zieman et al., 1984). In areas where seagrass and mangrove communities are in close association, however, the relative contribution of the seagrass and mangrove detritus to the food chain is difficult to determine. Mangrove estuarine environments are important nursery areas for fish populations, yet large numbers of organisms are supported by seagrass beds, the magnitude being related to the seagrass cover density (Zieman et al., 1984). This paper addresses the question of whether the influence of mangrove detrital flow is far reaching or localized. To answer this question we used stable carbon isotope abundance analysis of organisms from a mangrove riverine community and two seagrass communities, one of which was geographically close to a mangrove stand and the other was without any apparent mangrove detritus input. Stable isotope analysis provides a powerful tool to determine carbon flow in this particular ecosystem because there is a significant difference in the carbon-13 content of the primary producers in question (mangroves and seagrasses, Zieman et al., 1984). Further, consumers of mangroves and seagrass can be identified by isotopic analysis, because diet switching and turnover experiments for carbon clearly show that diet is the primary determinant of the animal isotopic composition (see Fry and Arnold, 1982). Unfortunately, nitrogen isotope analysis cannot be used in this situation since in some cases there are no differences in natural ^{15}N abundances between seagrass and mangrove detritus (Zieman et al., 1984).

METHODS

The two study sites were located in the coastal areas of central Biscayne Bay, Florida (Fig. 1). The first site was a tidal stream in Matheson Hammock Park extending approximately 324 m into a fringe/

L. da Silveira Lobo Sternberg to whom correspondence should be sent.

riverine mangrove forest dominated by *R. mangle* with seagrass beds running parallel to the shore. Seagrass beds extend from about 3 to 4 m away from the shore to Biscayne Bay. The depth of the tidal stream averaged 1.4 m with extremes of 0.4 m and 2.4 m. The depth along the transect extension from the mouth of the stream averaged 0.6 m. The second site, located at Bill Baggs State Park (details not shown in Fig. 1), was a strip of beach with seagrass beds 8–10 m offshore and *Casuarina* (*Casuarina* sp.) stands 100–200 m from the beach. Samples were collected in the Bill Baggs site on 10 April 1987. At the first site a transect following the course of the tidal stream was established from the most inland accessible point to the mouth, and from the mouth to a point 180 m offshore perpendicular to the coast with 15 sampling locales approximately 30 m apart (Fig. 1). Samples from this transect were collected on 2 December 1986. In addition samples approximately 200 m south of the stream mouth were collected on 10 April 1986. Two random handfuls of detrital material were retained from each sample point in the first 480 m of the transect. Pieces of seagrasses and mangroves were separated from each other and cleaned of epiphytes for more accurate results. Samples were dried at 60°C, and each component was weighed before grinding and combustion. For the seagrass species equal weights of five leaves were ground and mixed together. Carbon isotope ratios were determined from an aliquot of this mixture. Organisms such as crabs, snails, fishes and oysters were collected within 18 m of each sample point by hand or with a sweep net. Tissue from birds from Florida Bay was collected from victims of storms or natural diseases. It was impossible, however, to determine their exact location because of their high mobility. Muscle tissue was dissected out, freeze dried, and combusted.

Organic material from consumers was frozen for storage prior to cleaning, sorting, identification, and processing. Animal flesh was extracted (in some cases a HCl wash was necessary to remove carbonate material), freeze-dried, and ground before combustion. Approximately 5 mg of ground sample was sealed with copper and cupric oxide in evacuated Vycor cuvettes and combusted at 800°C as in Northfelt et al. (1981). Carbon-13 abundance is expressed in δ units where

$$\delta^{13}\text{C}\text{‰} = \left[\frac{R_{\text{sample}}}{R_{\text{standard}}} - 1 \right] 1,000, \quad (1)$$

and R is the $^{13}\text{C}/^{12}\text{C}$ ratio of sample and standard, respectively. The standard used was PDB carbonate, and the precision of analysis was $\pm 0.2\text{‰}$. Carbon isotope ratios were measured in a PRISM mass spectrometer (VG) in the Department of Geology of the University of Miami.

RESULTS

The percentage of mangrove detritus present relative to seagrass detrital material and other vegetative matter along the transect decreased from an average of 80% ($N = 6$) along the stream to about 10% at 90 m beyond the mouth of the stream (Fig. 2a). The location of this sharp drop in mangrove detritus, however, probably varies somewhat with tidal influences and the season of the year. There was a large difference between $\delta^{13}\text{C}$ values of mangrove ($-26.7 \pm 0.3\text{‰}$) and seagrass detritus ($-15.3 \pm 1.2\text{‰}$, Fig. 2b). $\delta^{13}\text{C}$ values of flesh from heterotrophic organisms increased along the transect from the mangrove estuary to the seagrass bed. There was a highly significant difference of 4.6‰ between the average $\delta^{13}\text{C}$ value of the organisms living in the riverine forest and those living in the flat beyond the mouth ($T = 11.9$, $P < 0.01$). Unlike mangrove detritus abundance, the differences between $\delta^{13}\text{C}$ values of mangrove and seagrass organisms are less sensitive to tidal and seasonal effect, since the carbon isotope ratio of an organism is an integrative indicator of the diet over a longer period of time. A smaller but significant difference was observed between $\delta^{13}\text{C}$ values of organisms sampled in front of the effluent flow (approximately 20 m to 180 m away, $-19.6 \pm 1.0\text{‰}$) and those sampled away from the effluent flow ($-18.6 \pm 1.5\text{‰}$, $T = 2.42$, $P < 0.05$).

The average $\delta^{13}\text{C}$ value of seagrass at Matheson ($-15.3 \pm 1.2\text{‰}$) was significantly less than that of seagrass at Bill Baggs ($-8.7 \pm 0.5\text{‰}$, $T = 8.4$, $P < 0.01$, Table 1). Also shown in Table 1 are the $\delta^{13}\text{C}$ values of organisms living at Matheson Hammock Park and Bill Baggs. $\delta^{13}\text{C}$ values of organisms living in the seagrass beds of Bill Baggs ($-15.4 \pm 1.0\text{‰}$) were significantly higher than those living in seagrass beds in Matheson Hammock Park ($T = 6.2$, $P < 0.01$). In both cases,

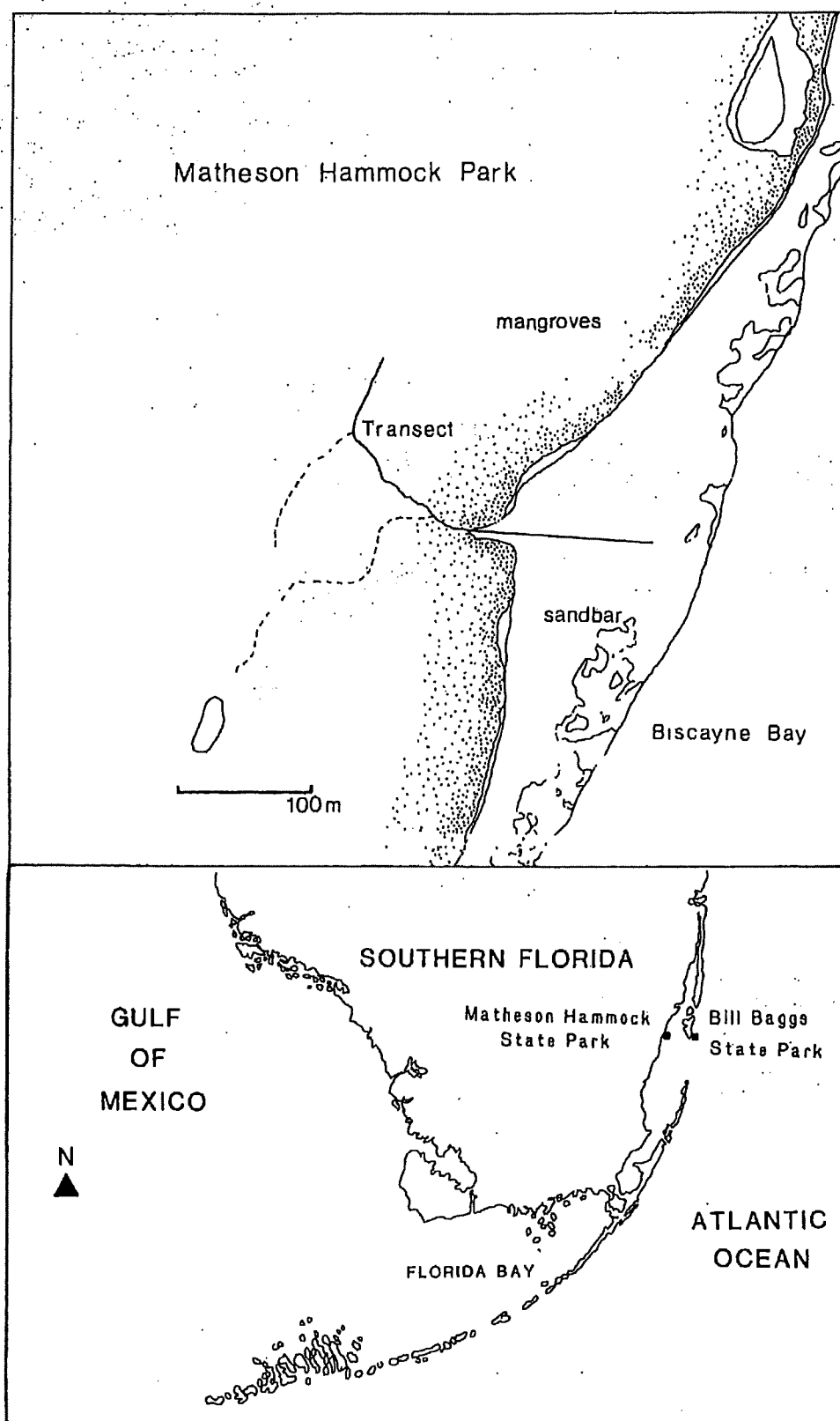


Figure 1. Location of sampling sites in southern Florida, and a detailed map of the Matheson Hammock Park site.

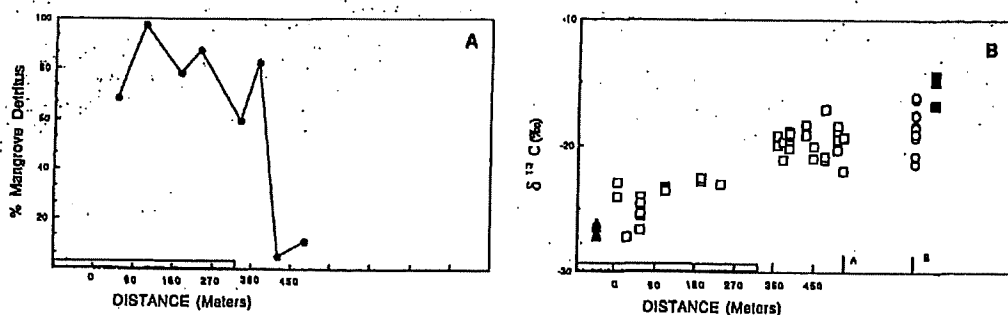


Figure 2. Percentage of identifiable mangrove detritus along the Matheson Hammock Park transect (Fig. 2a). $\delta^{13}\text{C}$ values of mangrove ($\blacktriangle$), seagrass ($\blacksquare$), several organisms collected along the Matheson Hammock Park transect ($\square$), and organisms collected away from the mouth of the estuary ($\circ$) (Fig. 2b). White rectangle at the bottom of graph indicates the length of the transect occurring within the mangrove forest.

however, the $\delta^{13}\text{C}$ values of the organisms were significantly lower than those of seagrasses at each site ($T = 6.96$, $P < 0.01$ for Bill Baggs, $T = 3.7$, $P < 0.01$ for Matheson Hammock Park). The isotope ratios of two species of birds (*Ardea herodias* and *Ajaja ajaja*) occupying a higher trophic level together with $\delta^{13}\text{C}$ values of their potential food are shown in Figure 4.

DISCUSSION

The ability to trace the carbon flow in an estuarine ecosystem by stable isotope techniques is possible because of large differences between the $\delta^{13}\text{C}$ values of mangroves and seagrasses. Further, there is little change in $\delta^{13}\text{C}$ values during the decomposition of mangrove and seagrass, thus their isotopic identity is maintained (Zieman et al., 1984). Atmospheric CO_2 , the source of carbon for plants, has a $\delta^{13}\text{C}$ value of about -7.6‰ (Francey et al., 1985), yet terrestrial plants that utilize the Calvin Cycle to fix carbon have an average $\delta^{13}\text{C}$ value of -27‰ . This degree of fractionation occurs because of the isotopic selectivity of the enzyme (ribulose-1,5-bisphosphate carboxylase) initially responsible for fixing atmospheric carbon dioxide (Estep et al., 1978) and to a minor extent because of fractionations associated with diffusion resistance (Farquhar and Richards, 1984). Carbon fixation in aquatic plants such as seagrasses, on the other hand, may be substantially affected by rate-limiting diffusion barriers that prevent isotopic selectivity and cause much smaller isotopic depletions relative to environmental CO_2 than in terrestrial plants (Fry and Sherr, 1989). The isotopic composition of the dissolved inorganic carbon (DIC) pool also affects the $\delta^{13}\text{C}$ value of aquatic plants since the DIC pool is their sole carbon source. The seagrass beds at the Bill Baggs site are well flushed by relatively clear water, but seagrass beds in Matheson Hammock Park are in shallow and relatively immobile water that may accumulate isotopically light CO_2 from the degradation of particulate organic material from the man-

Table 1. Average $\delta^{13}\text{C}$ values of seagrass and consumers at various locations (mean $\pm$ SE (n))

Location	$\delta^{13}\text{C}$ of seagrass	$\delta^{13}\text{C}$ of consumers
Matheson Hammock (mangrove forest)	—	$-24.2 \pm 1.4\text{‰}$ (14)
Matheson Hammock (mouth of effluent)	$-15.3 \pm 1.2\text{‰}$ (3)	$-19.6 \pm 1.0\text{‰}$ (24)
Matheson Hammock (away from effluent)	$-15.3 \pm 1.2\text{‰}$ (3)	$-18.6 \pm 1.5\text{‰}$ (11)
Bill Baggs (beach)	$-8.7 \pm 0.5\text{‰}$ (3)	$-15.4 \pm 1.0\text{‰}$ (8)

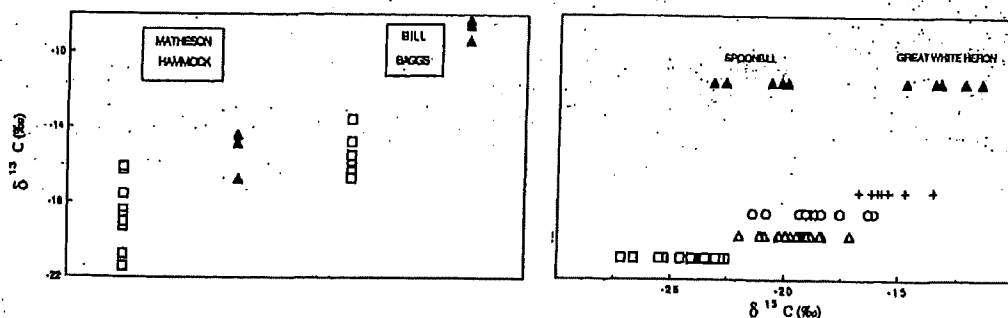


Figure 3 (left). $\delta^{13}\text{C}$ values of seagrasses ($\blacktriangle$) and heterotrophic organisms ($\square$) in Matheson Hammock Park and Bill Baggs Park.

Figure 4 (right). $\delta^{13}\text{C}$ values of organisms in mangroves and at several seagrass beds sampled in Matheson Hammock and Bill Baggs State Park as well as from several individuals from two bird species (Great White Heron and Roseate Spoonbill) collected in Florida Bay. Symbols are ($\square$) for organisms living within the mangrove forest, (Δ) organisms living in a seagrass bed at the mouth of the mangrove estuary, ($\circ$) organisms living in a seagrass bed within Biscayne Bay but away from the mouth of the mangrove estuary, (+) organism living in a seagrass bed at Bill Baggs without any terrestrial input, and ($\blacktriangle$) the two bird species.

grove tidal stream. Thus at Matheson Hammock Park an input of isotopically light respired CO_2 may be the cause of the lower $\delta^{13}\text{C}$ values in seagrass relative to those found at Bill Baggs. Zieman et al. (1984) observed similar variations between seagrasses in different localities in Florida Bay and proposed a similar cause for this variation.

As indicated in Figure 2b, there is a steady increase in $\delta^{13}\text{C}$ values of organisms proceeding along the transect from mangrove to seagrass at the Matheson Hammock site. Assuming that mangrove and seagrass detritus are the sole contributors of carbon to organisms at the site in front of the estuary, then the relative carbon contribution of each can be determined with a mass-balance equation: $\delta^{13}\text{C}_o = n\delta^{13}\text{C}_m + (1 - n)\delta^{13}\text{C}_s$, where $\delta^{13}\text{C}_o$, $\delta^{13}\text{C}_m$, and $\delta^{13}\text{C}_s$ are the $\delta^{13}\text{C}$ values for consumers, seagrass, and mangroves, respectively, and n is the proportion of carbon derived from mangrove detritus. For the Matheson Hammock Park site this equation becomes $-19.6\text{‰} = n(-27\text{‰}) + (1 - n)(-15.3\text{‰})$, and n equals 37%. Under this scenario, mangroves are providing 37% and seagrasses are supplying 63% of the carbon to organisms beyond the mouth of the stream in the seagrass bed. This scenario, however, many not be realistic. Results of carbon isotope analysis at the Bill Baggs site, which does not have a terrestrial input, indicate that there may be other marine sources with isotopically light carbon contributing to the food chain (Fig. 3). $\delta^{13}\text{C}$ values of seagrass and epiphytes at Bill Baggs were -8.7‰ and -12‰ , respectively, and yet $\delta^{13}\text{C}$ values of organisms living in the seagrass beds there averaged -15.4‰ . This discrepancy suggests that other marine sources, probably phytoplankton with an average $\delta^{13}\text{C}$ value of -20‰ (Peterson and Fry, 1987), are also important carbon sources. If the contribution by phytoplankton at the Matheson Hammock is as extensive as that observed in Bill Baggs, then the contribution of reduced carbon to organisms living in the seagrass beds by mangroves is much less than 37%. As has been observed in seagrass beds in other areas (Kitting et al., 1984), mangrove carbon may be further diluted by the extensive epiphytic growth in the seagrass beds at Matheson Hammock Park which has an average $\delta^{13}\text{C}$ value of -24‰ . Thus, we conclude that mangrove may be a major contributor of carbon on a very localized scale, such as within the riverine forest. Beyond the forest, seagrass and other

marine sources such as phytoplankton or seagrass epiphytes are the major suppliers of reduced carbon. Our conclusions about the localized contribution of reduced carbon by mangrove ecosystems are in agreement with those reported by Rodelli et al. (1984) in a study of Malaysian mangroves. Rodelli et al.'s (1984) results indicate that at distances as close as 2 km offshore, mangrove carbon is no longer an important contributor to the food chain and is replaced by phytoplankton.

The implications of these findings are also important for organisms at higher trophic levels. The same contrast between the two major sources of reduced carbon for the food chain has been previously observed in Florida Bay by Zieman et al. (1984). Thus isotopic ratios of organisms collected in the Florida Bay can be interpreted using our data. Analysis of stable carbon isotope ratios of muscle tissue of Great White Herons (*Ardea herodias*) collected in Florida Bay (Fig. 4) indicates that its diet is mainly dependent on seagrass organisms. The $\delta^{13}\text{C}$ values of its muscle tissue are only a few ‰ higher than small organisms living in a seagrass bed as observed in Bill Baggs. This small isotopic enrichment is probably caused by a trophic level effect (Mizutani and Wada, 1988). In contrast to the isotope ratios of the Great White Heron, flesh from Roseate Spoonbills (*Ajaia ajaja*) also collected in Florida Bay, which forage mostly in mangrove muds, have $\delta^{13}\text{C}$ values similar to those observed in organisms growing within the mangrove forest ecosystems.

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ADDRESS: *Department of Biology, University of Miami, Coral Gables, Florida 33124.*

The Contribution of Mangrove Swamps to Florida Fisheries

ERIC J. HEALD¹ AND WILLIAM E. ODUM²
*Rosenstiel School of Marine and Atmospheric Sciences
University of Miami
Miami, Florida*

INTRODUCTION

THE COASTAL MARGINS of southern Florida are characterized by a 700-square mile band of mangrove forests which received little public attention until recently. South Florida residents and visitors, while realizing that mangrove areas are good places in which to fish for snook, tarpon or mangrove snappers, have otherwise tended to dismiss them as mosquito-ridden wastelands, or, at best, as curiosities of dubious beauty. Land developers regard mangrove swamps as an impedence to the development of valuable waterfront properties, and the commercial fisherman rarely pays attention to mangroves unless he encounters an oyster-encrusted log in his net.

During the past decade there has been a growing recognition of the role played by south Florida's estuarine areas in the maintenance of commercial and sport fisheries, both in the immediate vicinity of land and further offshore in the Gulf of Mexico. Idyll (1965) points out that the pink shrimp, *Penaeus duorarum*, is dependent upon these mangrove-dominated estuaries as a nursery ground. One measure of the potential of such nursery areas is the volume of the commercial shrimp catches from the Dry Tortugas grounds. In 1965 its value exceeded 11 million pounds, worth over \$10 million (Anon., 1966). Most, if not all, of these shrimp grew up in the estuarine regions of Everglades National Park and adjacent areas.

Several other commercially valuable species including mullet, grey snapper, red drum and blue crab use these estuaries as a nursery area and feeding ground (Idyll et al., 1968). Tabb (1966) has shown that at least one species of economic and recreational value (the spotted seatrout) appears to be dependent upon the estuary during the greater part of its life cycle.

The primary food source for aquatic organisms in many shallow estuaries, particularly those at lower latitudes, is not phytoplankton, but vascular plant detritus (W. E. Odum, 1970a). Schelske and E. P. Odum (1962) demonstrated that the high productivity of the Georgia salt marshes is based mainly upon cordgrass, *Spartina alterniflora*. The system is characterized by a large standing crop of resistant vegetation which is not heavily grazed by terrestrial herbivores. The result is a continual production of large amounts of plant debris which is utilized in a detritus-based food web. In south Florida a similar situation exists with the red mangrove, *Rhizophora mangle*, and turtle grass, *Thalassia testudinum*, sharing the role of dominant primary producer and source of detritus.

¹Present address: United Fruit Co., Research Division, General Development Corporation Building, Miami, Florida.

²Present address: Department of Zoology, University of British Columbia, Vancouver, British Columbia.

In this paper we discuss the importance of mangrove swamps to the ecology of south Florida estuarine areas from the viewpoint of detritus production. Briefly, we describe the pathways by which the primary production of these plants is ultimately utilized by species of commercial value. The data upon which this discussion is based were gathered during 1967 and 1968 in the North River estuarine system of Everglades National Park, Florida. Information not presented in detail here can be found in Heald and Odum (1970) and Odum and Heald (1970a&b).

The North River arises as a poorly organized stream system near the edge of the sawgrass prairie and flows southwest into Whitewater Bay, about 14 km from its farthest traceable source. It drains an estimated 4,100 acres. The predominant drainage pattern is best described as a series of shallow, mud-lined ponds connected by meandering streams which eventually coalesce and enter the main river at various points.

The dominant cover vegetation is red mangrove, *Rhizophora mangle*, which borders all water areas and covers most of the land between them. In the upper reaches of the river the mangroves become progressively more restricted to its banks. Scattered marsh areas dominated by the black-rush, *Juncus roemerianus*, lie beyond these mangrove fringes.

The hydrography of the system is characterized by marked fluctuations of water level and salinity in response to the seasonal rainfall pattern of south Florida. In general the months of low rainfall are November to May when salinities rise to about 25 parts per thousand. The heavy summer rains normally begin in June, and freshwater conditions persist for most of summer and early fall.

Aquatic primary producers are nowhere abundant. Phytoplankton volumes were low throughout the study period, and benthic algae were not abundant. The bladderwort, *Utricularia lutea*, grows rapidly in shallow pools during the summer, but its contribution to the whole river system is very small.

OBJECTIVES AND PROCEDURES

Our investigations of the mangrove ecosystem determined the amount and ultimate fate of mangrove litter produced yearly. Rates of breakdown of litter under different environmental conditions were determined. Protein and caloric content during litter degradation was monitored. Measurements were made of the amount of mangrove detritus present in the river. A study of the food habits of all species of fish and invertebrates in the system was performed.

Litter production was measured monthly from catchment devices placed around selected trees. Details of the method are given by Heald (1969a). Breakdown rates of leaves were obtained from field experiments in which known weights of newly fallen leaves, placed in nylon mesh bags, were retrieved and weighed at monthly intervals. Chemical analyses included bomb calorimetry, micro-kjeldahl nitrogen determinations and fat extractions.

For the purpose of trophic studies, over 6,000 individuals representing approximately 90 species of fish, crustaceans, bivalves and insect larvae were captured by a variety of methods, and an analysis of stomach contents was made with conventional, phase and fluorescence microscopy. Capture methods and techniques of analysis are detailed by W. E. Odum (1970b).

From the data thus obtained we were able to reconstruct the biological

pathways of energy flow in the community and assess the value of mangrove material to the ecosystem.

THE PROCESS OF DETRITUS PRODUCTION

Production of mangrove leaf material in the North River estuary exceeds 2.9 dry tons per acre per year or 800 dry grams per square meter per year. This means that about 10,000 tons (dry wt.) of leaves are shed each year by the mangroves of the North River drainage. Lesser amounts of litter are contributed by twigs, bark and leaf scales.

Less than 5% of the leaf material is consumed on the tree by terrestrial herbivores; most of it falls from the tree. Very little of this fallen leaf material remains permanently in the system. About half is consumed by bacteria, fungi and detritus feeders. Most of the remainder is eventually flushed out of the river system into nearby shallow coastal embayments where it is available as food to detritus consumers in those ecosystems. Only 1 or 2% in the form of peat is actually lost to the animal communities.

The rate at which mangrove leaf material enters the aquatic food chains depends largely upon where it falls. Leaves falling directly into the water soon acquire a flora of bacteria and fungi which utilize the leaf material as a physical substrate and as a nutrient source. After 2 or 3 months the breakdown of the leaf is further speeded by amphipods and the crab *Rhithropanopeus harrisi* which shred the leaf into smaller fragments. Within 6 months of leaf fall, over 30% of the leaf material has been removed by scavenging organisms. Three more months are sufficient for the leaf to be reduced to fragments no larger than 1 cm in width. This gradual degradation of the leaf is accompanied by an increase in the caloric content and the relative percentage of protein of the leaf particle, due to its covering of microorganisms. For this reason the particle becomes an increasingly valuable food source as it diminishes in size.

Leaves which fall on river banks and other elevated areas are initially unavailable and break down slowly. When subsequently washed into the river during the rise in water level which accompanies the onset of the summer rainy season, they break down rapidly and are utilized by detritus feeders. Under present conditions in the North River basin, almost all mangrove leaf litter is broken down and fragmented within a year. If flood control measures or other water diversions interfere with the seasonal influx of fresh water much more detrital material would accumulate within the mangrove forest. This in turn would lead to an increased deposition of forest soil culminating in conditions unfavorable to the mangroves and favorable to a willow and buttonwood community.

DETRITUS PRODUCTION: THE PRINCIPAL PATH OF ENERGY FLOW

The detailed structure of the important food chains in the North River system have been investigated (Odum and Heald, 1970a) and analyzed (Odum and Heald, 1970b) elsewhere. In summary, the animals of this estuarine system are dependent upon an input of energy in the form of vascular plant detritus originating from mangrove leaves. The production of phytoplankton and benthic algae provides a limited amount of energy for the primary consumers, but is of limited importance compared to the production of vascular plant detritus.

The first link in the food web is provided by bacteria and fungi which convert the relatively undigestible lignin and cellulose of vascular plant tissue into a protein source which can be digested by the organisms of the second link in the food web. This next link is composed of opportunistic omnivorous crustaceans such as amphipods, mysids, harpacticoid copepods, caridean and penaeid shrimp along with chironomid midge larvae and a few omnivorous fishes such as the striped mullet, *Mugil cephalus*; the sheepshead minnow, *Cyprinodon variegatus*; and the molly, *Poecilia latipinna*. All of these ingest large quantities of detritus particles with their loads of digestible microorganisms. These particles pass through the digestive tract essentially unaltered except for mechanical grinding and the partial removal of absorbed microorganisms. Once the particle is excreted as fecal material, the entire process (excreted particle→ colonized by fungi and bacteria→ ingested by detritus feeder→ fungi and bacteria digested off particle by detritus feeder→ excreted particle) commences again and may be repeated a number of times. A particle may be ingested by several different detritus feeders before it is completely broken down. Even when the particle becomes very small (less than 5 or 10 microns) it may combine with other small particles to form conglomerates which continue in the cycle.

This group of detritus consumers also ingests other materials which include live benthic microalgae, animal material and fine inorganic particles which probably contain absorbed dissolved organic substances. The live microalgae, although usually present in relatively small volumes, appear to be necessary for adequate growth.

In the North River the critical detritus feeding link is composed of relatively few species: one xanthid crab, *Rhithropanopeus harrisi*; four mysids; four amphipods; one penaeid shrimp; two caridean shrimps; one snapping shrimp; three small forage fishes; and an undetermined number of harpacticoid copepod, isopod and chironomid midge larval species. Feeding on these detritus consumers are a large group of fishes including all of the juvenile game fishes.

THE PRODUCTION OF GAME FISH AND FOOD FISH

A number of gamefish depend upon the Everglades mangrove estuary for varying periods of their juvenile lives. Tarpon, *Megalops atlanticus*; snook, *Centropomus undecimalis*; and ladyfish, *Elops saurus*, utilize the mangrove belt from the time they reach the estuary as post-larvae. Grey snapper, *Lutjanus griseus*; sheepshead, *Archosargus probatocephalus*; spotted seatrout, *Cynoscion nebulosus*; and red drum, *Sciaenops ocellata*, spend the first few weeks of their lives in the grass beds of Florida and Whitewater Bays and then move into the mangrove habitat for the next several years. Of this group, the grey snapper is the most dependent upon the mangrove environment. Finally, there are other gamefish which are found in the mangrove zone and other areas: the crevalle jack, *Caranx hippos*; the gafftopsail catfish, *Bagre marinus*; and the jewfish, *Epinephelus itajara*.

All of these gamefish feed on detritus feeders. The young fish eat amphipods, mysids and chironomid larvae. As the fish grow larger they consume caridean and penaeid shrimp, snapping shrimp, crabs and detritus-consuming fish along with forage fishes such as the tidewater silver sides, *Menidia berylina*; the silver jenny, *Eucinostomus gula*; and the rainwater killifish, *Lucania parva*, which in turn feed on detritus feeders.

The value of these nursery areas to the commercial shrimp fishery has already been referred to. It need only be added that shrimp are opportunistic feeders, incorporating large amounts of detritus in their diet. Other commercially valuable species, such as spotted seatrout, red drum, striped mullet, *Mugil cephalus*, and blue crab, *Callinectes sapidus*, are equally dependent upon the estuarine detritus food-chain system. Catches of these species from a single Bureau of Commercial Fisheries statistical area extending from Naples to western Florida Bay can be cited as an indication of potential productivity. In 1965 this area yielded over 2.5 million lbs. of striped mullet, 164,000 lbs. of spotted seatrout, 30,000 lbs. of blue crabs and almost 40,000 lbs. of red drum (Heald, 1969b).

From the foregoing discussion it should be apparent that the production of game and food fish from mangrove systems such as the North River is directly linked to the production of the detritus feeders and ultimately to the production of vascular plant detritus in the form of decaying mangrove leaves, roots, bark and wood. If mangrove estuarine systems are destroyed, a valuable protective habitat for juvenile fishes will be lost. More importantly it will cut off the input of mangrove organic material which is largely contributed in leaf fall. The leaf fall supports the detritus based food webs, a large population of detritus feeders and their predators. The production of phytoplankton and benthic and epiphytic algae in such an area is so much less than the production of detritus of a mangrove origin that the yield of gamefish must decline in proportion to mangrove destruction.

The situation is somewhat different in more open areas such as Florida and Biscayne Bays where there is a significant detritus production from seagrasses (discussed by Wood et al., 1970) and benthic macroalgae. Nevertheless, the wholesale removal of mangrove swamps, even from these more diverse areas, will result in a reduced production of detritus feeders and the commercial and sports fish which consume them.

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Attachment E

Storm Surge Reduction by Mangroves

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Storm Surge Reduction by Mangroves



Anna McIvor, Tom Spencer, Iris Möller
and Mark Spalding

Natural Coastal Protection Series: Report 2
Cambridge Coastal Research Unit Working Paper 41

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Authors:

Anna L. McIvor, The Nature Conservancy, Cambridge, UK and Cambridge Coastal Research Unit, Department of Geography, University of Cambridge, UK. Corresponding author: anna.mcivor@tnc.org
Tom Spencer, Cambridge Coastal Research Unit, Department of Geography, University of Cambridge, UK.
Iris Möller, Cambridge Coastal Research Unit, Department of Geography, University of Cambridge, UK.
Mark Spalding, The Nature Conservancy, Cambridge, UK and Department of Zoology, University of Cambridge, UK.

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The Nature Conservancy's **Natural Coastal Protection** project is a collaborative work to review the growing body of evidence as to how, and under what conditions, natural ecosystems can and should be worked into strategies for coastal protection. This work falls within the Coastal Resilience Program, which includes a broad array of research and action bringing together science and policy to enable the development of resilient coasts, where nature forms part of the solution.

The **Mangrove Capital** project aims to bring the values of mangroves to the fore and to provide the knowledge and tools necessary for the improved management of mangrove forests. The project advances the improved management and restoration of mangrove forests as an effective strategy for ensuring resilience against natural hazards and as a basis for economic prosperity in coastal areas. The project is a partnership between Wetlands International, The Nature Conservancy, Deltares, Wageningen University and several Indonesian partner organisations.

About The Nature Conservancy

The mission of The Nature Conservancy is to conserve the lands and waters upon which all life depends. For general information, visit: www.nature.org. For more information about the Natural Coastal Protection project, visit: www.naturalcoastalprotection.org and www.coastalresilience.org.



About The Cambridge Coastal Research Unit

The Cambridge Coastal Research Unit is based in the Department of Geography in the University of Cambridge. It aims to provide the high quality scientific research to underpin sustainable coastal management. For more information, visit: <http://www.ccru.geog.cam.ac.uk/> and <http://www.geog.cam.ac.uk>.



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The mission of Wetlands International is to sustain and restore wetlands, their resources and biodiversity. Wetlands International is the only global non-profit organisation dedicated to the conservation and restoration of wetlands. It works through a network of 18 offices and many partners and experts to achieve its goals. For more information, visit <http://www.wetlands.org/>.



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Executive Summary

Storm surges occur when high winds and low atmospheric pressure raise water levels at the coast, causing sea water to surge onto the land. They are a major threat to low-lying coastal areas and their inhabitants. The largest storm surges are caused by tropical cyclones (also called hurricanes and typhoons in different regions); peak water levels can exceed 7 m in height, and can result in extensive flooding, loss of life and damage to property. Global climate change may result in increased storm surge flooding in some areas, through intensification of the cyclones driving the storm surges and as a result of sea level rise.

Mangroves can reduce storm surge water levels by slowing the flow of water and reducing surface waves. Therefore mangroves can potentially play a role in coastal defence and disaster risk reduction, either alone or alongside other risk reduction measures such as early warning systems and engineered coastal defence structures (e.g. sea walls).

Measured rates of storm surge reduction through mangroves range from 5 to 50 centimetres water level reduction per kilometre of mangrove width. In addition, surface wind waves are expected to be reduced by more than 75% over one kilometre of mangroves.

Few data are available on surge reduction rates through mangroves because of the difficulties associated with measuring water levels during storm surges. All data currently available are from the south-eastern United States, where networks of recorders have been placed in wetland areas. Numerical models and simulations, validated using this data, provide the only means of exploring the importance of different factors in reducing storm surge heights.

The numerical model of Zhang *et al.* (2012; *Estuarine, Coastal and Shelf Science* 102: 11-23) suggests that mangroves are more effective at reducing the water levels of fast moving surges than those of slow moving surges. The model also indicates that water level reduction through mangroves is non-linear, with the greatest reduction in surge height occurring near the seaward edge of the mangroves. Seaward of mangroves, a bulge of water can form as the water piles up in front of the mangroves; this can increase storm surge levels in this area.

Several topics relating to storm surge reduction by mangroves are yet to be explored, such as the effect of mangrove density, species composition and vegetative morphology. Dense mangrove forests, including species with aerial roots, are expected to increase storm surge reduction rates.

By reducing water levels and wave energy, mangroves can save lives and reduce storm-surge related damage to infrastructure: during a typhoon in north-east India, mangroves reduced the number of lives lost, as well as reducing damage to houses, crops and possibly coastal defence structures. Mangroves can also help people recover after coastal disasters by providing firewood, building materials and food sources (e.g. fish and shellfish that live among mangrove aerial roots).

Cyclones and storm surges also impact mangroves themselves; some trees may be defoliated or uprooted. Extreme events with very high water levels and wind speeds may severely damage or destroy mangrove areas, rendering them less effective at reducing surge heights. Natural recovery can take many years to decades; restoration projects may speed up recovery.

Further data on storm surge reduction by mangroves and further refinements to numerical models and simulations will improve our ability to understand and quantify the coastal defence services provided by mangrove forests against storm surges. Such information is needed to ensure that the coastal defence functions of mangroves are utilised appropriately, either alone or in combination with other measures, to reduce risk to people and infrastructure from storm surges.

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Introduction to the Natural Coastal Protection Series

The world's coastal margins are among the most densely populated and intensively used places on Earth. Coastal populations are growing rapidly, as is associated infrastructure, industry and agriculture. These populations and coastal lands can be at risk from natural hazards such as waves, storms and tsunamis. The numbers of people at risk are increasing with the expansion of human populations, and the risks will likely be exacerbated by the effects of climate change and associated sea level rise. At the same time, coastal ecosystems are also impacted as populations expand, reducing ecosystem resilience and their ability to provide ecosystem services such as coastal defence.

In the face of growing risks and vulnerability, increasing attention is being given to adaptation and disaster risk reduction in the coastal zone. An array of measures can help reduce the vulnerability of coastal populations, including: changes to planning and development patterns in near-shore zones; development of early warning systems and hazard response strategies; and coastal defence measures that maintain, enhance or develop structures or features that reduce the risk of impacts on coastal populations and lands. Against this background there have been growing calls for the consideration of the role of natural coastal ecosystems in coastal defence. Claims have been made that some coastal ecosystems, including mangrove forests, coral reefs and salt marshes, can help to reduce the risk associated with some coastal hazards. Such ecosystems also provide a host of associated ecosystem services which may be lost if natural systems are degraded or converted to alternative land uses.

If a case is to be made that ecosystems form a realistic part of coastal adaptation, however, it will depend on having a solid science foundation, and on the ability to predict when, and under what conditions, the ecosystem may be able to function effectively as a defence against coastal hazards. The Natural Coastal Protection Series are a series of technical reports investigating the role of ecosystems in coastal defence. The reports aim to summarize current information relating to this topic in a way that is accessible to a broad spectrum of people, including scientists, coastal engineers, decision makers, site managers and conservation advocates. The reports also aim to introduce the topic such that non-experts are able to understand and assess the current state of knowledge.

1. Introduction

This report explores the capacity of mangroves to reduce storm surge water levels and inundation, and hence to reduce the impact of storm surges on coastal populations and infrastructure.

Storm surges are movements of sea water onto land, caused by high winds and low atmospheric pressure raising water levels at the coast, so that the sea literally “surges” onto the land. Storm surges can be highly destructive, sometimes causing the loss of thousands of lives and extensive damage to property (Table 1). An extreme example was the 7 m high storm surge that hit the south-west part of Bangladesh in 1991; the storm surge was caused by a cyclone with a maximum recorded wind speed of 225 km/hr, and 138,000 people lost their lives (Bern *et al.*, 1993; Matsuda, 1993). Storm surges are a major risk in many low-lying coastal areas (Storch and Woth, 2008), and in terms of loss of life and property, they are probably the most destructive natural hazards of geophysical origin (Flather, 2001). Section 1.1 and Box 1 describe storm surges in more detail.

Global climate change may result in increased storm surge flooding in some areas, both through intensification of the cyclones driving the storm surges, and from underlying sea level rise (Mousavi *et al.*, 2011). The overall extent of regions susceptible to tropical cyclones and their associated storm surges may also increase geographically (Flather, 2001). In addition, the increasing numbers of people living in low-lying coastal regions result in greater risks from coastal flooding, as more people live in exposed areas and more infrastructure is built there to accommodate them (McGranahan *et al.*, 2007; Kron, 2008).

Table 1. Examples of some major storm surges over the past century. Data on these cyclones and associated storm surges sometimes vary between sources, and this is probably related to actual uncertainty e.g. in numbers of deaths or value of losses, and to different locations where wind speed and storm surge water levels were measured. *PWL: Peak water level (see Box 2); SSHWS: Saffir-Simpson Hurricane Wind Scale (see Box 3 below). Additional data from Blake (2011) and National Hurricane Center (2012a).

Name of cyclone	Year	Location	Surge PWL* (m)	Wind speeds (km/hr)	SSHWS* category	Deaths	Losses (US \$)	Sources
Extratropical storms – southern North Sea								
North Sea Flood	1953	Southern North Sea	3.0	75		1,783		Garrison, 1999; Pugh, 1987
Tropical cyclones – Atlantic Ocean								
Hurricane Carol	1954	Rhode Island	5	185	3	60	41 million	Garrison, 1999
Hurricane Camille	1969	Mississippi coast	7	305	5	256	1.4 billion	Garrison, 1999; Pugh, 2004
Hurricane Andrew	1992	South Florida and Louisiana	5.1	280	5	26	30 billion	Garrison, 1999; Pugh, 2004; NHC 2012b
Hurricane Katrina	2004	Gulf coast	8.5	190	3	1200	125 billion	Kron, 2008, NHC; 2012b
Tropical cyclones – Indian Ocean								
Cyclone Bhola	1970	Bangladesh	10 - 12	222	3	300,000		Garrison, 1999
Cyclonic storm BOB 01	1991	Bangladesh	5 - 8	225	4	138,000		Matsuda, 1993; Bern <i>et al.</i> , 1993
Cyclone 05B	1999	Orissa, India	7 to 8	250	4	10,000		Pugh, 2004
Cyclone Nargis	2008	Myanmar	5	210	4	138,000	10 billion	Fritz <i>et al.</i> 2009

In areas that are particularly susceptible to storm surges, a variety of measures are usually in place to protect people and property. Structural measures include sea walls and levees, and these are used alongside early warning systems, evacuation plans and refuge. Recently, there has been increasing interest in the use of ecosystems as a cost effective form of coastal defence against hazards such as storm surges; for example, mangrove restoration projects often cite coastal protection as one of the benefits that the restored mangroves will provide (Tri *et al.*, 1998; Jegillos *et al.*, 2005; Primavera and Esteban, 2008; Erwin, 2009; Powell *et al.*, 2011). While it is clear that the dense vegetation of a mangrove forest can significantly reduce wind and swell waves over a few hundred metres (McIvor *et al.*, 2012), there is less evidence supporting the ability of mangroves to reduce storm surges. Such information is urgently needed for people to assess when and where mangrove forests can form part of a coastal defence strategy against surges.

Mangroves are coastal forests found in tropical and sub-tropical regions, many of them in areas subject to cyclones, hurricanes and typhoons and their associated storm surges. Local people often believe that mangroves can provide protection from storm surges (Walters, 2003 and 2004; Badola and Hussain, 2005; Walton *et al.*, 2006; Walters *et al.*, 2008; Warren-Rhodes *et al.*, 2011). This report reviews available information about the capacity of mangroves to reduce storm surges, in order to inform decision makers, planners and coastal engineers about the potential role that mangroves can play in coastal defence against storm surges. Where mangroves can contribute important coastal defence functions, there may be considerable benefits to ensuring their inclusion in coastal planning and adaptation strategies. Such coastal defence functions are in addition to the multitude of other benefits provided by mangroves, such as firewood, materials for building, and food in the form of shellfish and fish that live among the mangrove roots (Barbier *et al.*, 2011).

The first section of this report provides some basic information about storm surges and the factors affecting surge levels and surge reduction. Section 2 explores the evidence for mangroves reducing storm surge water levels, followed by Section 3, which focuses on the attributes of mangroves that affect the level of storm surge reduction. Section 4 reviews the effect of mangroves on surface wind speeds, which affect wave generation and hence surge water levels. Section 5 reviews the literature on the ability of mangroves to reduce loss and damage caused by storm surges. Section 6 then explores how mangroves are themselves affected by storm surges and cyclones.

1.1 Storm surges

Storm surges may be defined as abnormally high sea water levels in coastal areas caused by a short-lived atmospheric disturbance such as a hurricane or storm. Box 1 compares the characteristics of storm surges with tsunamis, tides and other sea waves.

Storm surges are created by tropical cyclones (also called hurricanes and typhoons in different geographical regions), extratropical storms (mid-latitude frontal storms or mid-latitude depressions) and other types of atmospheric disturbance such as polar lows (Storch and Woth, 2008); the different characteristics of tropical cyclones and extratropical storms are shown in Table 2. Tropical cyclones are usually more intense but of shorter duration and less extensive spatially. They typically originate in the tropics, but can travel to sub-tropical latitudes. The following descriptions and explanations relate primarily to tropical cyclones and the storm surges they produce, because tropical cyclones are the dominant atmospheric disturbance in areas where mangroves are found.

Box 1. Comparing storm surges, tsunamis, tides and other water waves

Table 1.1 compares the characteristics of storm surges with other types of sea waves. Storm surges differ from tides and tsunamis in that the increased water levels during a storm surge are caused by atmospheric disturbances, as opposed to astronomical forces, which cause lunar or solar tides, or crustal disturbances, i.e. earthquakes, volcanic eruptions or landslides, which can cause tsunamis (Groen and Groves, 1962). Storm surges interact with astronomical tides, and are at their most dangerous if they coincide with a high spring tide, when even a modest surge can create a significant flooding hazard (see Section 1.1.1).

Storm surges act like very high tides, except that the rise in water level is unpredictable and faster than during normal tidal changes in water level (Garrison, 1999; Krauss, 2009). Storm surges differ from tsunamis in that tsunamis may affect much larger areas of coasts, usually consist of several waves, come onto land much more rapidly than storm surges, and also flow off the land more rapidly. Storm surges and tsunamis can result in similar depths of flooding and inundation extents.

Table 1.1. Different types of waves, the physical mechanisms causing them and their wave periods (the time between two successive peaks passing a given point; see Fig. 1). *For storm surges, this time period refers to the duration of the change in water level experienced at the coast. (From Massel, 1996; Pugh, 1987; Groen and Groves, 1962).

Wave type	Physical mechanism	Wave period
Wind waves	Wind shear, gravity	< 15 s
Swell waves	Wind waves with a longer period and wavelength that were created far from the shore. Shorter period wind waves travel more slowly, are dissipated more quickly and some of their energy is subsumed by the longer waves, resulting in only the longer waves reaching distant shores.	< 30 s
Tsunami	Earthquakes, volcanic eruptions, landslides, submarine slumping	10 min – 2 hours
Tides	Gravitational action of the moon and sun, earth's rotation	12 – 24 hours
Storm surges	Wind stresses and atmospheric pressure variation in combination with local bathymetry and geomorphology (occur during storms, hurricanes, cyclones, typhoons)	1 hour to 4 days*

Storm surges can temporarily increase coastal sea levels by several metres (Table 1 and 2). Tropical surges are relatively short-lived phenomena, with raised water levels usually lasting for less than 12 hours (Groen and Groves, 1962); their interaction with tides is described in the next section. The raised water levels flood coastal areas, and this can cause loss of life and property and damage to infrastructure and agriculture (Table 1). Surges can be particularly destructive on micro-tidal, low wave energy coasts, where high water levels are extremely rare and the design of local infrastructure may not take such events into consideration.

Table 2. Comparison of storm surges caused by tropical cyclones and extratropical storms (adapted from Storch and Woth, 2008)

Parameter	Tropical cyclone	Extratropical storm
Spatial scale of storm	500 ± 200 km	1000 ± 500 km
Surge height	Larger; extreme surge heights between 5 and 12 m (Table 1)	Smaller; extreme surge heights between 2 and 4 m
Surge duration	Several hours, up to half a day	2-5 days
Length of coastline affected by the surge	Usually < 200 km	Several hundred kilometres

1.1.1 Timing of surges relative to astronomical tides

Storm surges are particularly dangerous when they coincide with high spring tides. In areas with large tidal ranges, the timing of the surge relative to the tidal level is critical, and can make the difference between a storm surge having little effect and causing major flooding (Flather, 2001). Figure 1 shows the changes in predicted tidal level, storm surge, and observed water level (storm tide) during a storm surge in Bangladesh in 1970. The highest surge height occurred slightly after the predicted high tide; if the two had occurred simultaneously, the peak water level (highest storm tide) would have been higher than observed.

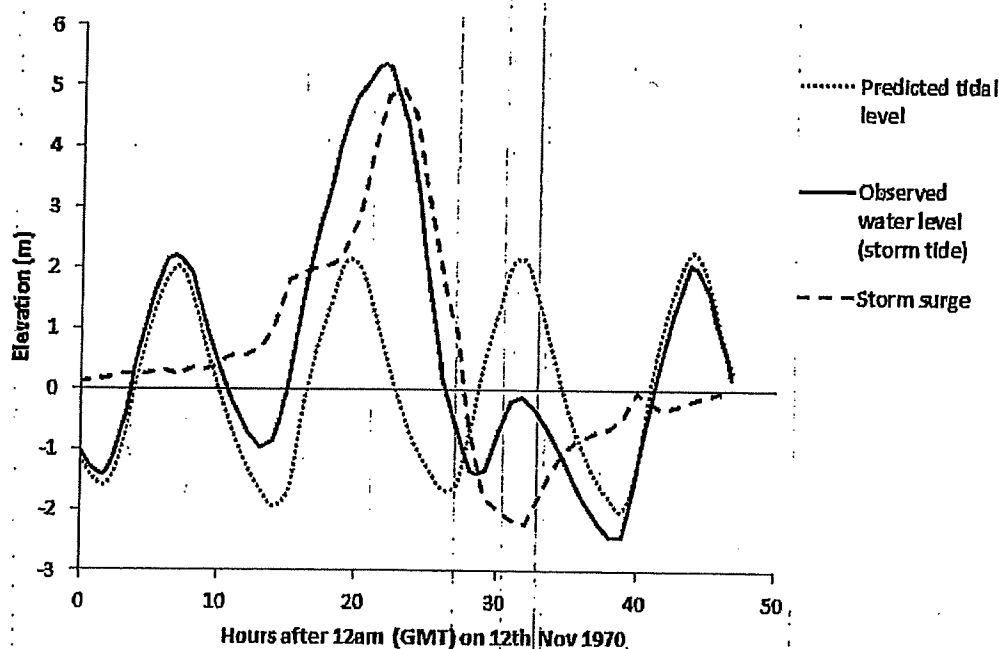


Figure 1. The observed water level, predicted tidal level and storm surge residual at Sandwip Island, Bangladesh, during the storm surge of 12-13 November 1970. The peak water level was 5.4 m at this location. Adapted from Flather (2001).

1.1.2 Where do tropical storm surges occur?

Figure 2 shows the storm tracks of tropical cyclones between 1985 and 2005, illustrating how tropical cyclones, and the storm surges they create, are most common on the eastern shores of North America, South East Asia and southern Africa, the west coast of Mexico and Central America, within the Bay of Bengal and along the northern coast of Australia. Very few cyclones or storm surges occur along the equator, around the coast of South America or on the west coast of Africa. Tropical cyclones only form where sea surface temperatures exceed 26.5 °C (i.e. they do not form in the temperate zones); their formation is dependent on the local vertical component of the Earth's rotation (the Coriolis effect), and this explains why they do not form within 5° of the equator (Flather, 2001). Atmospheric disturbances that form over the warm tropical oceans are generally carried westward by the easterly trade winds (Ahrens, 2007), resulting in hurricanes occurring more frequently on the eastern margins of land masses.

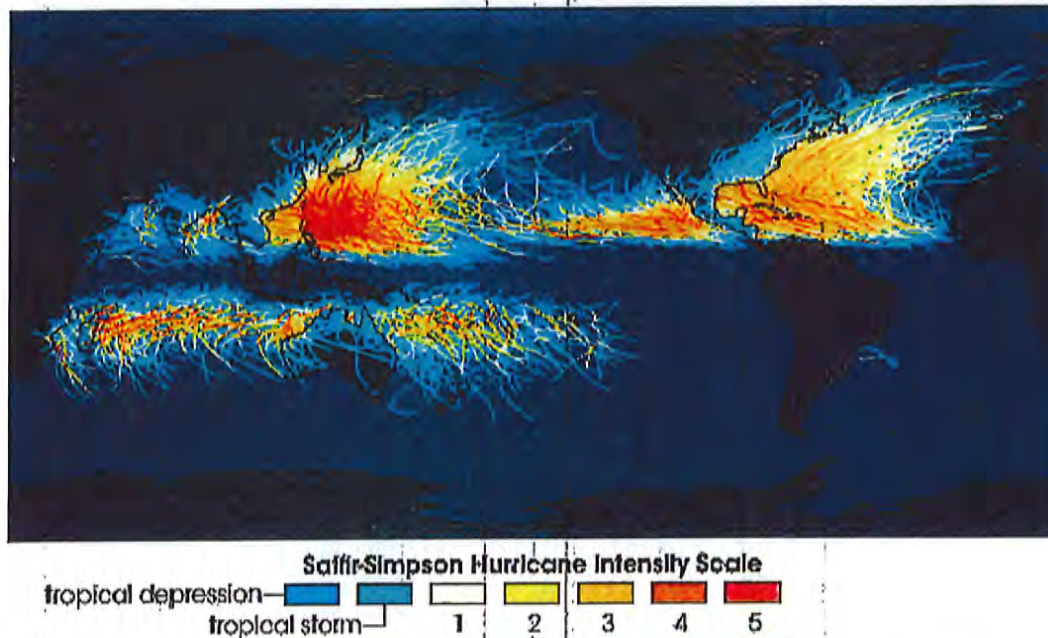


Figure 2. The tracks of tropical cyclones that formed between 1985 and 2005. The colours represent the strength of the cyclone according to the Saffir-Simpson Hurricane Wind Scale (see Box 3 for a description of this scale). (Image by Robert A. Rohde, Global Warming Art, NASA Earth Observatory; <http://earthobservatory.nasa.gov/IOTD/view.php?id=7079>.)

1.1.3 Characteristics of storm surges

Storm surges are measured in a number of different ways, and these ways are summarized in Box 2. The destructiveness of a storm surge depends on:

- the inundation extent, i.e. the area that is flooded;
- the depth of flooding and the peak water level, i.e. the maximum depth of the water;
- the duration of inundation (this may be a particular issue for agricultural areas, where crops may survive brief floods, as described in Badola and Hussain, 2005);
- the height of the waves riding on top of the storm surge, which may cause more severe damage to structures;

Box 2: Measures of storm surges

Various characteristics of storm surges are measured in order to describe the surge:

- **Storm surge** (also called **storm surge height** or **amplitude**) is the difference between observed water levels during the surge and predicted water levels caused by astronomical tides (Box 1 and Section 1.1.1). The precise technical term for this is the **non-tidal residual** or **surge residual** (Flather, 2001; Pugh, 2004); throughout this report we use the term “storm surge height”. Calculating the storm surge residual from the observed water level (storm tide; see below) and predicted tidal level is complicated by the fact that the two interact: for example, the timing of the astronomical high tide may shift because the tidal wave is travelling through deeper water (Groen and Groves, 1962). Therefore it is not possible to simply subtract the predicted tidal level from the observed water level to estimate the surge residual. However in many cases simple subtraction can provide an adequate estimate of the surge residual. **Skew surge** is also used to refer to the difference between the highest observed water level and the predicted water level of the nearest high tide (de Vries *et al.*, 1995).
- **Storm tide** refers to the total water depth above a reference level, usually mean sea level. It includes both the storm surge and the tidal level. For example, a 2 m storm surge on top of a 0.6 m tide produces a 2.6 m storm tide (National Hurricane Center, 2012b; Figure 1.1).
- **Peak water level** is usually used to refer to the highest water level experienced during a storm surge in a particular area, i.e. the largest storm tide recorded during a particular storm surge in that area. It usually refers to the **still water level** i.e. the water level without the effect of waves. Still water levels can be difficult to measure; an example of a still water level would be the height that water reached within a building where waves did not enter.
- **Inundation extent** refers to the area of land flooded during the storm surge. This can often be estimated from lines of debris made up of grass, seeds, and rubbish (National Hurricane Centre, 2012b). The **flooded volume** is the volume of water that floods onto land. As flooded volume increases, so does inundation extent; the relationship between the two will depend on local topography.

For more information about how these are measured in the field, see the National Hurricane Centre (2012b) document “Introduction to Storm Surge”.

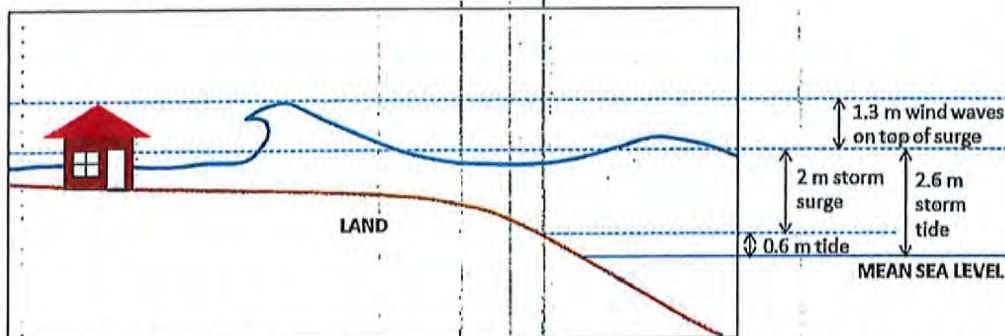


Figure 1.1. Schematic diagram of a storm surge, showing Mean Sea Level (MSL), the predicted high tide (0.6 m), the storm surge residual (2 m), the storm tide (2.6 m) and the wind waves on top of the surge (an additional 1.3 m). (Figure adapted from a diagram in National Hurricane Centre (2012b) document, “Introduction to Storm Surge”.)

- and the strength of currents that are generated within the water body, which can cause coastal erosion (Flather, 2001).

Tropical storm surges are usually accompanied by very high winds and heavy rainfall which may also cause extensive damage.

Other characteristics of storm surges include the speed of travel of the surge, which is measured as the velocity of travel of the peak water level, and the storm track, i.e. the location and direction of travel, which interact with the local bathymetry, topography and geomorphology to influence the peak water level and inundation extent (described in more detail in Section 1.3).

While tropical cyclones are routinely assigned a category based on their maximum sustained wind speeds (Box 3), storm surges are quantified using measures such as storm surge height, storm tide, peak water level, and inundation extent, as described in Box 2. Storm surge height is only partially related to the category of a cyclone, as many other factors also affect it, as described in Box 3 and Sections 1.2 and 1.3 below.

Box 3. The relationship between hurricane category and storm surge

Hurricanes in North America are routinely assigned a category according to the Saffir-Simpson Hurricane Wind Scale (Simpson, 1971, Table 3.1). Earlier versions of the scale included central atmospheric pressure and storm surge height as components of the categories (Simpson, 1971), but the scale has recently been revised such that the categories are based on maximum sustained wind speeds only (Schott *et al.*, 2012). This is because the inclusion of storm surge height within the categories can be misleading, as even a low category storm can produce a large surge (Table 3.1). This is because various factors other than wind speed affect the surge produced, notably hurricane areal extent (i.e. the geographical extent of hurricane-force winds), local bathymetry (i.e. the depth of near-shore areas), local topography, the hurricane's forward speed, and the angle at which the hurricane approaches the coast (Schott *et al.*, 2012). These factors are discussed in more detail in Section 1.3 below. Alternative classifications are often used for typhoons and cyclones in India, Japan, Australia and the Southwest Indian Ocean; these have different names for the categories, boundaries between categories, and different ways of measuring maximum wind speeds.

Table 3.1. The Saffir-Simpson Hurricane Wind Scale categories and wind speeds, with examples of hurricanes in each category, showing the peak water levels of the surges.

Saffir-Simpson Scale		Example	
Category	Sustained wind speed (km/hr)	Name of hurricane	Surge height (m)
1	119-153	Hurricane Irene (2011)	3.4
2	154-177	Hurricane Ike (2008)	6.1
3	178-208	Hurricane Katrina (2005)	8.5
4	209-251	Hurricane Charley (2004)	2.4
5	252+	Hurricane Camille (1969)	6.9

1.2 Forces creating tropical storm surges

Several forces contribute to creating a storm surge, as shown in Figures 3 and 4. For tropical storm surges, the largest forcing is created by the cyclonic winds. Wave set-up can also contribute a large part of the surge level. The rise in water level due to the low atmospheric pressure is relatively small by comparison. The different forcings are described in more detail below. (The following descriptions are simplified and omit much of the complexity of storm surge dynamics; for more detailed descriptions of storm surge forcings, see Flather (2001), Resio and Westerink (2008) and other references included in the text below.)

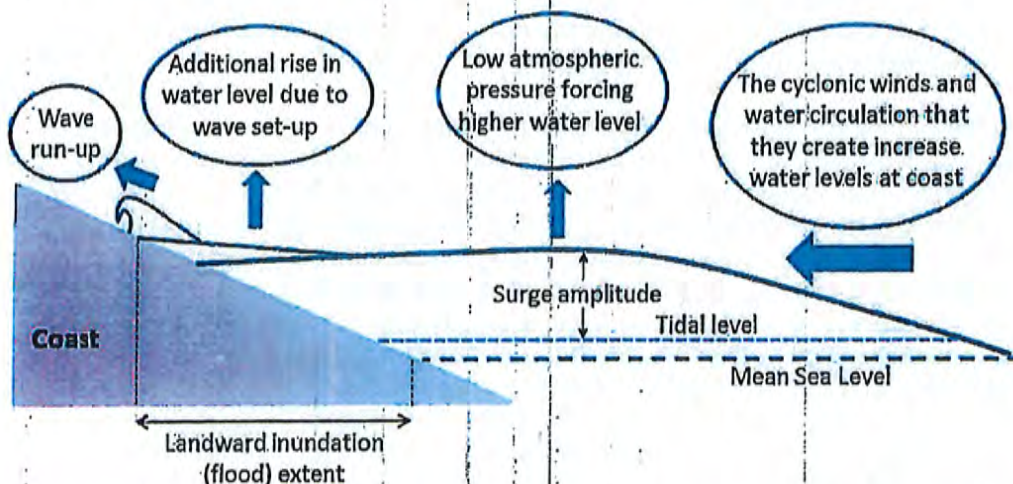


Figure 3. Schematic diagram showing how tropical storm surges are created, with the different forces involved.

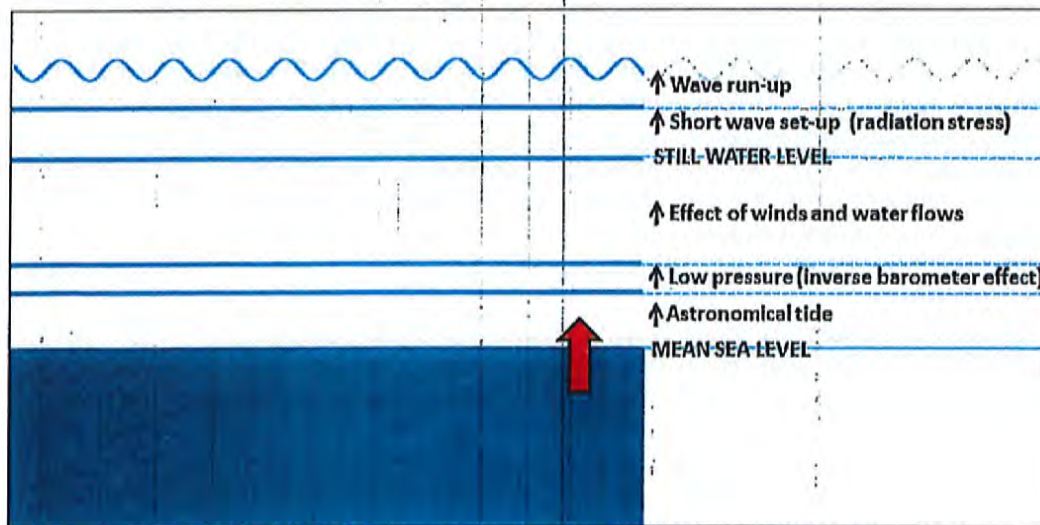


Figure 4. Schematic diagram of the relative contributions of the different forcings that produce a storm surge, approximately showing their relative magnitude, although this will vary greatly with each storm surge (for example, sometimes astronomical tide will be below mean sea level; the relative importance of short wave set-up is still under discussion, see text). (Adapted from Graber *et al.*, 2006).

As described in Section 1.1.1, storm surges occur on top of **astronomical tides**, and the tidal level can have a large influence on peak water levels in areas with large tidal ranges. Astronomical tides are caused by the gravitational forces from the moon and sun, and tidal ranges can be greater than 6 m in some areas (Woodroffe, 2002).

The **low atmospheric pressure** present in the centre of tropical cyclones results in a dome of water forming, as the sea responds to atmospheric pressure variations by adjusting sea level, such that the water pressure at depth is uniform (Flather, 2001). This is called the inverse barometer effect, and a decrease in atmospheric pressure of 100 pascals ($\approx$ 1 millibar) produces an increase in sea level of approximately 1 cm (Flather, 2001). The maximum water level increase caused by this effect is approximately 1 m. Before the cyclone reaches the shore, this effect is responsible for the majority of the storm surge height.

The **high winds** associated with cyclones are the strongest force increasing storm surge height when the cyclone nears the coast, as wind forcing is greatest in shallow water (Flather, 2001). The winds increase water levels in several ways. Winds blowing toward the shore create water flows in the same direction, creating a water surface elevation gradient and increasing water levels at the coast (Flather, 2001). Winds blowing parallel to the shore create a net transport of water to the right of the direction of the wind in the northern hemisphere, and to the left of the direction of the wind in the southern hemisphere, in a process called Ekman transport (Stull, 2000); this can also pile up water along a shoreline. Another mechanism results from the cyclonic winds setting up a circular motion within the water, with the water being forced either clockwise or counter-clockwise (in the southern and northern hemisphere respectively), and towards the centre; this forces water to flow downwards towards the sea floor in the centre. In shallow water, this downward flow of water is blocked, resulting in a dome of water forming at the surface (Simpson and Riehl, 1981; Masters, 2012).

The waves created by the high winds produce **wave set-up**, which also contributes to increased water levels. Wave set-up occurs when waves break onto the shore; the set-up produces a rise in the mean water level above the still water level of the sea (Komar, 1998). The set-up can be observed as an upward slope of the water towards the land (Komar, 1998; shown schematically in Figure 3). Wave set-up is caused by the transfer of wave momentum to the water column that occurs when waves break; this forces a change in the water level (Weaver, 2004) (this effect is also referred to as radiation stress). In Florida, Dean and Bender (2006) estimated that wave set-up could contribute 30 to 60% of the total 100-year return period storm surge peak water levels.

When the waves break, the water rushes up the shore until the energy in the flowing water is dissipated by friction and by its work against gravity. This final flow of water from breaking waves is called **wave run-up**, and the zone receiving only run-up is called the swash zone.

1.3. Factors affecting surge height and surge reduction

In addition to the forces described in Section 1.2, a number of other factors also affect surge height and extent, either increasing it (e.g. estuaries may funnel water, resulting in increased water levels and inundation extent) or decreasing it (e.g. surface friction can slow water flows, lowering surge height and inundation extent). These additional factors include:

- **hurricane size** (measured as the radius of maximum winds): hurricanes with large geographical extents generate higher peak water levels and greater flooded volumes (i.e. larger inundation extents, dependent on local topography) (Rego and Li, 2009);

- **hurricane forward speed:** surges created by hurricanes with faster forward speeds are expected to create higher surges, but lower flooded volumes (and therefore smaller areas of inundation): Surges with lower forward speeds produce more flooding but lower peak water levels (Rego and Li, 2009);
- **near-shore bathymetry:** coastal shelves with large shallow-water areas produce larger surges than steep off-shore slopes (Flather, 2001; Resio and Westerink, 2008; Rego and Li, 2009);
- **geometry of the coastline:** concave coasts concentrate the surge into a smaller area, resulting in higher water levels, while when the coast is more convex, the water is able to flow sideways and the surge height is reduced (Flather, 2001);
- **inland channels interconnecting water bodies:** these allow the surge to flow more easily and quickly through the landscape, and the surge can propagate further inland;
- **the storm track** has a major effect on the characteristics of a storm surge, through the interaction between the cyclone and landscape features (i.e. coastline geometry and coastal topography), which may affect the build-up of water. For example, Hurricanes Donna and Betsy occurred in the same area (the Florida Keys) and were of comparable size and intensity (they were category 4 and category 3 hurricanes respectively when they made landfall in Florida). However, the south to north track of Hurricane Donna produced a 4 m storm surge in the Middle Florida Keys, while the east to west track of Hurricane Betsy across Florida resulted in a smaller 3 m storm surge at Biscayne Bay (Perkins and Enos, 1968); and
- **frictional resistance**, usually referred to as **surface roughness**, of the land surface: surface roughness is affected by topographic features, ecosystems (e.g. mangroves and saltmarshes), and man-made structures. Increased surface roughness will slow the rate at which water flows inland, and also lead to a steepening of the surge front (i.e. water levels rise more quickly in any given location) (Resio and Westerink, 2008).

1.4 Storm surge reduction by mangroves

Of the factors known to affect surge height that are described above, mangroves directly affect surface roughness (discussed in Sections 2 and 3), height of surface wind waves (Section 3.2 and McIvor *et al.*, 2012), and the speed of the wind directly over the water surface within areas where the vegetation reaches above the water level (Section 4).

Over the longer term (decades to centuries), mangroves can also alter the surface elevation of the shore (influencing the bathymetry and topography), the local geometry (e.g. through progradation, which is the expansion of wetland areas towards the sea) and the location of channels (Spencer and Möller, 2012; McIvor *et al.*, in prep), all of which also influence the height of surges (Section 1.3 above).

In this report we focus on the ability of mangroves to reduce storm surge water levels through their immediate effect on surface roughness and wind waves. In separate reports, we describe the attenuation of wind and swell waves by mangroves in more detail (McIvor *et al.*, 2012) and the change in surface elevation within mangroves (McIvor *et al.*, in prep.).

2. Evidence for storm surge reduction by mangroves

Evidence for the ability of mangroves to reduce storm surges and associated damage comes from three sources: direct observations of water level heights (Section 2.1); the use of well-validated numerical models that simulate storm surge behaviour in the presence or absence of

mangroves (Section 2.2); and observations of the damage caused and the number of lives lost from storm surges (Section 5).

2.1 Water level observations

Very few studies have measured storm surge water levels within mangrove areas. Here we describe the study by Krauss *et al.* (2009). Zhang *et al.* (2012) also have water level data, which they use to validate numerical models of storm surges, as described in Section 2.2 below. For comparison, estimated reductions in peak water levels through saltmarshes are given in Table 3; these range from 1.7 to 25.0 cm/km.

To investigate whether mangroves can reduce the height of peak water levels as storm surges pass through, Krauss *et al.* (2009) analysed water level measurements in wetland areas during Hurricanes Charley (2004) and Wilma (2005) in Florida (Table 4). They used a network of water level recorders that collected water level data at hourly intervals in two different wetland ecosystems containing mangroves and saltmarshes (Table 4), as described below.

As the storm surge from Hurricane Charley passed through the Ten Thousand Islands National Wildlife Refuge (NWR), the peak water level reduction was 9.4 cm/km through an area that included both mangroves and saltmarsh. The following calculations based on data given in Krauss *et al.* (2009; Figure 2 and p. 145) show how the reduction in peak water level through the mangrove area may have been higher. At the first recording point 2.3 km from Faka Union Bay, the peak water level was 78.6 cm above ground level and 43.6 cm above the expected high tide level; at the second recording point 3.2 km further inland, at the transition between the mangrove and the marsh, the peak water level was 40 cm above ground level and 29.6 cm higher than the water level prior to the arrival of the storm surge. This implies a decrease in peak water level of 14.0 cm (reduction in water level relative to high tide/antecedent water levels) over 0.9 km, equivalent to a reduction in peak water level through mangroves of 15.8 cm/km.

Table 3. Measured rates of storm surge reduction across coastal marshes in Louisiana. Most of the data (except the first two rows) are based on a data-set from Hurricane Rita collected by McGee *et al.* (2006).

Hurricane (year, category)	Location	Vegetation type	Surge reduction (cm/km)	Reference
Multiple (1907-1957)	Louisiana	coastal wetlands	6.9 (range 1.7 to 20)	US Army Corps of Engineers (1963) in Engle (2011) (also referred to in Wamsley <i>et al.</i> (2010) and Krauss <i>et al.</i> (2009))
Andrew (1992; cat. 5)	Louisiana	marsh and open water	4.4 to 4.9	Wamsley <i>et al.</i> (2010) and Engle (2011) using data from Lovelace (1994)
Rita (2005; cat. 3)	Cameron Prairie, Louisiana	marsh	10.0	Wamsley <i>et al.</i> (2010), calculated with data from McGee <i>et al.</i> (2006)
Rita (2005; cat. 3)	Sabine, Louisiana	marsh	25.0	Wamsley <i>et al.</i> (2010), calculated with data from McGee <i>et al.</i> (2006)
Rita (2005; cat. 3)	Vermillion, Louisiana	marsh	4.0	Wamsley <i>et al.</i> (2010), calculated with data from McGee <i>et al.</i> (2006)
Rita (2005; cat. 3)	Vermillion, Louisiana	marsh	7.7	Wamsley <i>et al.</i> (2010), calculated with data from McGee <i>et al.</i> (2006)
Rita (2005; cat. 3)	Louisiana	marsh	13.5 (range 3.3 to 23.3)	Kemp (2008), based on data from McGee <i>et al.</i> (2006)

As the storm surge from Hurricane Wilma passed through the mangrove forest along the Shark River in the Everglades National Park, peak water levels were reduced by 4.2 cm/km; this was measured across 3 recording stations set back from the river by 50 to 80 m. The highest water level reduction was between the 2 inland stations that were located 9.9 and 18.2 km from the mouth of the river: peak water level fell from 104.0 cm to 46.2 cm, equivalent to a peak water level reduction of 6.9 cm/km. Between the seaward recording stations located 4.1 and 9.9 km from the river mouth, there was a slight increase in water level, presumably because of river water backing up behind the surge (Krauss *et al.*, 2009).

Krauss *et al.* (2009, p. 147-8) conclude by pointing out that “while our observations indicate that water levels were reduced as storm surge moved through coastal mangrove ecosystems, uncertainty remains over the relative contribution of mangroves over other wetland types, open water or microtopographic relief along the Gulf Coast over similar distances”; i.e. it is unclear what the contribution of mangroves was to the reduction in peak water level, as it is impossible to control for these other factors that may also affect water level changes. Because of this, numerical models that include this greater range of factors have an essential role to play in helping us understand the relative contribution of mangroves to storm surge reduction; such models are described in the following section.

Table 4 Peak water level reduction during storm surges passing through mangrove wetlands in Florida (data from Krauss *et al.*, 2009). Hurricane categories refer to the Saffir-Simpson Hurricane Wind Scale (Schott *et al.*, 2012; Box 3). *These statistics have been calculated from data in Figure 2 of Krauss *et al.* (2009), as described in the text above.

Location	Associated hurricane	Wetland type	Water level recording points	Peak water level height reduction (cm/km)
Ten Thousand Islands National Wildlife Refuge, Florida, USA	Hurricane Charley, a category 4 hurricane, 13 August 2004, with maximum sustained winds of 240 km/hr at landfall; the location of peak water level travelled at 0.4 km/hr	mangrove/interior marsh community; in mangrove area, dominant species was <i>Rhizophora mangle</i>	4 points approx. 1 km apart and in line with each other, laid out in a landwards direction; area between 1st two points was mangrove, other areas were salt marsh	9.4 across all 4 recording points, which included salt marshes and mangroves; 15.8 in mangrove area*
Along the Shark River (Everglades national Park) in south western Florida, USA	Hurricane Wilma, a category 3 hurricane, 24 October 2005, with maximum sustained winds of 195 km/hr and a very wide eye 89 – 105 km in diameter (Smith <i>et al.</i> , 2009); location of peak water level travelled at 1.4 km/hr up river; peak water level 5 m in some locations; the hurricane crossed the Florida peninsula in 4.5 hours	riverine mangrove swamp, dominant species is <i>Rhizophora mangle</i> (Chen and Twilley, 1999)	recorders placed 50-80m from the river's edge at river-km 4.1, 9.9 and 18.2	4.2 across all 3 recording points; -0.2 between lower pair of recorders due to river water backing up, 6.9 between upper recorders

2.2 Numerical modelling studies

Numerical simulations of storm surges that are well-validated against field observations offer a complementary approach to understanding the factors affecting storm surge water levels. When such simulations can be shown to accurately represent storm surge behaviour in the presence of mangroves, then they may be used to look at the effect of varying parameters such as storm surge height, forward travel speed and the width of the mangrove forest, as described below. Such models might also be applied to predicting storm surge reductions due to existing mangroves or planned mangrove restorations.

Four studies that have used numerical modelling approaches to better understand the factors affecting storm surge inundation in mangroves are summarised below. The first two studies relate specifically to mangroves; the second two studies are based on models that include vegetation which is relatively similar to mangroves.

2.2.1 The ELCIRC model

Xu *et al.* (2010) used an unstructured Eulerian-Lagrangian Circulation (ELCIRC) model (Zhang *et al.*, 2004) to model the surge from Hurricane Andrew (1992; category 5; Table 1) at Biscayne Bay on the east coast of Florida. They found that their model overestimated peak water levels and flooding extent in the southern part of the bay, an area containing mangrove zones with widths of 1 to 4 km and tree heights of 1 to 20 m. This suggested that land cover types, in particular the large areas of mangroves, were having significant effects on flood levels and extent. To improve the accuracy of the storm surge simulation, they incorporated the effect of land cover into the model by varying Manning's coefficient, a measure of surface roughness (described in more detail in Box 4). They tested three different values of Manning's coefficient (0.05, 0.1, and 0.15) for areas with mangroves, and found that surge inundation extents most closely matched the observed debris line when a coefficient of 0.15 was used (a Manning's roughness coefficient of 0.15 is relatively high, and is typical of dense woodlands; Box 4). They concluded that changes in roughness coefficients due to vegetation can significantly influence the local inundation patterns during storm surges.

2.2.2 The CEST model

Zhang *et al.* (2012) used a different model, the Coastal and Estuarine Storm Tide (CEST) model, to simulate the passage of Hurricane Wilma (category 3; more storm characteristics given in Table 4) as it passed over the Gulf Coast of South Florida in 2005. This is a 200 km length of coastline, with a mangrove belt varying between 6 and 30 km in width. The mangrove trees are 4 to 18 m high, with stem diameters of 5 to 60 cm, and scrub mangrove is found further inland; the dominant species are *Rhizophora mangle*, *Laguncularia racemosa* and *Avicennia germinans*. Hurricane Wilma resulted in extensive coastal flooding with a maximum storm surge of 5 m (Smith *et al.*, 2009). Zhang *et al.* (2012) were able to compare the model's outputs with abundant field data describing water levels, collected by various agencies including NOAA, USGS, FEMA and academic researchers.

In particular, Zhang *et al.* (2012) focused on whether the model's predictions were improved by including mangroves in the model. The geographical extent of mangroves was taken from the National Land Cover Dataset created by the US Geological Survey in 2001. Following Xu *et al.* (2010), the drag force from mangroves was included in the model by adjusting Manning's roughness coefficient. While Xu *et al.* (2010) used a Manning coefficient of 0.15, Zhang *et al.* (2012) reduced it to 0.14 because of the large number of lakes, rivers and creeks inside the mangrove zone in this area.

Box 4. Manning's Roughness Coefficient n

Manning's roughness coefficient (also called Manning's friction factor; denoted by the symbol n) is one of the most important parameters for describing water flow over surfaces (Li and Zhang, 2001). It is used to quantify the resistance to flow in channels, floodplains and areas affected by storm surges (Chow *et al.* 1988). It includes the shear stress caused by the boundary roughness (i.e. friction between the fluid and the surface), and, when vegetation is present, it also includes a drag force caused by the vegetation (Jin *et al.*, 2000). It is usually considered to be dimensionless (Chow, 1959; see p. 98-99 for a discussion of this issue).

Manning's n has been empirically measured for a variety of surfaces that may occur in river channels and floodplains; Table 4.1 gives some values for various surfaces. On floodplains, n usually varies with the water depth, and is highest when the water is shallow (Chow, 1959). In vegetated areas, n can also vary with season and is highest in the growing season (Chow, 1959).

Table 4.1 Manning's roughness coefficient (Chow, 1959; Mattocks and Forbes, 2008).

Land use type	Manning's coefficient
Open water/sand	0.02
Scattered brush/shrub/scrub	0.05 (0.035 to 0.07)
Forest/estuarine forested wetland	0.10 (0.08 to 0.12)
Dense woods (e.g. dense willows)	0.15 (0.11 to 0.20)

Zhang *et al.* (2012) ran simulations using: i) a constant Manning coefficient of 0.02 for all spatial cells (a Manning coefficient of 0.02 is typical of the seabed, and so in this simulation, mangroves were not included) and ii) with varying Manning coefficients that reflected the presence of mangroves in some cells. The best match between the simulation and the observed data was seen when mangroves were included in this way; the root mean square error of computed peak surge heights versus observed ones decreased from 0.60 m (mangroves not included) to 0.39 m (with mangroves included). The inundation areas predicted by the model were 4,220 km² without mangroves and 2,450 km² with mangroves, suggesting that mangroves had a large effect on the inundation extent. Flooding was restricted within the mangrove zone when mangroves were included in the model, and this matches well with the measured inundation extent taken from surge-induced sediment deposits, which were limited to a zone less than 14 km from the Gulf of Mexico.

Storm surge reduction rates were between 20 and 50 cm/km through the mangrove areas (Zhang *et al.*, 2012). The simulations indicated that without the mangrove zone, surge amplitudes would decrease by 6 to 10 cm/km. The simulations suggested that storm surge reduction was non-linear across the mangrove width, and this is discussed further in Section 3.1 below.

While the peak water level height was reduced as the storm surge passed through the mangroves, Zhang *et al.*'s (2012) simulations showed a 10-30% increase in water levels in front of the mangrove zone, compared to simulations without mangroves. This is because the

mangroves act as an obstruction to the flow of water, causing water to build up in front of them. Increased friction within mangroves may also lead to a steeper surge front as the surge moves inland (Resio and Westerink, 2008).

2.2.3 A model exploring wave set-up

In addition to the long period wave that makes up the storm surge, short period wind waves are often present on top of the storm surge. These wind waves increase the damage caused by the storm surge, and can increase the area that is flooded through wave set-up. However, such wind waves may be reduced by mangroves over relatively short distances: small wind waves can be reduced in height by more than 75 to 100 % over 1 km of mangroves (Mazda *et al.*, 2006; Quartel *et al.* 2007; these values are calculated from the wave reduction rates per metre given in these original studies, using equation 2 from Mazda *et al.*, 2006; note that these wave reduction rates were not measured during storm surge conditions). Wave reduction is expected to be dependent on the surge water level and the density of vegetation (i.e. aerial roots or branches); the largest rates of wave reduction occur when the waves encounter the densest vegetation (Quartel *et al.*, 2007). Therefore mangroves may be able to significantly reduce wave set-up and run-up during storm surges, thereby reducing impacts on local infrastructure.

Dean and Bender (2006) used a numerical modelling approach to explore the effect of vegetation (modelled as an array of cylinders) on wave set-up. They found that when waves are modelled based on the Airy wave theory (linear wave theory), which assumes that wave height is small compared to water depth and wave length (Komar, 1998), vegetation in shallow water should reduce the set-up to one-third of the amount that would have been present without vegetation. Vegetation in deeper water may result in a set-down (i.e. a reduction in water level), and the water depth at which this change from a set-up to a set-down occurs is kh , where k is the wave number ($= 2\pi / \text{wavelength}$) and h is the still water depth (i.e. depth without waves). For waves modelled using non-linear wave equations (based on third-order equations, which no longer assume that wave height is small relative to water depth; Stive and Wind, 1982), vegetation also resulted in a set-down (Dean and Bender, 2006). Dean and Bender's results are as yet unvalidated, but they suggest that vegetation such as mangroves could have a very large effect on storm surge water levels in those areas where wave set-up makes a large contribution to the raised water levels.

2.2.4 A model of the storm surge and wind waves from Cyclone Sidr

Tanaka (2008) developed a different approach to numerically model the storm surge from Cyclone Sidr, which made landfall in Bangladesh in 2007. He modelled the passage of short period wind waves (wave period 1 or 2 minutes) and a longer period storm surge (wave period 1 or 2 hours) through trees; both types of waves were modelled separately and in combination. The modelled vegetation characteristics were based on the non-mangrove tree species *Casuarina equisetifolia*; trees were modelled as cylinders, 10 m high and 16 cm in diameter, with 0.35 trees/m² in a triangular arrangement. He used a one-dimensional non-linear long wave differential equation to explore different short wave and long wave storm surge conditions and wind conditions, in the presence or absence of a 150 m wide band of vegetation. The underlying topography and vegetation measurements matched those seen in transects in Mathbaria, Bangladesh, and the model results were compared with observations of how this area was affected by Cyclone Sidr.

Using his model, Tanaka (2008) found that a 150m band of vegetation had a small effect on the wind's ability to raise water levels (assuming wind speeds of 60 m/s, but not including

wind waves); the water level was 4 cm lower behind the vegetation than would be expected with no vegetation present. Vegetation had no effect on the water depth caused by the long wave component (i.e. the storm surge) although the vegetation slightly decreased the velocity of water inside the vegetation zone and the arrival time of the peak of the storm surge. When wind waves were included on top of the storm surge as short waves with a period of 1 or 2 minutes, the water depth behind the vegetation was reduced by 12 or 28 cm respectively (compared to no vegetation being present). This supports the idea that the largest effect of vegetation on storm surge water levels may be through their effect on the short-period wind waves on top of the surge.

Additionally Tanaka (2008) explored the effect of changing the ground slope. He found that when the gradient of the land was 1 in 500 (as opposed to 1 in 100, as used in the previous model runs), and with a long wave with a period of 2 hours and short waves with a period of 2 minutes, water depth behind the vegetation was 80 cm lower than it would have been without the vegetation. Tanaka concluded that the vegetation had a larger effect on reducing water depths (i.e. storm surge water levels) when the storm surge was passing over a lower angle slope (i.e. closer to horizontal).

The results from Tanaka's model broadly match observations: Tanaka (2008) reported that during cyclone Sidr in Bangladesh in November 2007, a 150 m band of river-side vegetation near Mathbaria may have caused a 0.5 to 1.0 m difference in water level (including the effects of wind and swell waves) behind the trees. The observed reduction in water level was based on interviews with local people about where the water reached.

Tanaka's model demonstrates that a relatively narrow band of trees (150 m) may result in a fairly large reduction in water levels (0.8 m), when the effect of vegetation on wind waves is also taken into account. Further research is needed to verify this result for mangroves.

3. Factors affecting storm surge reduction in mangroves

Storm surge reduction through mangroves is expected to depend on a number of factors including: mangrove characteristics, such as forest width, tree density and structural complexity (roots, stems, branches and foliage) of the dominant species; physical characteristics, such as the presence of channels and pools, and the topography of the area (both are influenced by mangroves); and storm characteristics, such as the size and forward speed of the cyclone (which may interact with mangroves to influence storm surge reduction). In this section we review what is known about how these different factors affect storm surge reduction. Few quantitative data are available; where data exist, they are generally derived from numerical models rather than observations.

3.1 Mangrove width

Measurements of storm surge reduction rates through coastal wetlands are often quoted as a certain number of centimetres of water level reduction per metre of inland distance, usually measured in the direction of travel of the surge (e.g. Tables 3 and 4). However such constant attenuation rates imply a linear reduction in water level with distance into the mangroves (i.e. mangrove width). This is rarely true, both because the landscape is usually heterogeneous (i.e. it is usually a mixture of channels, pools and vegetation with a varied topography), and also because the underlying rate of reduction might not be linear even if the environment were homogeneous, as described below. Consequently, such attenuation rates should be regarded with caution. At best they may serve as rules of thumb around which there is usually a high degree of scatter (Resio and Westerink, 2008; Wamsley *et al.*, 2010). Taking this into

account and based on the studies described in Section 2, the rate of reduction of surges through mangroves appears to range between 5 and 15 cm/km (observed reduction rates; Krauss *et al.*, 2009) up to 50 cm/km (well-validated numerical models; Zhang *et al.*, 2012).

Zhang *et al.* (2012) used the simulations described in Section 2.2.2 to explore the effects of different widths of mangroves being present, and they found that surge attenuation through mangroves was non-linear: the largest reduction in peak water levels occurred at the seaward edge of the mangroves, while further inland the water level changed more slowly (Figure 5). They suggest that this might explain the relatively low rates of peak water level reductions measured by Krauss *et al.* (2009; described in Section 2.1), whose measurements start some distance into the mangroves; the water level reduction in the most seaward mangroves might have been higher.

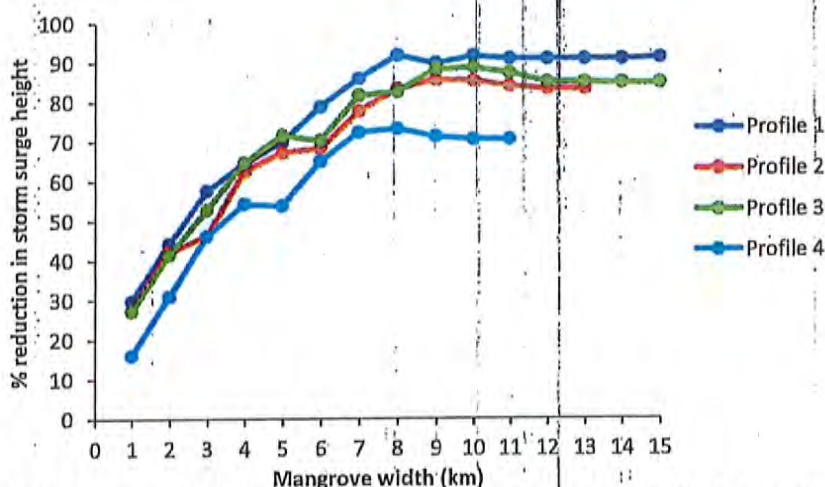


Figure 5. The reduction in storm surge height as the mangrove belt width increases for four different shore profiles (adapted from Zhang *et al.*, 2012).

3.2 Mangrove vegetation characteristics

The density of mangrove vegetation and the diameter of aerial roots and stems are expected to affect the ability of mangroves to reduce storm surge water levels (Krauss *et al.*, 2009; Alongi, 2008). However, few data are yet available to support this assumption.

Mazda *et al.* (1997a) studied tidal flows, which are relatively similar to storm surge flows, in an area with young *Kandelia candel* trees (less than 7 years old). They found that the tides rose faster at the early stage of the flood tide and fell more slowly at the latter stage of the ebb tide than in a nearby location without mangroves. They attribute this difference to the flow resistance from the mangrove vegetation and the bottom mud. They note that the changes in flow speed were considerably smaller than those seen in mangrove swamps dominated by *Rhizophora* spp. or *Bruguiera* spp., as measured by Wolanski *et al.* (1992) and their own unpublished data. Unlike *Kandelia candel*, these other species have prop roots or pneumatophores, which are likely to slow water flows more than the trunks of *Kandelia* (Mazda *et al.*, 1997a).

Mazda *et al.* (1997b) created an "effective vegetation length scale" to quantify the effect of mangrove vegetation on tidal flows. The effective vegetation length scale is calculated from the projected area (i.e. silhouette) of the vegetation and the volume of the vegetation, and

varies with tidal depth. Mazda *et al.* (1997b) conclude that mangrove species, density and tidal level strongly affect the hydrodynamics in mangrove forests.

The characteristics of mangroves known to reduce wave height are also likely to reduce wave set-up and wave run-up. The most important mangrove characteristic for reducing wave height is the projected area of vegetation (i.e. the silhouette of the vegetation, as seen from the direction of on-coming waves) (Quartel *et al.*, 2007). Dense forests containing species with aerial roots and dense canopies are therefore expected to reduce waves and wave set-up more efficiently. However most studies of wave attenuation in mangroves have focused on much smaller waves than those usually seen during storm surges (wave heights studied have mostly been less than 70 cm. (See McIvor *et al.* (2012) for a detailed review of this topic.)

3.2.1 Including vegetation characteristics within numerical models

Improved representation of vegetation (i.e. mangroves or saltmarshes) in numerical surge models could increase the accuracy of estimates of inundation extent and duration (Medeiros *et al.*, 2012). In order to include variation in mangrove density or morphology in numerical models such as the CEST model used by Zhang *et al.* (2012; Section 2.2.2), Manning's roughness coefficient would need to vary in a way that realistically reflected the geographical variation in mangrove characteristics. Currently, roughness is estimated from the National Land Cover Dataset (NLCD); land cover types such as grassland, woody wetland, open water and commercial uses are distinguished, and each of these is associated with a range of Manning's coefficient values.

Medeiros *et al.* (2012) tested the validity of using the NLCD to estimate roughness. They directly estimated Manning's roughness coefficient at 24 sites affected by storm surges in Florida, based on a visual assessment of vegetation density, microtopography and obstructions (e.g. tree stumps) using descriptions and images in Arcement and Schneider (1989). They then compared these observed values with the values of Manning's roughness coefficient estimated from the NLCD classification of the site. At some of the sites, their estimate of Manning's coefficient was quite dissimilar from the roughness coefficient based on the NLCD. This could be because of variability of surface roughness within land-cover types, original misclassification of the land cover type, or a lack of information about the roughness coefficient for some land cover types. They suggest that LiDAR (Light Detection and Ranging) data could be used instead of land cover data derived from aerial photographs to estimate Manning's roughness coefficient for use in numerical storm surge models. Methods are being developed to estimate surface roughness from LiDAR data (Straatsma, 2008; Forziera *et al.*, 2010); this approach would allow local variation in mangrove vegetation to be included in storm surge models as variation in the surface roughness coefficient.

An alternative approach to including variation in vegetation in numerical storm surge models has been proposed by Sheng *et al.* (2012). They propose a three-dimensional numerical model of storm surges based on the coupled CH3D-SWAN (Curvilinear-Hydrodynamics 3D – Simulating Waves Nearshore) model (more information about the SWAN model is given in Booij *et al.*, 1999, and Suzuki *et al.*, 2011). Sheng *et al.* (2012) demonstrate the model by simulating the flow of a surge through vegetation similar to that found in marshes. The model allows them to vary the height, density and width of the vegetation, and they find that increases in height, density and/or width result in a reduction in inundation volume. Their model is yet to be applied to mangrove vegetation.

3.3 The presence of channels and pools within mangrove areas

Both Krauss *et al.* (2009) and Zhang *et al.* (2012) consider the presence of channels and pools as likely to decrease the ability of mangroves to reduce peak water levels, because the water is able to pass more easily along the rivers and penetrate further inland.

Krauss *et al.* (2009) recorded higher rates of reduction of peak water levels in intact, relatively unchannelized expanses of mangroves in the Ten Thousand Islands NWR than through riverine areas along the Shark River (9.4 cm/km versus 4.2 cm/km respectively). However they point out that such differences may have been due to differences in the storm characteristics or other factors. The peak water level also travelled more quickly up the Shark River mangrove area (1.4 km/hr in the Shark River area, compared to 0.4 km/hr in the Ten Thousand Islands NWR).

Zhang *et al.* (2012) found that surge height decreased at a rate of 23 cm/km through an area with a mixture of mangrove islands and open water, while in areas with less open water, surge height reduction rates ranged from 40 to 48 cm/km.

3.4 Topography

Topography is the most important local factor affecting inundation from storm surges, interacting with the peak water level to influence the extent of inundation. In addition to the potential interaction between topography and mangrove vegetation in reducing storm surge water levels (Tanaka, 2008, as described in Section 2.2.4), mangroves also affect local topography over the longer term through their effect on tidal creeks. The channels that usually drain wetland areas and the levees that form around these channels increase topographic roughness, thereby increasing drag on water flows, and potentially reducing storm surge water levels. Mangroves play a role in maintaining such features, both influencing creek depth and strengthening the banks of creeks (Spencer and Möller, 2012).

Mangroves influence creek depth as follows: once submerged during tidal inundation, the surface roughness induced by the trees holds back the water on the ebb part of the tidal cycle (during the falling tide), while water levels in the creeks themselves fall rapidly. This causes water surface slopes as high as 1 m per 1000 m; the high water surface gradients result in large tidal ebb flow velocities within the creeks once the water levels fall below their bank height (Wolanski *et al.*, 1992). These high flows within the creeks help to maintain the network of creeks (Wolanski *et al.*, 1992). Mangrove species composition and morphology may influence ebb flow velocities (and hence possibly creek networks): species with dense prop roots and pneumatophores, e.g. *Rhizophora* and *Bruguiera*, may hold the water back more effectively than species without, resulting in higher flow velocities within creeks as the tide lowers (Mazda *et al.*, 1997b).

The roots of mangrove trees also hold the soil together, reducing bank erosion, and trap sediments, causing the formation of levees near creek margins (Thom, 1967; Augustinus, 1995; Spencer and Möller, 2012). Some species of mangroves may reduce bank erosion more effectively than others; for example, Teas (1980) suggests that black and white mangroves (*Avicennia germinans* and *Laguncularia racemosa*) form denser mats of roots than red mangroves (*Rhizophora mangle*), and are therefore more able to stabilize shorelines. Likewise, species composition can influence accretion of sediment, with higher rates of accretion observed under the prop roots of *Rhizophora* than the pneumatophores of *Sonneratia* in Micronesia (Krauss *et al.*, 2003).

3.5 The size and speed of the cyclone and storm surge

Storm surges associated with hurricanes with a fast forward speed may be reduced more by mangroves than surges created by hurricanes with a slower forward speed: Zhang *et al.* (2012) used numerical simulations to show that south Florida mangroves may be expected to protect the area behind them against flooding from a Category 5 hurricane with a fast forward speed of 11.2 m/s, but not from a Category 5 hurricane with a slow forward speed of 2.2 m/s. They estimate that a mangrove forest with a width of tens of kilometres would be needed to attenuate a 2 to 3 metre storm surge from a slow-moving Category 5 hurricane, as the mangroves have little effect on the slow flows seen in surges created by such hurricanes.

Storm surge reduction by mangroves also depends on initial surge height and surface wind speeds as a result of the damage that very large surges and very high wind speeds can do to mangroves. Extreme storm events with very high winds and very large surges may severely damage or destroy mangroves, resulting in smaller surge reduction levels either during that storm or in subsequent storms (see Section 6).

4. Reduction of surface winds by mangroves

Mangroves buffer the water surface from the effects of wind, thereby reducing the generation of wind-waves, wave set-up and run-up, which make a substantial contribution to storm surge flood levels and damage. However, it should be noted that the wind waves riding on top of a storm surge mostly originate from the effect of wind on the water surface outside the mangrove area; by reducing wind speeds over the water surface within the mangrove area, wind waves do not increase in size in this area (and they are usually reduced due to the presence of the mangrove obstacles). This section presents a brief overview of what is known about the ability of mangroves to reduce surface wind speeds and the likely effects of surface wind speed reduction on storm surge heights.

4.1 Measurements of wind speeds behind mangroves

It is well-known that forest canopies modify surface wind speeds (e.g. Raupach and Thom, 1981). Chen *et al.* (2012) measured wind speed and direction close to mangrove plantations (*Sonneratia apetala* and *Kandelia obovata*) in Sanjiang Bay in Haitian Province, South China. They took measurements 2 m above the ground, 50 m from the mangrove forest. They found that mean wind speeds up to 5 m/s were reduced by more than 85% by the mangrove forests, and greater reductions were seen near the *Kandelia* forest. When the mean wind speed was greater than 15 m/s, wind speed was reduced by between 58.9% and 63.6%. This reduction rate remained stable at higher wind speeds such as those seen during typhoons, when mean wind speed and extreme wind speed were reduced by 59.4% and 53.2% respectively (these latter wind speed reductions were measured for the *Sonneratia* forest). The authors note that the *Sonneratia* plantation reduced wind speeds more effectively during the warm season; presumably this resulted from denser foliage on the trees.

4.2 Estimating how reductions in wind speed caused by vegetation affect storm surges

It is not possible to directly measure the effect of vegetation-reduced wind speeds on storm surge heights, because the effects of reduced wind speeds would never occur independently of other effects such as increased drag on the water flow from the vegetation.

Using the Advanced Circulation numerical model, Westerink *et al.* (2008) explored how peak water levels varied in hindcasts of Hurricanes Betsy and Andrew when surface wind speeds were modified to reflect the differences in land cover (e.g. dense forested canopies, marshland, or buildings). Using two sources of information on atmospheric forcing, they then

modified surface wind speeds to account for the higher surface roughness over land, as well as the level of local inundation (once a land feature was underwater it no longer affected wind speeds). They also took wind direction into account. They found that in the hindcasts of Hurricanes Betsy and Andrew, peak water levels were more than one metre lower in some areas when the modified wind speeds were included, as opposed to wind speeds assuming open-ocean marine conditions. (See Westerink *et al.* (2008) for a detailed explanation of their procedures for reducing wind speeds based on local surface roughness.) This implies that the effects of vegetation on wind speeds could significantly influence storm surge water levels.

5. Reduced damage and loss of life behind mangroves

A small number of studies have investigated whether mangroves help to reduce damage and loss of life during storm surges.

Badola and Hussain (2005) conducted a study based on local people's perceptions of the protective services provided by mangroves against storm surges caused by cyclones. They worked in three villages in Orissa, India, following a cyclone with a 9 m storm surge in October 1999. The three villages were Bankual, located within the Bhitarkanika mangrove ecosystem wildlife sanctuary (145 km²); Bandhamal, surrounded by embankments that were breached by the storm surge; and Singdi, which was not surrounded by either mangroves or embankments. The villages were within 15 km of each other and chosen to be otherwise similar in terms of their distance from the coast and the damage attributable due to wind (village elevation above sea level is not given, but the authors state that "water logging" was similar, i.e. flood water levels were similar).

Badola and Hussain (2005) found that in the mangrove-protected village, damage to houses and other adverse effects were lowest, while crop yields and other positive factors were least impacted. The loss incurred per household was highest in the village that was protected by an embankment (US\$153.74), followed by the village with no protection (US\$44.02), with the lowest losses in the village protected by mangroves (US\$33.31). The reason losses were so high in the embankment-surrounded village was that the embankments were breached. This allowed sea water in, but subsequently the sea water took time to drain out of the breaches, so that crops were damaged more than in the village with no protection, whose fields suffered the highest level of inundation but the sea water quickly drained away, resulting in less crop damage. Badola and Hussain (2005) note that embankments near the mangrove forest were not breached while those further away were breached in a number of places, implying that mangroves may have helped to protect these embankments.

A larger scale study using data from several hundred villages was conducted by Das and Vincent (2009) after the same cyclone and in the same region as the study by Badola and Hussain (2005). Nearly 10,000 people died in the 1999 super cyclone, and more than 70% of these drowned. Das and Vincent (2009) used data from the Kedrapada District in Orissa, a district just north of the cyclone's landfall, to explore whether the presence of mangroves in front of villages reduced the number of villagers who lost their lives. They included 409 villages in their study; all villages were known to have had mangroves between them and the coast in 1944, and this ensured that where villages were not fronted by mangroves in 1999, this was due to loss of vegetation and not a lack of suitable habitat. They used a regression analysis to test whether the number of deaths was related to the mangrove width in front of villages, and included a large number of other factors in their statistical model, such as

distance from the coast and height of storm surge, the result was robust to the inclusion of these other variables¹.

Das and Vincent (2009) found that villages with wider mangroves between them and the coast had significantly fewer deaths than villages with narrower mangrove belts or no mangroves. They predicted that there would have been 1.72 additional deaths per village within 10 km of the coast if mangroves had not been present. They point out that mangroves saved fewer lives than an early warning issued by the government, which saved 5.84 lives per village. However, for those people who stayed behind despite the warning, the mangroves reduced the number of deaths (Vincent and Das, 2009). This example also illustrates how a combination of different risk reduction measures (in this case, early warning systems and mangrove forests) can provide an increased level of protection in comparison to single measures alone.

6. The effect of tropical cyclones and storm surges on mangroves

While mangroves can reduce storm surge peak water levels, they can also be affected by cyclones and their accompanying storm surges. The impacts of storms on mangrove ecosystems will depend on the mangrove ecology and local geomorphology.

Smaller cyclones and storm surges may result in some tree mortality and defoliation, but in most cases the structural complexity of the forest is maintained, and the forest is able to recover. During rare extreme events, tree mortality can be more extensive, caused by the breaking of trunks; uprooting of trees, the loosening or shredding of bark, and severe defoliation (Jimenez *et al.*, 1985; McCoy *et al.*, 1996; Lacambra *et al.*, 2008; Figure 6). The flooding and siltation that may accompany cyclones can cause further tree mortality (Jimenez *et al.*, 1985; Lacambra *et al.*, 2008). If mass tree mortality occurs, it may result in the subsequent collapse of sediment and loss of surface elevation, as dead roots decay (Cahoon, 2006). The resulting elevation of the mangrove soil surface may be below mean sea level, making the areas unsuitable for colonisation by mangroves until such a time as the surface has risen in elevation to mean sea level (mangroves cannot live below mean sea level because the increased frequency and duration of tidal inundation create soil conditions that are harmful to the mangrove roots).

Tanaka (2008) examined trees that had been damaged by Cyclone Sidr in Bangladesh (2007) and found that the patterns of damage were similar to those seen after tsunamis or river floods. Trees were bent or their trunks were broken; some trees were overturned or uprooted;

¹ In response to Das and Vincent's (2009) study, Baird *et al.* (2009) questioned whether other important factors such as distance from the coast were taken into account and whether formal methods of model selection were used in the statistical analysis; they also suggested that the correlation coefficient between mangrove widths and village deaths was very low. In their reply, Vincent and Das (2009) pointed out that distance from the coast was included in the model, along with several other potentially relevant factors; that formal methods were used to progressively add in groups of related variables into their model, with no effect on the significance of their results; and that Baird *et al.* (2009) had wrongly interpreted the correlation coefficient because the discrete nature of the data for number of deaths required that count-data models were used: in fact, the data show that for villages within 10km of the coast, mangroves reduced the average number of deaths by 69%. Vincent and Das (2009) agree with Baird *et al.*'s (2009) point that mangroves saved fewer lives than the early warning system, but point out that many people did not evacuate despite the early warning system, and therefore mangroves played an important role in reducing the death toll amongst those that stayed behind.

the substrate had been eroded away from under some of the trees and there was local scour around the trees. Tree damage can be caused both by the force of the water in the storm surge and by the high winds. Larger trees are more likely to be damaged, and some species of mangroves may fare better than others: for example, after Hurricane Andrew passed across Florida in 1992, *Rhizophora mangle* fared better than *Avicennia germinans*, and both fared better than *Laguncularia racemosa* (McCoy *et al.*, 1996).

Regeneration after a storm can take different pathways depending on the severity of the damage caused, which may determine whether recolonisation occurs primarily via sprouting from surviving trees (more likely from *Avicennia* or *Laguncularia* species) or via new seedling establishment (for *Rhizophora* species) (Baldwin *et al.*, 2001). In some cases, an understory of mangrove seedlings can facilitate rapid regeneration after storms (Figure 6).

While cyclones can alter the structure of mangrove vegetation, in many areas such cyclones are infrequent and mangrove forests are usually able to recover their structural integrity over a number of decades, before another cyclone hits the same area (Krauss *et al.*, 2009).

The speed of recovery after events is likely to be determined by the magnitude of the event and local conditions. This makes it difficult to predict the long-term effects of cyclones and storm surges on mangroves in any particular location. Lacambra *et al.* (2008) and Spencer and Möller (2012) review the effects of cyclones and storm surges on mangroves in more detail.



Figure 6. Left: A mangrove forest on the southwest coast of Everglades National Park in October 2005 after the passage of Hurricane Wilma, showing how trees have been defoliated and some trees have been knocked over. Photo: T.J. Smith III, US Geological Survey. Right: Large numbers of seedlings can be a common feature of the mangrove understory, and can play a critical role in recovery from extreme storm events. Photo: Mark Spaulding.

7. Future research needs and directions

Several areas of future research are needed in order to better understand the role of mangroves in storm surge reduction. Most importantly, further measurements of water levels and inundation extents in mangrove areas during storm surges would help to validate numerical storm surge models. All currently available data are from Florida; mangrove

forests in other parts of the world include different species, with different vegetative structures, which may affect storm surge reduction rates.

An understanding of how surface roughness varies with mangrove forest characteristics might help to refine numerical storm surge models. Data on the geographical variation in mangrove characteristics would be required to include such variation within models.

The effect of mangroves on wave set-up and run-up under storm surge conditions is potentially large, but current understanding of this is entirely from numerical models. Measurements during storm surges are needed in order to validate these models.

Currently our understanding of the reduction of wind and swell waves through mangroves comes from measurements of small waves only (less than 70 cm in height); measurements of the reduction of larger waves during storm surges are also needed to better understand the capacity of mangroves to reduce wave-related damage during storm surges.

8. Conclusions

Both empirical data and numerical models suggest that mangroves can play a role in reducing storm surge peak water levels when the mangroves are present over sufficiently large areas. Mangroves slow the flow of water as the surge moves inland and reduce the waves riding on top of the surge, lowering water levels and reducing damage behind the mangroves.

Our current understanding of the effect of mangroves on storm surges comes from relatively few studies. These studies measured reductions in peak water levels of 5 to 50 cm per kilometre of mangrove. This implies that a mangrove belt several kilometres wide is needed to significantly reduce storm surge water levels. Such large areas of mangroves are still present in many parts of the tropics that are affected by cyclones and storm surges, including Mexico, the Caribbean, Florida, Bangladesh, India, Indonesia and Australia. In these locations, the conservation and restoration of mangroves can contribute to a risk reduction strategy against storm surge inundation and damage.

While mangroves can only reduce storm surges when they are present over large areas, the wind and swell waves on top of the storm surge may be reduced over much shorter distances (wave height reduction is expected to be greater than 75% over 1 km of mangrove; this is based on studies of smaller waves). By reducing wave height, mangroves are expected to reduce wave set-up and run-up, which contribute to the raised water levels, inundation and damage caused by storm surges. Mangroves also buffer the water surface from winds that would otherwise cause larger wind waves to form on the surge water surface.

There is considerable variability in the recorded levels of storm surge reduction by mangroves. Storm surge reduction is influenced by the characteristics of individual storm surge events, the local physical setting, and the characteristics of the mangrove communities. The relationship between storm surge reduction, bathymetry, topography, distance from shore and width of mangrove vegetation is highly complex; numerical models based on the underlying physics of wind forcing and water movement are best able to represent the behaviour of storm surges (Resio and Westerink, 2008). Such models are needed to explore the effects of mangrove characteristics on storm surge reduction: for example, Zhang *et al.* (2012) used numerical models to explore the effect of changing the width of the mangrove belt. They showed that peak water levels are expected to decline non-linearly with distance, with the greatest reduction in peak water level per unit distance occurring at the seaward

margin. Therefore an increase in the width of the mangrove belt may not provide a proportional increase in water level reduction.

The ability of mangroves to reduce storm surges also depends on the storm surge forward speed, the height of the storm surge and the cyclone intensity. Numerical models suggest that mangroves will be more efficient at reducing surge height for fast-moving surges. Extreme events, with very strong winds or surges many metres high, may damage or destroy mangroves, reducing their ability to reduce surge height. The threshold at which such damage occurs is likely to depend on mangrove species and height (Lacambra *et al.*, 2008). Such damage is usually localised to areas that are relatively close to the storm track.

One limitation of the current numerical models is their inability to include spatial variation in mangrove characteristics, such as mangrove density. It is very likely that the ability of mangroves to reduce peak water levels depends on mangrove characteristics, with sparse, fragmented or channelized areas reducing storm surge water levels less effectively than dense mangrove vegetation. Currently, mangroves are represented in numerical models as an increase in surface roughness, and a single value for the roughness coefficient is used for all mangroves areas (Xu *et al.*, 2010; Zhang *et al.*, 2012). Including mangrove variation would probably improve the prediction of storm surge heights, and would therefore aid in planning the use of mangroves in coastal defence.

Where extensive areas of mangroves currently exist, reducing the threats they face from development, sea level rise and other anthropogenic factors will help to maintain the coastal defence functions that they currently provide against storm surges. In other areas, large-scale restoration or afforestation of mangroves may provide increased levels of protection from storm surges. In such settings, numerical storm surge models will generally be required to calculate the potential benefits of mangroves, based on the known frequency and magnitude of surges in the region, and the physical characteristics of the mangroves (species composition and morphology) and the coast (topography, bathymetry, geomorphology, and the presence of other ecosystems that may provide coastal defence functions). Where mangrove planting is proposed as a means of reducing risk from storm surges, many other considerations should also be taken into account, including the chances of successful mangrove planting, which is dependent both on the methods employed (Lewis, 2005; Lewis and Perillo, 2009; Twilley and Rivera-Monroy, 2005) and on the social and legal frameworks, which may greatly influence future use and stability of tenure (Primavera and Esteban, 2008).

The most appropriate use of mangroves in coastal defence is likely to be in combination with other risk reduction measures. For example, sea walls and levees placed on the landward side of mangrove forests are likely to experience reduced water levels and wave energy during storm surges, greatly reducing the likelihood of the wall being overtopped or damaged during a storm surge; this could significantly reduce the design specifications and therefore the cost of the sea wall (such combinations are sometimes referred to as 'hybrid engineering'). Another example discussed by Das and Vincent (2009) demonstrated how early warning systems and evacuation centres had the greatest effect on reducing the death toll during a cyclone in India, but mangroves further reduced the death toll among those people who did not evacuate.

As Williams *et al.* (2007) point out, it is not just the presence of mangroves which is required to provide coastal defence services, but good coastal planning. This can ensure that

evacuation plans and procedures are in place, that people are informed about these plans and procedures, and that they are willing to comply. When Cyclone Larry hit Australia in 2006, commercial, recreational and naval vessels in the port of Cairns sheltered in deep mangrove creeks. The protection given to the mangrove forests, and the careful planning that ensured that all vessel operators knew where and when to go, resulted in all vessels riding out the storm safely with no loss of life (Williams *et al.*, 2007).

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Attachment F
Historical Aerials

2017



2016



NORTH

2015



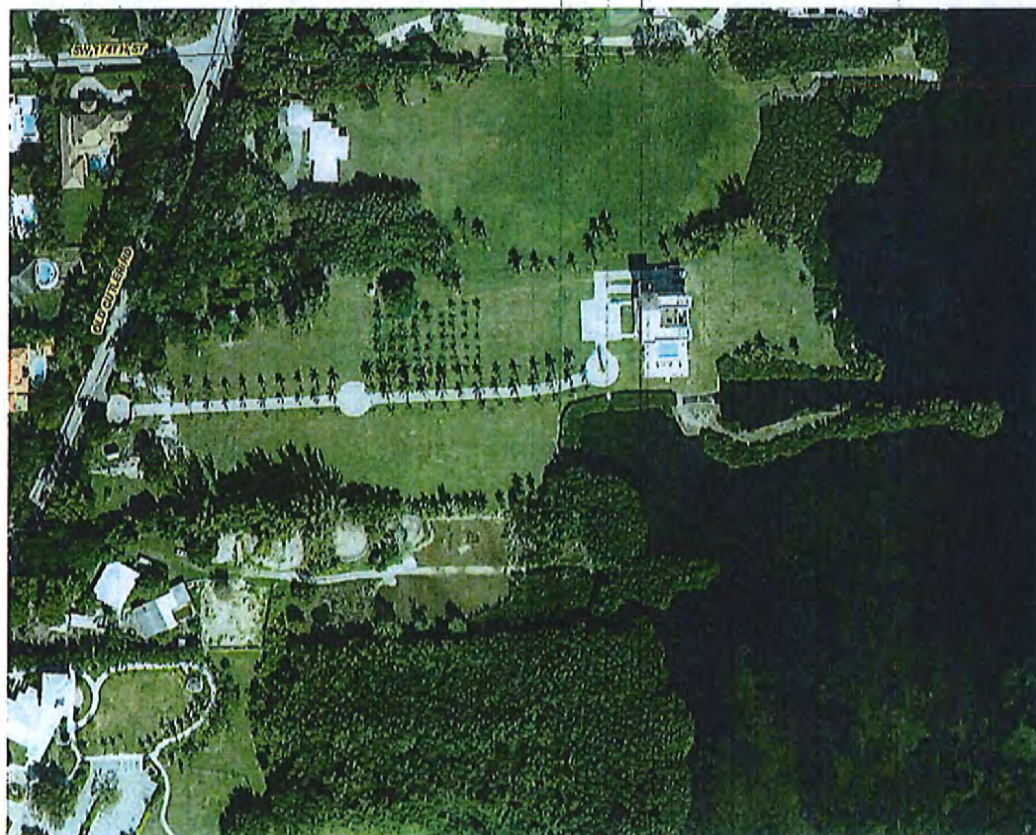
2014



2013



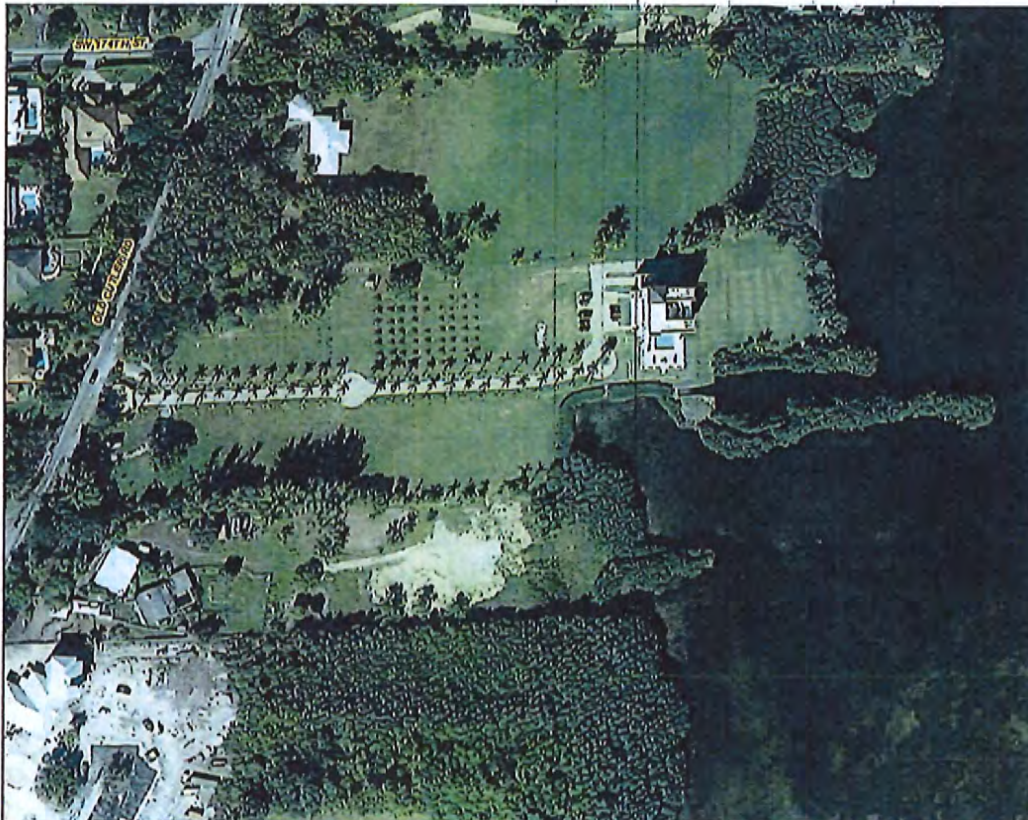
2012



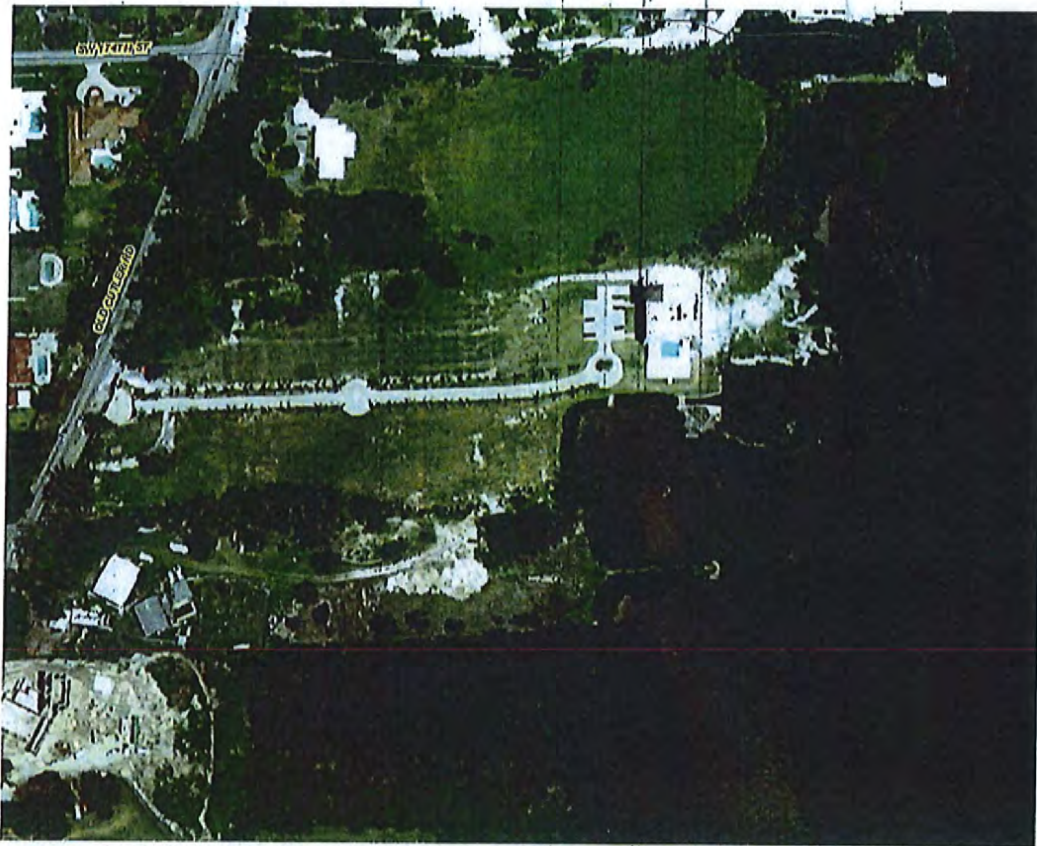
2010



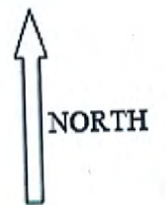
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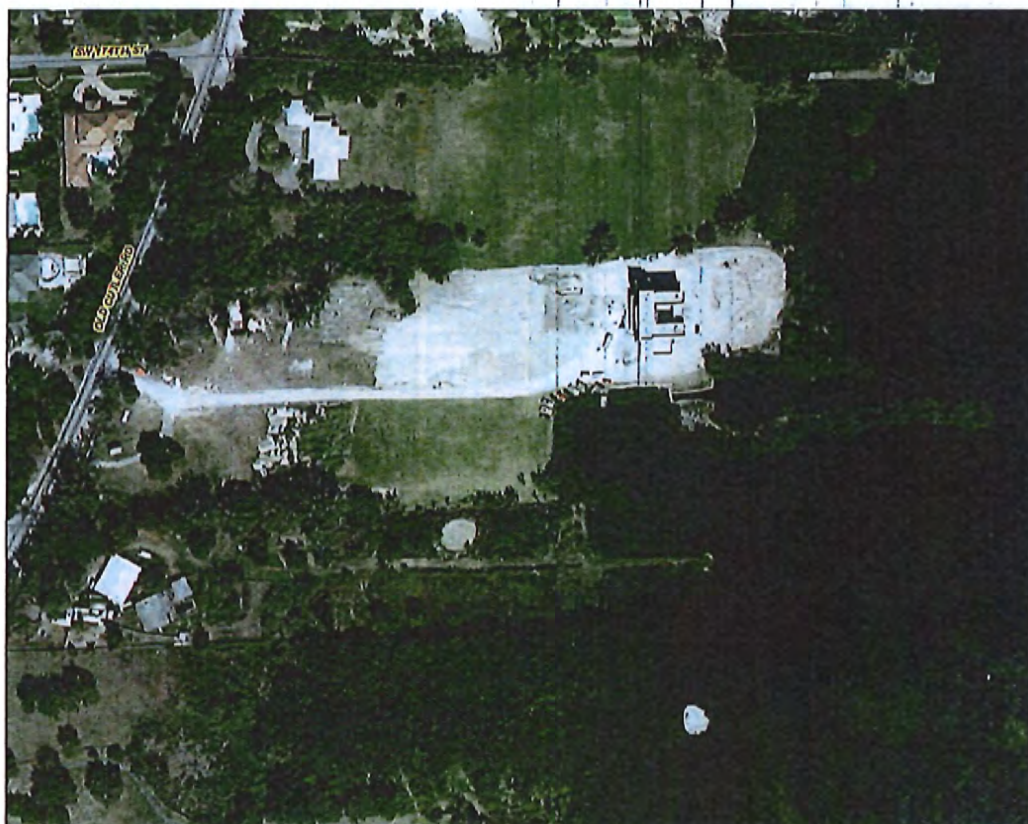
2007



2006



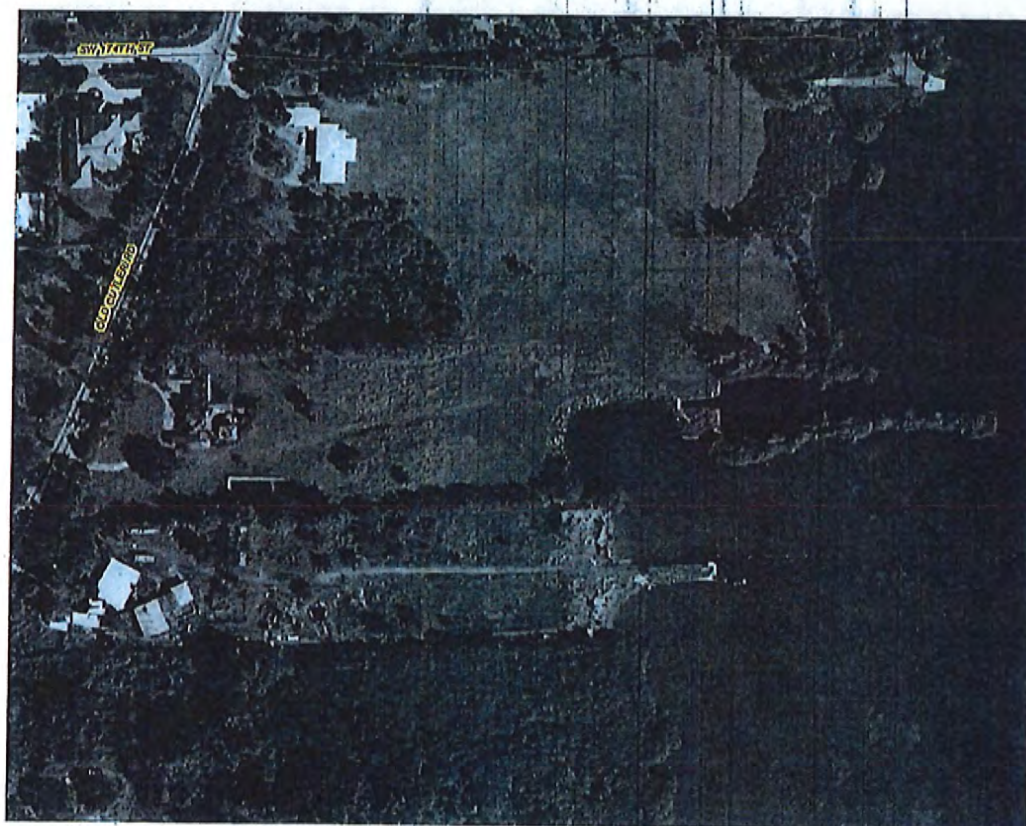
2005



2002



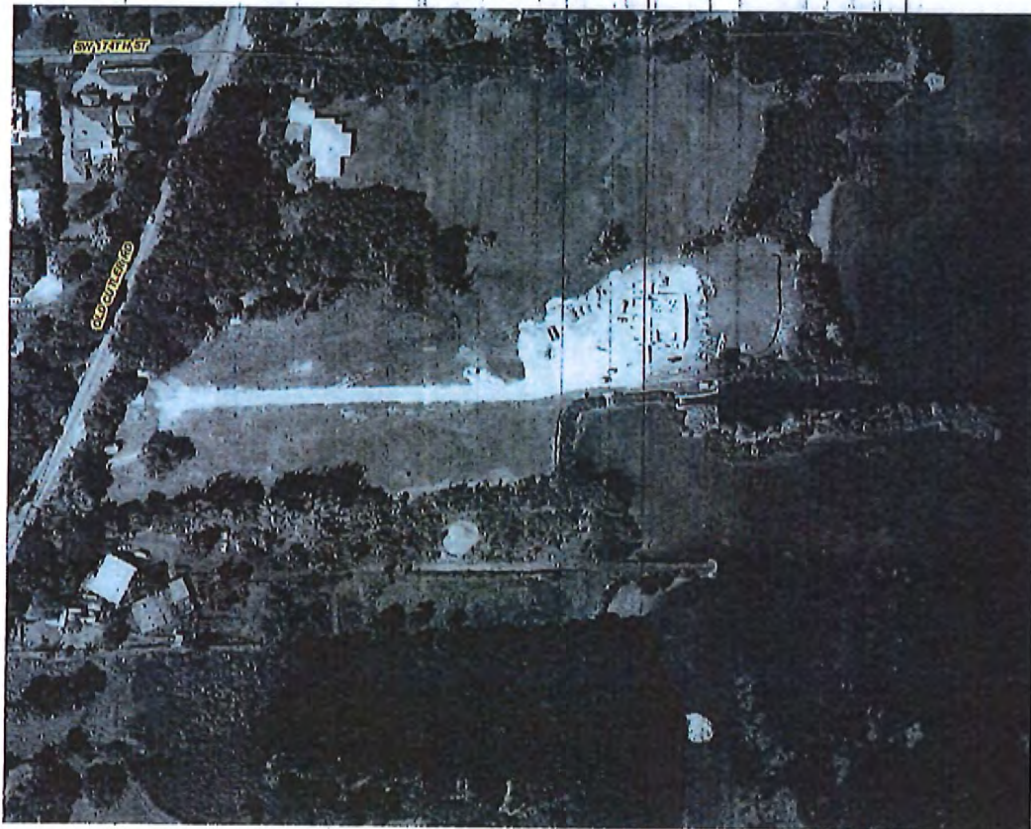
1999



2004



2003



NORTH

Attachment G

Excerpts from the Miami-Dade County Public Works Manual

SECTION C-3

REQUIREMENTS FOR FILLING LAND

A PORTION OF

PART 2 - PUBLIC WORKS MANUAL

SECTION C 3

REQUIREMENTS FOR FILLING LAND IN UNINCORPORATED DADE COUNTY

C3.01 GENERAL INFORMATION

1. PURPOSE

The following requirements are set forth as a means of controlling the quality and placement of fill necessary to bring land, including tidal areas, to the proper elevation. The information contained herein supplements Section 28-15(b)(4), 33-4, and 2-103 of the Code of Metropolitan Dade County.

2. GENERAL

The quality and placement of fill in the unincorporated areas of Dade County (including subdivisions and tidal areas) shall be in accordance with these requirements. The final fill, after settlement, shall be to an elevation no less than the approved elevation of the crown of the road fronting the property to be filled, or to an elevation no less than the elevation shown on the Dade County flood criteria maps, whichever is higher. Such elevations shall be referred to mean sea level, U.S. Coast and Geodetic Datum.

If the area to be filled is tidal or bay bottom, steps must be taken prior to the filling operation to acquire title to the land and obtain permits from the U.S. Army Corps of Engineers, the Trustees of the Internal Improvement Fund, and the County Commission.

If the area to be filled is within a subdivision, the fill material must be placed and accepted prior to recording the plat of the subdivision or a bond must be filed with the Public Works Department, in an amount approved by the Director of Public Works (not to exceed 110% of the estimated cost of the fill) to insure satisfactory completion of the fill, before the plat of the subdivision will be accepted for recording. Refer to Section 28-16(e) of the Code.

C3.02 QUALITY AND PLACEMENT OF FILL FOR PRIVATE LANDS

1. REMOVAL OF OVERBURDEN FROM EXISTING SURFACE

The removal of organic material or other undesirable overburden may be required by the Public Works Department prior to filling. Factors affecting the decision for removal will include; depth and quality of undesirable overburden, depth of fill to be placed above overburden, and proposed use of the area to be filled.

2. FILL MATERIAL FOR AREAS WHERE DEPTH OF FILL WILL BE MORE THAN TWO FEET

a. Acceptable Fill Material

- (1) A sand and rock mixture is considered to be the most desirable material for fill.
- (2) In lieu of sand and rock for the total fill, portions of the fill may be composed of reinforced concrete rubble masonry provided the largest dimension does not exceed three feet nor two-thirds the depth of fill, whichever is smaller. The reinforcing steel from reinforced concrete rubble shall not protrude more than twelve inches nor shall it project above the finished surface of the fill.

b. Unacceptable Fill Material

Peat, muck, wood, tree stumps, roots and branches, metal products or other material that will not readily compact into an unyielding mass shall not be used as fill.

3. FILL MATERIAL FOR AREAS WHERE DEPTH OF FILL WILL BE TWO FEET OR LESS OR FOR THE UPPER TWO FEET OF A FILL THAT WILL BE GREATER THAN TWO FEET

a. Acceptable Fill Material

Same as for fills greater than two feet, except that a percentage of the fill in lawn or park areas may be peat or muck sufficient to provide a good quality top soil.

b. Unacceptable Fill Material

Same as for fills greater than two feet, except for the peat or muck content, as described in a. above.

4. PLACEMENT OF UPPER THREE FEET OF FILL

In addition to the above requirements, the upper three feet of fill shall be of such quality that when placed it will provide natural surface seepage of an infiltration rate of one-inch per hour after saturation.

5. FILL PROTECTION

The extremities of an embankment that has been placed in accordance with these requirements and is not protected by bulkheading shall be protected against excessive erosion by rip-rapping or other acceptable preventive measures.

C3.03 FILLING OF AREAS DEDICATED OR PROPOSED TO BE DEDICATED AS PUBLIC RIGHT-OF-WAY FOR ROADS

All areas dedicated or proposed to be dedicated as public right-of-way for

road purposes shall be filled in accordance with the requirements as set forth in Section 30 of the Dade County Standard Road Specifications entitled "Excavation and Embankment."

C3.04 CONTROL OF WORK

All filling operations made pursuant to these requirements shall be performed under continuous supervision of an engineer, registered in the State of Florida. A certification from this engineer may be required at the completion of the work as an assurance that the fill has been placed in accordance with these requirements. A subdivision bond involving fill will not be released without such certification.

C3.05 BUILDING FOUNDATION

It is not to be construed that fill placed in accordance with these requirements will produce a foundation suitable for a building structure of any type. The foundation requirements for buildings are covered in Articles 2402.1, 2402.2, 2403.1, 2403.2, 2403.3, and 2403.4 of the South Florida Building Code. Building permits issued by Dade County for structures to be located on fill require an engineer's certification as to the bearing value of the soil at the building site. When the building site is known prior to filling, it would be advisable to place the building foundation in a manner that will satisfy the Building Code concurrent with filling the surrounding area.

C3.06 FILL PERMITS

Section 2-103.2 of the Code of Metropolitan Dade County requires that a permit issued by Public Works Department will be secured before placing fill on tidal or bay bottom land. The fee for such permit is specified in Section 2-103.2(b) of the Code.

SECTION D-5

COASTAL AND OTHER WATERFRONT CONSTRUCTION

A PORTION OF

PART 2 - PUBLIC WORKS MANUAL

SECTION D5 - COASTAL AND OTHER WATERFRONT CONSTRUCTION

D5.01 GENERAL INFORMATION

1. PURPOSE

It is the purpose of this section of the Public Works Manual to prescribe minimum standards for design and construction of coastal and other waterfront structures.

2. SCOPE

Requirements of this section apply to all coastal construction such as excavation, dredging, filling, and waterfront construction in, upon, or contiguous to tidal and baybottom lands in the unincorporated areas of Metropolitan Dade County and in, upon, or contiguous to those canals, lakes, bays, rivers, and all waterways under the direct control of Metropolitan Dade County whether or not in the unincorporated areas.

Said coastal construction includes, but is not limited to, excavation, dredging, filling, docks, piers, wharves, bridges, groins, jetties, moles, breakwaters, seawalls, revetments, causeways, artificial nourishment of beaches, quays, slips, moorings, marinas, ports, and related structures.

3. AUTHORITY

The Code of Metropolitan Dade County establishes the requirements and procedure for securing a permit to perform the various types of coastal construction.

4. OTHER REQUIREMENTS

In addition to the requirements set forth in this section, permits may be required from the U.S. Army Corps of Engineers, Trustees of the Internal Improvement Fund, Division of Beaches and Shores of the Florida Board of Conservation, Central and Southern Florida Flood Control District; and also the Dade County Building and Zoning Department, principally for private lands and private water areas not affected in any manner by establishment of the official bulkhead line or by State or Federal permits.

Construction in navigable waters requires a permit from the U.S. Army Corps of Engineers. Work that may affect the water-control system of the Central and Southern Florida Flood Control District will require a permit issued by that agency. Ultimate control of submerged or tidal lands rests with the Trustees of the Internal Improvement Fund.

Section D5 - Coastal and Other Waterfront Construction

TIIF permits cover tidal and baybottom dredging and filling and construction of open-trestle docks, piers, wharves, bridges, boat slips and houses, concessions, marinas and shipping facilities, and similar structures of permeable or open-type design--as contrasted with solid or highly impermeable structures likely to physically affect existing coastal conditions and the natural shoreline.

The Division of Beaches and Shores of the Florida Board of Conservation require permits covering solid or highly impermeable groins, jetties, moles, breakwaters, seawalls, revetments, causeways, and artificial beach nourishment or other deposition or removal of beach material--structures or activities which are likely to have a material physical effect on coastal conditions and the natural shoreline.

D5.02 DESIGN CRITERIA

1. GENERAL

All coastal structures shall be designed by an engineer registered in the State of Florida and all plans submitted to the Public Works Department for approval shall bear his signature and seal.

2. LOADS

Coastal structures are to be designed and constructed to resist the erosive and corrosive effects of the elements and where applicable to withstand the horizontal and vertical forces or loads listed below.

- a. Earth (Soil)
- b. Water
- c. Waves
- d. Wind
- e. Currents
- f. Weight of Structure
- g. Weight of Decks, Platforms or other Attachments
- h. Pedestrian Live Load (80 lbs. per square foot minimum).
- i. Vehicular Loading (AASHTO-H-20 minimum where vehicular loads are anticipated).
- j. Impact
- k. Material and Equipment Stored
- l. Anchors or Tiebacks
- m. Boats, Barges, or other Vessels

Section D5 - Costal and Other Waterfront Construction

3. ALLOWABLE STRESSES

Allowable stresses for structural design of steel, concrete, wood, and other materials shall be as specified in the South Florida Building Code.

D5.03 STRUCTURES—LOCATION, TYPE, GENERAL SPECIFICATIONS

Any type of construction not prohibited by the South Florida Building Code may be approved if it is adaptable to the site conditions and to the purpose of the structure.

If the proposed type or method of construction does not have an experience record sufficient to justify approval, the Public Works Department may require special tests or demonstrations to prove the acceptability of the project.

1. BULKHEADS AND SEAWALLS

a. Location

Bulkheads, revetments, and retaining walls shall normally be located to obtain uniformity of alignment and compatibility with the natural shoreline. They shall be placed on or upland of the mean high water line or such other line officially established for this or other set back purposes, and their location including revetments shall comply with local and state regulations for set back of structures along shorelines.

Bulkheads along canals or other waterways that are part of the official Dade County Water Control Plan shall be located to provide the required waterway section of the canal or waterway. They shall be continuous without abrupt change of direction and their location shall not be detrimental to adjacent property.

Bulkheads on lakes or similar water areas under the direct control of Dade County shall be on or landward of the water-fronting property line, or in the absence of such line shall be located on or landward of the ordinary high water line of the lake or water area; and if said lakes and water areas are privately owned, or if the proposed wall is located entirely on private property, then bulkhead and wall location requirements shall be as set forth in required permits from the County Building and Zoning Department. However, approval by the Building and Zoning Department pursuant to its applicable regulatory jurisdiction, shall not preclude or dispense with the necessity for all other permitting procedures required by other local and state regulations regarding such works.

Bulkheads proposed between two properties where bulkheads already exist shall be designed to connect such bulkheads. Bulkheads proposed adjacent to property not bulkheaded shall be designed to return along the side property line a distance sufficient to protect the back fill and prevent damage to adjacent property, but not less than 25 feet along the ocean and bay or 10 feet along canals, rivers, and other water areas. The return wall shall be protected from erosion by riprap or slope pavement.

Section D5 - Coastal and Other Waterfront Construction

b. Types of Walls

The use of vertical face bulkheads without a rock revetment will normally be prohibited, although exceptions may be allowed for projects such as canals, and for enclosed marinas, when it can be shown that equal or better conditions of environment or wave action can be attained with a vertical wall, and that such wall is necessary for the contemplated use, and is not within the Biscayne Bay Aquatic Preserve.

On the ocean front, seawalls along sand beaches subject to wave action, shall be an approved sloping high-energy absorbing type, or vertical walls with sloped energy-absorbing rock revetment on the face subject to wave action. The revetments slope shall be one vertical to two horizontal or flatter. Revetment composed of native limestone rock is considered desirable.

In bay areas bulkheads, seawalls, revetments, and retaining walls shall normally be sloped walls, or vertical walls with sloping rubble mound revetment on the water side. Such slopes shall be one vertical to two horizontal or flatter. Native limestone rock is considered desirable as revetment material.

Inside the Biscayne Bay Aquatic Preserve, bulkheads, seawalls, revetments, retaining walls and dikes shall meet all the foregoing requirements for bay areas, and in addition, provisions must be made in the wall, revetment, or dike for seepage and relatively free movement of groundwater from the upland side into the bay in order to maintain, insofar as feasible, natural shoreline conditions of groundwater flow.

Exceptions for bay areas and the Aquatic Preserve may be allowed in reference to the foregoing requirements, if permitted by existing laws, and whenever it can be shown that equal or better conditions of the bay environment can be attained by use of other type of walls, revetments, walls, or dikes, and that such other type structure is essential to the purpose for which it is to be built—for example, to retain the bank of coastal canals, or of enclosed marinas.

Whenever the beach in front of an existing vertical wall has eroded to such extent that water reaches the bottom of the wall at mean high tide, a sloping rock revetment conforming with the foregoing requirements shall be placed in front of the wall. Whenever any existing vertical wall located on the oceanfront or in bay areas is in need of major repairs, it shall not be repaired or replaced unless the foregoing requirements concerning a sloping face or revetment are first met. A proposed repair will be considered major when the total cost of repairs within the previous twelve (12) months, if any, plus the estimated cost of the proposed repair is more than fifty percent (50%) of the current cost of replacement.

Section D5 - Coastal and Other Waterfront Construction

c. General Specifications

All bulkheads shall have a concrete cap designed to withstand the various loads placed upon it. The cap shall be large enough to provide no less than four inches of concrete cover between the piles, panels or masonry and nearest exterior face of cap.

The elevation of the top of cap should be above the official flood criteria. (Such criteria provide for a minimum fill elevation, but not for storm wave heights.) Other cap elevations may be approved but only when land useage, proximity of buildings, and effect on adjacent property have been considered.

Safety curbs or guardrails shall be provided for bulkheads adjacent to roadways. Handrails shall be provided for bulkheads adjacent to walkways.

Cables or steel rods used in tiebacks must be protected by at least three inches of concrete encasement if the cable or rod is less than one inch in diameter. Tiebacks not encased in concrete are to be protected by coating and wrapping with bituminous or other corrosive-resistant material.

Anchors for tiebacks, whether piles or other types, shall bear on undisturbed or well compacted soil and shall be designed to provide adequate horizontal support.

Precast concrete panels of tee-pile and panel bulkheads shall have the foot of the panels placed in a manner that will prevent undermining of the backfill material.

Fill material placed on the water side of a bulkhead shall not be considered to offer any passive resistance when such fill is subject to erosion.

Gravity type bulkheads of stone and concrete combination will be permitted, provided they are constructed of no less than 40 percent cast in place concrete by cross sectional area and volume.

2. PIERS AND DOCKS

a. Location

Piers and docks at right angles to the shoreline, or nearly so, shall be located not closer to the side property line, or said line extended, than a distance equal to the length of the pier or dock itself, provided however, no such distance shall be less than 10 feet.

Section D5 - Coastal and Other Waterfront Construction

PIERS AND DOCKS (Continued)

Where the zoning is residential or where the area is subdivided into tracts smaller than one acre each, piers and docks are to be located within the middle one-half of the water frontage.

b. Types

Structures such as piers which are to project beyond the bulkhead line, if allowed, shall be of an open type construction.

Wharves, piers, or docks of solid fill construction will be approved only where such construction will not extend seaward of the approved bulkhead line.

c. General Specifications

In areas where the zoning is residential or in areas where no tract is larger than one acre, piers and docks shall be no more than 30 feet wide. In no case shall piers or docks obstruct navigation or interfere with drainage facilities. The projection of a pier or dock into a restricted waterway such as a canal, river, creek or basin shall be no greater than 10 feet or 20% of the waterway width, whichever is smaller. Public Works Department's approval may be given for piers projecting more than 10 feet into open water areas such as bays and sounds provided the projecting pier does not obstruct navigation or encroach upon the rights of adjacent property owners.

3. GROINS

a. Location

Groins are to be located so that the entire system of groins will provide the maximum benefit without adverse effects. Groins shall be anchored sufficiently landward to prevent flanking.

b. Types

Groins shall be either very low impermeable nonadjustable or impermeable adjustable, designed and maintained in adjustable condition for their entire life. The use of permeable groins shall be limited to special conditions.

Section D5 - Coastal and Other Waterfront Construction

GROINS (Continued)

c. General Specifications

Groins may be used to stabilize the beach if adjoining beaches are not adversely affected.

Groins should be impermeable, and adjustable to meet variations in natural conditions, and to produce the desired elevation of the beach.

Adjustable groins shall be maintained at elevations in accord with actual beach needs and development of desirable change of the beach profile, and so as to avoid damage to adjacent beaches. In no case shall the top of such groins be set higher than 2 feet above the beach profile. Impermeable, non-adjustable groins shall not extend seaward beyond the mean low water line, and their top elevation shall not be higher than 6 inches above the beach profile.

Groins must be constructed or adjusted low enough to provide pedestrian access between mean high and mean low water, or they must be provided with an adequate stairway for pedestrian access across them.

Consideration of the degree of beach protection to be provided by proposed groins, and the acceptability of such installations, will be based primarily on the following factors:

- Direction and Volume of Littoral Drift
- Wave Force and Direction
- Wind Force and Direction
- Land Usage
- Type of Bulkhead
- Type of Groin
- Spacing and Length of Groins

A complete coastal engineering study may be required before approval is given to the number, type, and location of groins.

4. BEACH NOURISHMENT

Artificial nourishment of sand beaches, or creation of new beach areas are treated as construction projects in the issuance of permits by the County and by the Division of Beaches and Shores of the Florida Board of Conservation. Typical profiles for such projects consist of a 50-foot level berm at elevation 6 ft. MSL; a 1 on 20 slope from there to MLW; and a 1 on 30 slope seaward to existing bottom.

Section D5 - Coastal and Other Waterfront Construction

Special agreement between the upland owner proposing such a project and the County (also between the owner and the State) may be required in order to adequately protect and permanently safeguard any public rights existing at the proposed site.

5. JETTIES AND BREAKWATERS

Jetties and breakwaters shall be designed in accordance with the latest issue of the U.S. Army Corps of Engineers' Technical Report No. 4 entitled "Shore Protection, Planning and Design".

6. MOORING PILES AND BUOYS

All mooring piles and buoys shall be placed within the limits of the owner's water frontage, and shall be located in a manner not to interfere with navigation. Outer mooring piles and buoys shall not constrict a navigable waterway except as permitted by the appropriate agency having jurisdiction over the waterway.

7. BOAT SLIPS AND BOATHOUSES

Boat slips and boathouses to be located on private property require approval and permit from the Building and Zoning Department. Bulkheads proposed to be constructed for retaining the banks of the boat slip shall meet the requirements of this section of the manual. The location of boat slips shall conform to the same requirements as for piers and docks.

Boat slip and bank-line excavations proposed downstream of salt dams and inland of the County salt barrier line require approval of the County Commission (after proper certification by the Public Works Department).

Boathouses may be constructed over boat slips or as a separate structure subject to the following conditions:

- a. The boathouse may not be used as a dwelling, guest house or servant's quarters unless specially constructed as such to the requirements of the Building and Zoning Department.
- b. The boathouse does not extend into a water area a distance greater than that permitted for a dock or pier.
- c. The overall size of the boathouse does not exceed 25 feet in width, 45 feet in length, or 18 feet in height, except commercial marinas and drydocks may be permitted larger boathouses constructed in compliance with applicable zoning and building regulations.

8. BRIDGES AND CULVERTS

The design of bridges and culverts shall be in accord with Section D6 of this manual.

Section D5 - Coastal and Other Waterfront Construction

D5.04 MATERIALS

1. FILES

a. Wood Piles

Wood piles may be used in bulkheads only to support or anchor tiebacks and only when cut off or capped in concrete one foot below mean low water. Wood piles are not to be used in groins.

Wood piles are to comply with Article 2404.3 of the South Florida Building Code.

b. Steel Piles

Piles of rolled structural steel shapes shall comply with Article 2404.7 of the South Florida Building Code.

Steel sheet piles shall meet the requirements of ASTM Designation A328 and shall be at least 3/8-inch thick.

Steel pipe, if used as piling, shall be filled with concrete and shall meet the requirements of AASHTO Specifications M94 and shall be at least 3/16-inch thick.

c. Concrete Piles

Concrete piles, whether cast in place, precast or prestressed, shall meet the requirements of Articles 2404.4, 2404.5, and 2404.6 of the South Florida Building Code.

d. File Foundations

When piles are used as foundations for coastal structures, they shall meet the requirements of Articles 2404.1 and 2404.2 of the South Florida Building Code.

2. CONCRETE

Concrete used in coastal structures shall be designed, mixed, transported and placed to meet the requirements of Chapter 25 of the South Florida Building Code.

Concrete to be placed under water shall have a slump between four and eight inches, a minimum cement content of 7.0 bags per cubic yard and a maximum net water cement ratio of 6.0 gallons per bag.

To prevent segregation--concrete to be placed under water shall be placed carefully in a compact mass in its final position by means of a tremie or other approved method.

Concrete shall not be placed in running water.

Concrete seals shall be placed in one continuous operation.

Section D5 - Coastal and Other Waterfront Construction

The tremie used for placing concrete shall consist of a tube at least 10 inches in diameter, constructed in sections and shall have flanged couplings fitted with gaskets.

3. STEEL

a. Reinforcing Steel

Reinforcing steel shall meet the requirements of Article 2504.3 of the South Florida Building Code. The use of special large size bars meeting ASTM Designation A431 and A432 is permitted.

b. Steel Wire Fabric

Steel wire fabric shall meet the requirements of ASTM Designation A185.

c. Structural Steel

Structural steel shall meet the requirements of ASTM Designation A7.

d. Nuts and Bolts

Nuts and bolts shall meet the requirements of ASTM Designation A307.

e. Steel Tieback Rods

Steel tieback rods may be either reinforcing steel or stress relieved strands or stress relieved wire, meeting the requirements of ASTM Designation A416 and A421.

f. Steel Fasteners for Wood

Steel fasteners for wood shall meet the requirements of the applicable portions of Chapter 29 of the South Florida Building Code.

4. WOOD

Timbers and wood structural members shall meet the requirements of AASHTO Specifications M168.

5. STONE

Stone used for revetments for bulkheads, or the construction of groins, jetties or breakwaters shall be dense, hard, and durable.

The size, range of sizes, and gradation shall be selected to meet the requirements of the individual situation and site.

6. FILL MATERIAL

Material for fills shall comply with Section C3 of this manual.

Section D5 - Coastal and Other Waterfront Construction

D5.05 INFORMATION REQUIRED ON AND FOR THE PREPARATION OF CONSTRUCTION PLANS

Construction plans must be prepared by an engineer registered in Florida. They shall be on sheet size 22"x36", arranged and numbered as a set and contain all (or applicable portions) of the following:

- (1) Plan, elevation, and sections showing the complete structure.
- (2) Details of structural components including precast members, structural connections, steel reinforcement, and expansion joints.
- (3) Complete description of all materials to be used.
- (4) Design loading and minimum penetration of piles.
- (5) Location control.
 - (a) Horizontal control referred to a section line, road, or permanent landmark, and including property lines and the Official Bulkhead Line.
 - (b) Vertical control referred to U.S. Coast and Geodetic Survey Datum (MSL) including elevations landward, soundings in water areas, and the mean high water line.
- (6) Graphical representation of test borings or soil profile parallel to and within five feet of proposed structures.

D5.06 PRELIMINARY APPROVAL OF CONSTRUCTION PLANS

The design engineer, when requesting preliminary approval of construction plans, shall submit:

1. Three (3) complete sets of construction plans and specifications.
2. Test boring data as may be required to determine the adequacy of the structure's foundation.
3. Design calculations if requested by the Public Works Department.

Section D5 - Coastal and Other Waterfront Construction

If the Public Works Department finds that the design and construction plans are in accord with this section, two sets of the plans will be returned stamped "APPROVED." Such approval does not constitute a construction permit.

D5.07 CONSTRUCTION PERMIT

A permit for construction may be issued to a certified contractor provided:

1. Four (4) sets of construction plans, identical to the approved plans, are presented.
2. Satisfactory evidence is presented that all other applicable approvals and permits have been obtained.
3. The required permit fee is paid.
4. A performance bond has been provided, in an amount determined by the Director of the County Public Works Department but not to exceed 100% of the estimated cost of the structure.
5. It is certified that a registered civil engineer has been retained to provide engineering supervision throughout the construction period.

D5.08 INFORMATION REQUIRED DURING CONSTRUCTION

The design engineer may be required to furnish the Public Works Department any portion of the following information during construction:

1. A complete pile driving log.
2. A report on the manufacture of all precast members including the stressing operation of prestressed members.
3. Test reports from a certified laboratory on all concrete used, including precast members.
4. Mill certificates for structural and reinforcing metals used.

D5.09 INFORMATION REQUIRED BEFORE FINAL ACCEPTANCE

When the structure is complete, and prior to final acceptance, the design engineer shall furnish the County (and other public agencies which may have issued permits) a complete set of As-Built drawings, together with his certification that the structure has been built in accord with the approved plans and specifications as finally revised.

Section D5 - Coastal and Other Waterfront Construction

D5.10 MAINTENANCE

These minimum requirements are not to be construed as an implication that Dade County proposes to maintain these structures when completed. A coastal structure shall be maintained at all times by the owner including any portion thereof that may extend into public property. All structures that are available for public use shall be maintained by the owner in a safe condition. Maintenance of groins will include periodic adjustments of the height to compensate for changes in the beach profile.

Attachment H

Class I Permit Application

MIAMI-DADE
COUNTY

RECEIVED

Class I
Permit Application

LB

FOR DEPARTMENTAL USE ONLY	
Date Received: DEC 13 2018	Application Number: CLI 2018-0467
NATURAL RESOURCES DIVISION DEPARTMENT OF REGULATORY AND ECONOMIC RESOURCES	Application Fee: \$845.00

Application must be filled out in its entirety. Please indicate N/A for non-applicable fields.

1. Applicant Information: Name: John Dubois Address: 18495 S. Dixie Hwy PMB 107 Cutler Bay, FL Zip Code: 33157 Phone #: 305-588-4000 Fax #: Email: john@eyecast.com * This should be the applicant's information for contact purposes.	2. Applicant's Authorized Permit Agent: Agent is allowed to process the application, furnish supplemental information relating to the application and bind the applicant to all requirements of the application. Name: Jeffrey P. Leary, Esq. Address: 18495 S. Dixie Hwy #439 Cutler Bay, FL Zip Code: 33157 Phone #: (585)-747-2952 Fax #: N/A Email: mdepr.jl@gmail.com
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3. Location where proposed activity exists or will occur (latitude and longitude are only necessary for properties without address or folio #):	
Folio #(s): 33-5035-001-0180	Latitude: 25°36'29.31" N Longitude: 80°18'30.84" W
Street Address: 17575 Old Cutler Rd	Section: 35 Township: 55 Range: 40
In City or Town: Village of Palmetto Bay	Near City or Town:
Name of waterway at location of the activity: Biscayne Bay	

4. Describe the proposed activity (check all that apply):	
☒ Seawall ☒ New/Replacement Seawall ☒ Seawall Cap ☒ Batter Piles ☒ King Piles ☒ Footer/Toe Wall ☒ Riprap	☐ Dock(s) ☐ Pier(s) ☐ Viewing Platform ☐ Boatlift ☐ Mooring Piles ☐ Fender Piles ☐ Davits ☐ Dredging ☐ Maintenance ☐ New ☒ Filling ☐ Mangrove Trimming ☐ Mangrove Removal
☐ Other:	
Estimated project cost = \$ 25,000	
Are you seeking an after-the-fact approval (ATF)? ☐ Yes ☒ No If "Yes", describe the ATF work:	

5. Proposed Use (check all that apply): ☒ Single Family ☐ Multi-Family ☐ Private ☐ Public ☐ Commercial ☐ Industrial ☐ Utility	6. If the proposed work relates to the mooring of vessels provide the following information (please also indicate if the applicant does not have a vessel): Proposed Vessel Type (s): N/A Vessel Make/Model (if known): N/A Draft (s)(range in inches.): N/A Length (s)(range in feet.): N/A Total Number of Slips: N/A
---	--

7. List all permits or certifications that have been applied for or obtained for the above referenced work:				
Issuing Agency	Type of Approval	Identification Number	Application Date	Approval Date
State of Florida DEP US ACOE	Environmental Resource Permit	13-0336703-002-E1		12.16.16
* See attached Letter of Intent				

8. Contractor Information (If known):

Name: TBD License # (County/State): _____
Address: _____ Zip Code: _____
Phone #: _____ Fax #: _____ E-mail: _____

9. IMPORTANT NOTICE TO APPLICANTS: The written consent of the property owner is required for all applications to be considered complete. Your application **WILL NOT BE PROCESSED** unless the Applicant and Owner Consent portion of the application is completed below. You have the obligation to apprise the Department of any changes to information provided in this application.

Application is hereby made for a Miami-Dade County Class I permit to authorize the activities described herein. I agree to or affirm the following:

- I possess the authority to authorize the proposed activities at the subject property, and
- I am familiar with the information, data and plans contained in this application, and
- To the best of my knowledge and belief, the information, data and plans submitted are true, complete and accurate, and
- I will provide any additional information, evidence or data necessary to provide reasonable assurance that the proposed project will comply with the applicable State and County water quality standards both during construction and after the project is completed, and
- I am authorizing the permit agent listed in Section 2 of this application to process the application, furnish supplemental information relating to this application and bind the applicant to all requirements of this application, and
- I agree to provide access and allow entry to the project site to inspectors and authorized representatives of Miami-Dade County for the purpose of making the preliminary analyses of the site and to monitor permitted activities and adherence to all permit conditions.

A. IF APPLICANT IS AN INDIVIDUAL

John Dubois
Signature of Applicant

John Dubois
Print Applicant's Name

12/10/2018
Date

B. IF APPLICANT IS OTHER THAN AN INDIVIDUAL OR NATURAL PERSON
(Examples: Corporation, Partnership, Trust, LLC, LLP, etc.)

Print Name of Applicant (Enter the complete name as registered)
Registration/Incorporation _____ Type (Corp, LLC, LLP, etc.) _____ State of _____

Under the penalty of perjury, I certify that I have the authority to sign this application on behalf of the Applicant, to bind the Applicant, and if so required to authorize the issuance of a bond on behalf of the Applicant. (If asked, you must provide proof of such authority to the Department). *****Please Note: If additional signatures are required, pursuant to your governing documents, operating agreements, or other applicable agreements or laws, you must attach additional signature pages.*****

Signature of Authorized Representative _____ Print Authorized Representative's Name _____ Title _____ Date _____

C. IF APPLICANT IS A JOINT VENTURE Each party must sign below (If more than two members, list on attached page)

Print Name of Applicant (Enter the complete name as registered)
Registration/Incorporation _____ Type (Corp, LLC, LLP, etc.) _____ State of _____

Print Name of Applicant (Enter the complete name as registered)
Registration/Incorporation _____ Type (Corp, LLC, LLP, etc.) _____ State of _____

Under the penalty of perjury, I certify that I have the authority to sign this application on behalf of the Applicant, to bind the Applicant, and if so required to authorize the issuance of a bond on behalf of the Applicant. (If asked, you must provide proof of such authority to the Department). *****Please Note: If additional signatures are required, pursuant to your governing documents, operating agreements, or other applicable agreements or laws, you must attach additional signature pages.*****

Signature of Authorized Representative _____ Print Authorized Representative's Name _____ Title _____ Date DEC 13 2018

Signature of Authorized Representative _____ Print Authorized Representative's Name _____ Title _____
NATURAL RESOURCES DIVISION
DEPARTMENT OF REGULATORY
AND ECONOMIC RESOURCES
Revised 11/16/09

10. WRITTEN CONSENT OF THE PROPERTY OWNER OF THE AREA OF THE PROPOSED WORK

I/We are the fee simple owner(s) of the real property located at 17575 Old Cutler Rd Miami-Dade
County, Florida, otherwise identified in the public records of Miami-Dade County as Folio No. 33-5035-001-0180.

I am aware and familiar with the contents of this application for a Miami-Dade County Class I Permit to perform the work on or adjacent to the subject property, as described in Section 4 of this application. I possess the riparian rights to the area of the proposed work (if applicable) and hereby consent to the work identified in this Class I Permit application.

A. IF THE OWNER(S) IS AN INDIVIDUAL

John Dubois 12/10/2018
Signature of Owner Print Owner's Name Date

Signature of Owner Print Owner's Name Date

B. IF THE OWNER IS OTHER THAN AN INDIVIDUAL OR NATURAL PERSON
(Examples: Corporation, Partnership, Joint Venture, Trust, LLC, LLP, etc.)

Print Name of Owner (Enter the complete name as registered) Type (Corp, LLC, LLP, etc.) State of Registration/Incorporation

Address of Owner

Under the penalty of perjury, I certify that I have the authority to sign this application on behalf of the Owner, to bind the Owner, and if so required to authorize the issuance of a bond on behalf of the Owner, (If asked, you must provide proof of such authority to the Department). *****Please Note: If additional signatures are required, pursuant to your governing documents, operating agreements, or other applicable agreements or laws, you must attach additional signature pages.*****

Jeffrey P. Leary, Esq. Attorney
Signature of Authorized Representative Print Authorized Representative's Name Title Date

Signature of Authorized Representative Print Authorized Representative's Name Title Date

Please Review Above

Appropriate signature(s) must be included in:

Box 9: either A, B or C

AND

Box 10: either A or B

RECEIVED
DEC 13 2018
NATURAL RESOURCES DIVISION
DEPARTMENT OF REGULATORY
AND ECONOMIC RESOURCES

Attachment I

Owner/Agent Letter, Engineer Letter, and Project Sketches

PERMIT APPLICANT / AUTHORIZED AGENT STATEMENTNote: Please insert applicable informationDate: 9/11, 2019

To:
 Miami Dade County RER
 Class I Permitting Program
 701 NW 1st Court
 Miami FL, 33136

Re: Class I Permit Application Number CLI, 2018-0467
Proposed Shoreline Restoration Project (insert description of proposed work)

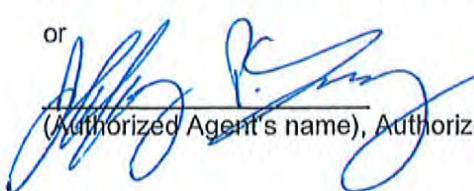
By the attached Class I permit application with supporting documents, I,
Jeffrey P. Leary, Esq. (insert name), am the permit applicant / applicant's authorized (circle
 one) agent and hereby request permission to perform the following: Seek approval of Class I Permit
Application CLI 2018-0467 (describe proposed
 work). I understand that a Miami-Dade County Class I Standard Form Permit is
 required to perform this work.

If approval is granted for the proposed work by the Board of County Commissioners,
 complete and detailed plans and calculations of the proposed work shall be prepared by
 an engineer licensed in the State of Florida in accordance with the minimum
 requirements of Chapter 24 of the Code of Miami-Dade County, Florida. Said plans and
 calculations shall be subject to the review and approval of the Department. The permit
 applicant will secure the services of an engineer licensed in the State of Florida to
 conduct inspections throughout the construction period, and said engineer shall prepare
 all required drawings of record. In the event that the proposed work which is the subject
 of this Class I Permit application involves the cutting or trimming of a mangrove tree(s),
 a detailed plan of the proposed cutting or trimming shall be prepared by a licensed
 landscape architect and submitted to the Department for review and approval, and the
 permit applicant will secure the services of a licensed landscape architect to supervise
 the trimming or cutting.

Respectfully submitted,

 (Permit Applicant's name), Permit Applicant

or


 (Authorized Agent's name), Authorized Agent

ENGINEER LETTER OF CERTIFICATION

Date: 11-20-19

Miami-Dade County Department of Regulatory and Economic Resources
Class I Permitting Program
701 NW 1st Court
Miami, Florida 33136

RE: Class I Permit Application Number CLI-2018-0467

Ladies and Gentlemen:

This letter will certify that I am an engineer licensed in the State of Florida, qualified by education and experience in the area of engineering design and inspection, and that to the best of my knowledge and belief, the proposed work does not violate any laws, rules, or regulations of the State of Florida or any provisions of the Code of Miami-Dade County which may be applicable; that diligence and recognized standard practices of the engineering profession have been exercised in the engineer's design of the proposed work; and in my opinion based upon my knowledge and belief, the following will not occur:

- a. Harmful obstruction or undesirable alteration of the natural flow of the water within the area of the proposed work.
- b. Harmful or increased erosion, shoaling of channels or stagnant areas of water. (Not applicable to class IV permits)
- c. Material injury to adjacent property.
- d. Adverse environmental impacts from changes in water quality or quantity. (Applicable to class IV permits only)

Further, I have been retained by the applicant to provide inspections throughout the construction period and to prepare a set of reproducible record prints of drawings showing changes made during the construction process based upon the marked-up prints, certified surveys, drawings, and other data furnished by the contractor to me.

Sincerely,

Jose Mendez

P.E. # 53288





LOCATION MAP AND LEGAL DESCRIPTION



PROJECT SITE LOCATION:
17575 Old Cutler Road
Miami, FL 33157

LATITUDE: 25°36'29.31"N
LONGITUDE: 80°18'30.25"W

FOLIO No.: 33-5035-001-0180

LEGAL DESCRIPTION:
35 55 40 6.15 AC
RICHMONDS SURVEY PB B-17
ALL OF BLKS 108-109 & 110 LYING
ELY OF HWAY LESS N75FT AS PER
PB 3-166
LOT SIZE IRREGULAR
OR 19241-2770-71 072000 2 (2)
F/A/U 30-5035-001-0180

GENERAL NOTES:

- ELEVATIONS SHOWN REFER TO THE NATIONAL GEODETIC VERTICAL DATUM (NGVD) OF 1929.
- ALL DIMENSIONS ON PLANS ARE SUBJECT TO VERIFICATION BY THE CONTRACTOR.
- IT IS THE INTENT OF THESE PLANS TO BE IN ACCORDANCE WITH APPLICABLE CODES AND AUTHORITIES HAVING JURISDICTION. ANY DISCREPANCIES BETWEEN THESE PLANS AND APPLICABLE CODES SHALL BE IMMEDIATELY BROUGHT TO THE ATTENTION OF ENGINEER BEFORE PROCEEDING WITH WORK. CONTRACTOR AND ALL SUBCONTRACTORS ARE RESPONSIBLE FOR ALL LINES, ELEVATIONS, AND MEASUREMENTS IN CONNECTION WITH THEIR WORK.
- IT IS THE INTENT OF THESE PLANS AND THE RESPONSIBILITY OF THE CONTRACTOR TO COMPLY WITH LOCAL, STATE, AND FEDERAL ENVIRONMENTAL PERMITS ISSUED FOR THIS PROJECT. IT SHALL BE THE CONTRACTOR'S RESPONSIBILITY TO FAMILIARIZE AND GOVERN HIMSELF BY ALL PROVISIONS OF THESE PERMITS.
- APPROVED CONTRACTOR SHALL BE RESPONSIBLE FOR THE ACCURACY OF EXISTING STRUCTURES AND VERIFY ALL DIMENSIONS. THE APPROVED CONTRACTOR IS RESPONSIBLE FOR ALL METHODS, MEANS, SEQUENCES AND PROCEDURES OF WORK.
- DO NOT SCALE DRAWINGS FOR DIMENSIONS.
- CONTRACTOR TO VERIFY LOCATION OF EXISTING UTILITIES PRIOR TO COMMENCING WORK.
- CONTRACTOR TO PROPERLY FENCE AND SECURE AREA WITH BARRICADES.
- ANY DEVIATION AND/OR SUBSTITUTION FROM THE INFORMATION PROVIDED HEREIN SHALL BE SUBMITTED TO THE ENGINEER FOR APPROVAL PRIOR TO COMMENCEMENT OF WORK.
- NEW MATERIALS AND/OR PATCHWORK SHALL BE PROVIDED TO MATCH EXISTING MATERIALS AND/OR EXISTING WORK.
- CONTRACTOR SHALL USE ALL ACCEPTABLE METHODS TO PROTECT ALL EXISTING MATERIALS, SURFACES, AND FINISHINGS FROM DAMAGE DURING ALL PHASES OF CONSTRUCTION.
- THE LICENSED CONTRACTOR TO INSTALL AND REMOVE ALL SHORING AND BRACING AS REQUIRED FOR THE PROPER EXECUTION OF THE WORK.
- ALL NEW WORK AND/OR MATERIALS SHALL CONFORM TO ALL REQUIREMENTS OF EACH ADMINISTRATIVE BODY HAVING JURISDICTION IN EACH PERTAINING CIRCUMSTANCE.
- THE CONTRACTOR WILL EMPLOY AND MAINTAIN ADEQUATE SEDIMENT AND EROSION CONTROL MEASURES TO PROTECT BISCAYNE BAY FROM SEDIMENT AND CONSTRUCTION DEBRIS.

BOLTS

- ALL BOLTS SHALL BE STAINLESS STEEL, UNLESS OTHERWISE NOTED.

CONCRETE

- CONCRETE SHALL CONFORM TO ACI 318 (LATEST EDITION) AND SHALL BE REGULAR WEIGHT, SULFATE RESISTANT, WITH A DESIGN STRENGTH OF 5,000 PSI AT 28 DAYS WITH A MAX WATER-CEMENTitious MATERIALS RATIO, BY WEIGHT, NORMAL WEIGHT AGGREGATE CONCRETE OF 0.40.
- OWNER SHALL EMPLOY AND PAY FOR TESTING SERVICES FROM AN INDEPENDENT TESTING LABORATORY FOR CONCRETE SAMPLING AND TESTING IN ACCORDANCE WITH ASTM.
- LICENSED CONTRACTOR IS RESPONSIBLE FOR THE ADEQUACY OF FORMS AND SHORING AND FOR SAFE PRACTICE IN THEIR USE AND REMOVAL.
- CONCRETE COVER SHALL BE 3" UNLESS OTHERWISE NOTED ON APPROVED DRAWINGS.
- REINFORCING STEEL SHALL BE IN CONFORMANCE WITH THE LATEST VERSION OF ASTM A615 GRADE 60 STEEL. ALL REINFORCEMENT SHALL BE PLACED IN ACCORDANCE WITH ACI 315 AND ACI MANUAL OF STANDARD PRACTICE.
- SPLICES IN REINFORCING BARS SHALL NOT BE LESS THAN 48 BAR DIAMETERS AND REINFORCING SHALL BE LOCATED AROUND ALL CORNERS AND CHANGES IN DIRECTION. CONTINUITY SHALL BE PROVIDED AT CORNERS OR CHANGES IN DIRECTION BY BENDING THE LONGITUDINAL STEEL AROUND THE CORNER 48 BAR DIAMETERS.

PILE DRIVING NOTES

- PILE DRIVING OPERATIONS SHALL BE OBSERVED BY A SPECIAL INSPECTOR, INCLUDING TEST PILES SUFFICIENT TO DETERMINE THE APPROXIMATE LENGTH REQUIRED TO MEET DESIGN CAPACITY.
- PILES SHALL BE DRIVEN USING AN APPROVED CUSHION BLOCK, CONSISTING OF MATERIAL SO ARRANGED SO AS TO PROVIDE THE TRANSMISSION OF THE HAMMER ENERGY.
- PILES SHALL BE DRIVEN TO REQUIRED CAPACITY A MIN 12 INTO BERM.
- PILES SHALL BE DRIVEN WITH A DROP HAMMER OR GRAVITY HAMMER PROVIDED THE HAMMER SHALL WEIGH NO LESS THAN 3,000 POUNDS, AND THE FALL OF THE HAMMER SHALL NOT EXCEED 6 FT.
- PILES SHALL BE DRIVEN WITH A VARIATION OF NOT MORE THAN 2 PER FOOT FROM THE VERTICAL OR FROM THE BATTER LINE INDICATED, WITH A MAX VARIATION OF THE HEAD OF THE PILE FROM THE PORTION SHOWN ON THE PLANS OF NOT MORE THAN 2".
- WHERE PILING MUST PENETRATE STRATA OFFERING HIGH RESISTANCE TO DRIVING, THE STRUCTURAL ENGINEER OF RECORD OR SPECIAL INSPECTOR MAY REQUIRE THAT PILES BE SET IN PRE-DRILLED OR PUNCHED HOLES. THE PILES SHALL REACH THEIR FINAL PENETRATION BY DRIVING.

TURBIDITY CURTAIN NOTES:

- A TURBIDITY CURTAIN WILL BE USED AS APPROPRIATE TO MAINTAIN AMBIENT WATER QUALITY DURING ALL CONSTRUCTION ACTIVITIES.
- THE TURBIDITY CURTAIN WILL BE USED AS APPROPRIATE TO MAINTAIN AMBIENT WATER QUALITY DURING ALL CONSTRUCTION ACTIVITIES.
- THE TURBIDITY CURTAIN WILL BE WEIGHTED AND EXTEND TO WITHIN ONE FOOT OF THE SUBMERGED WATERWAY BOTTOM.
- THE TURBIDITY CURTAIN WILL BE OUTSIDE OF ANY NAVIGABLE CHANNEL.

17575 OLD CUTLER ROAD SHORELINE RESTORATION PROJECT

CLIENT:
Miami, Florida
MR. JEFFERY LEARY, ESQ.
1809 S. Dixie Hwy.
PMB 107
Miami, FL 33157

ENVIRONMENTAL CONSULTANT:
OCEAN
CONSULTING, LLC
340 Minerva Avenue, Suite 7
Coral Gables, Florida 33134
Tel: (305) 921-8244
Fax: (305) 677-3254

CONTRACTOR:

PROJECT ENGINEER:
MENDEZ PROFESSIONAL
ENGINEERING CORP
1385 Coral Way, Ste 205
Miami, FL 33145
Phone: (305) 456-8338

SEAL / SIGNATURE / DATE

Jose A. Mendez
P.E. FL 53288, CA 08324

PERMIT DRAWING

Issue # Issue Date
① January 30, 2017
② May 14, 2019
③ January 13, 2020

PROJECT: 15-4915

PROJECT LOCATION & NOTES

SCALE: AS SHOWN
SHEET NO.

S-1

S-2



S-2.1



**17575 OLD CUTLER ROAD
SHORELINE
RESTORATION PROJECT**

CLIENT:
Miami, Florida
MR. JEFFERY LEARY, ESQ.
18495 S. Dixie Hwy.,
PMB 107
Miami, FL 33157

ENVIRONMENTAL CONSULTANT:
**OCEAN
CONSULTING, LLC**
340 Minorca Avenue, Suite 7
Coral Gables, Florida 33134
Tel: (305) 921-9944
Fax: (305) 677-3254
CONTRACTOR:

PROJECT ENGINEER:
**MENDEZ PROFESSIONAL
ENGINEERING CORP**
1385 Coral Way, Ste 203
Miami, FL 33145
Phone: (305) 450-8238

SEAL / SIGNATURE / DATE

Jose A. Mendez
P.E. FL 51288, CA 08324

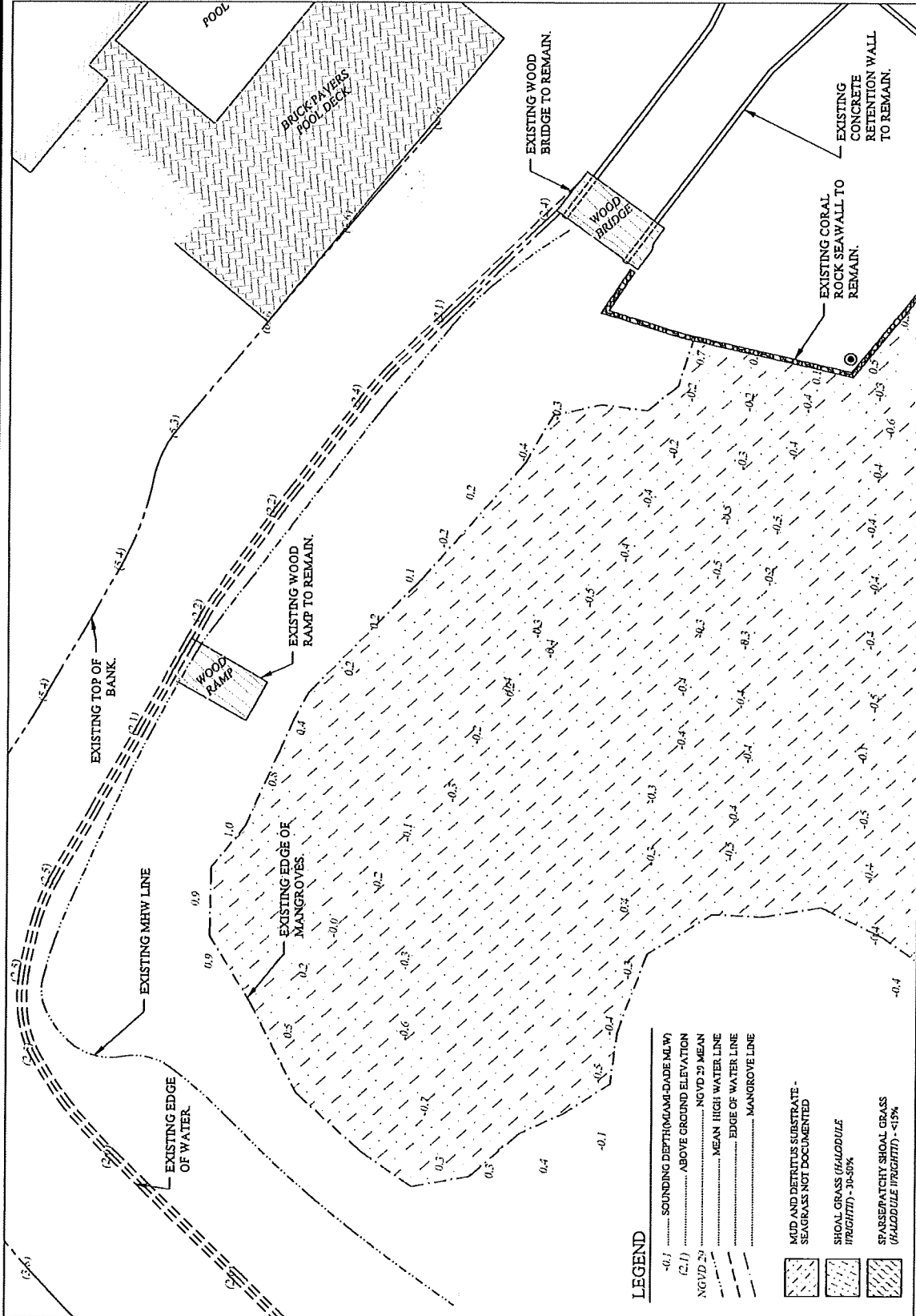
PERMIT DRAWING

Issue # Issue Date
① January 30, 2017
② May 14, 2019
③ January 13, 2020

PROJECT: 15-4915

**EXISTING HYDROGRAPHIC
DATA AND SITE CLOSE-UP**
SCALE: AS SHOWN
SHEET NO.

S-2.2



17575 OLD CUTLER ROAD
SHORELINE
RESTORATION PROJECT

CLIENT:
MR. JEFFERY LEARY, ESQ.
18495 S. Dixie Hwy,
PMB 107
Miami, FL 33157

ENVIRONMENTAL CONSULTANTS:
OCEAN
CONSULTING, LLC
340 Minnesota Avenue, Suite 7
Coral Gables, Florida 33134
Tel: (305) 921-9244
Fax: (305) 677-3254

CONTRACTOR:

PROJECT ENGINEER:
MENDEZ PROFESSIONAL
ENGINEERING CORP.
1385 Coral Way, Ste 203
Miami, FL 33145
Phone: (305) 450-8238

SEAL / SIGNATURE / DATE

José A. Méndez
P.E. FL 53288, CA 08324

PERMIT DRAWING

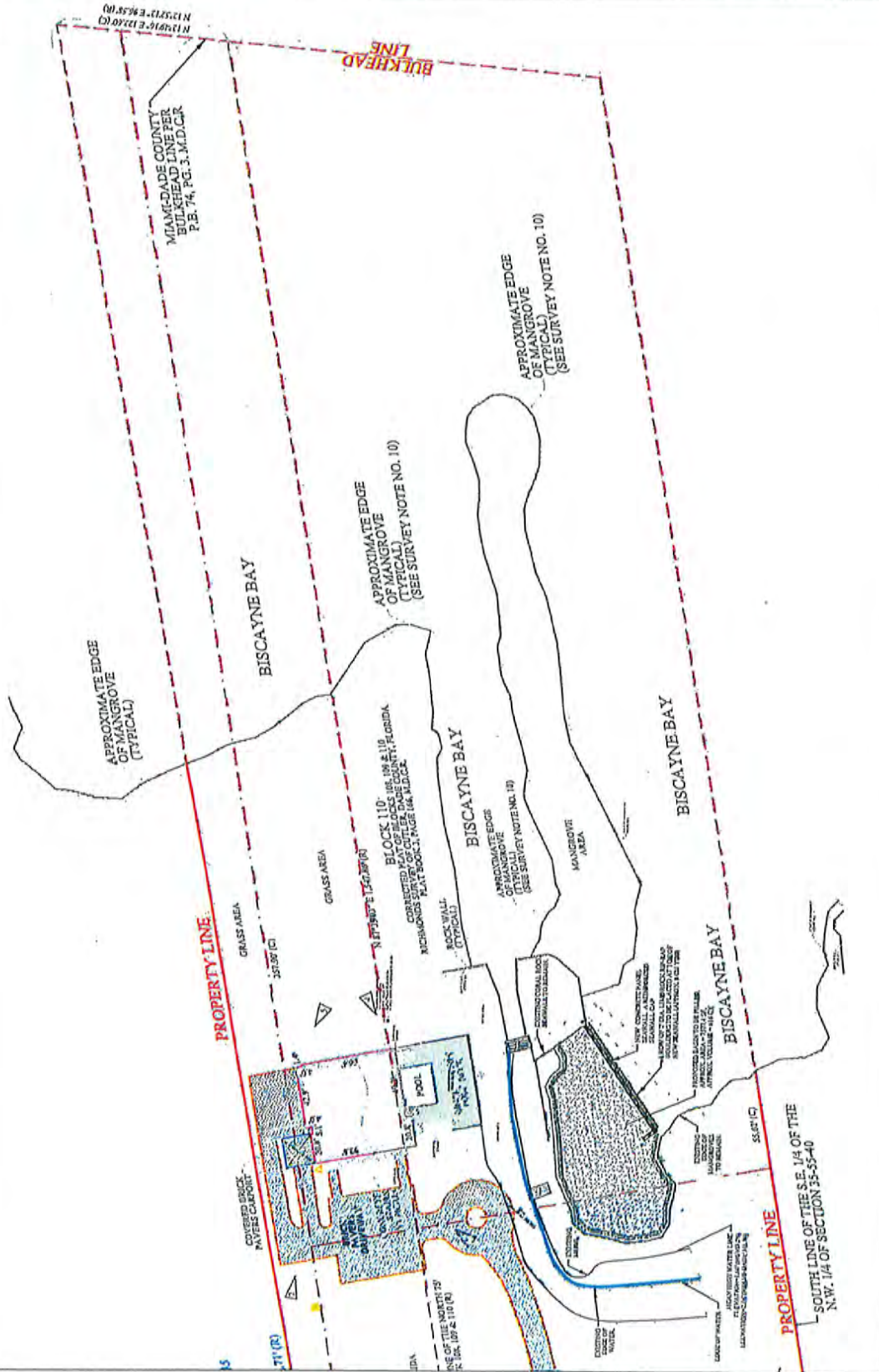
Issue #	Issue Date
①	January 30, 2017
②	May 14, 2019
③	January 13, 2020

PROJECT: 15-4915

PROPOSED SITE
CONDITIONS

SCALE: AS SHOWN
SHEET NO.

3-5



GRAPHIC SCALE
0 100
feet

17575 OLD CUTLER ROAD
SHORELINE
RESTORATION PROJECT

CLIENT:
MR. JEFFERY LEARY, ESQ.
18498 S. Dixie Hwy.
PMB 107
Miami, FL 33157

ENVIRONMENTAL CONSULTANT:
OCEAN
CONSULTING, LLC
340 Miranda Avenue, Suite 7
Coral Gables, Florida 33134
Tel: (305) 921-9144
Fax: (305) 677-3254

CONTRACTOR:

PROJECT ENGINEER:
MENDEZ PROFESSIONAL
ENGINEERING CORP
1383 Coral Way, Ste 203
Miami, FL 33145
Phone: (305) 450-4238

SEAL / SIGNATURE / DATE

Jose A. Mendez
P.E. FL 53283, CA 08324

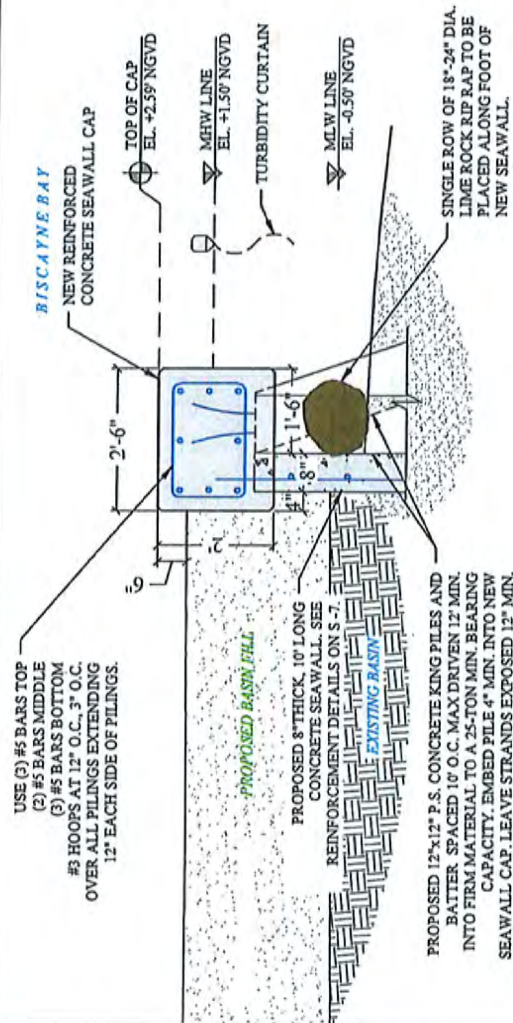
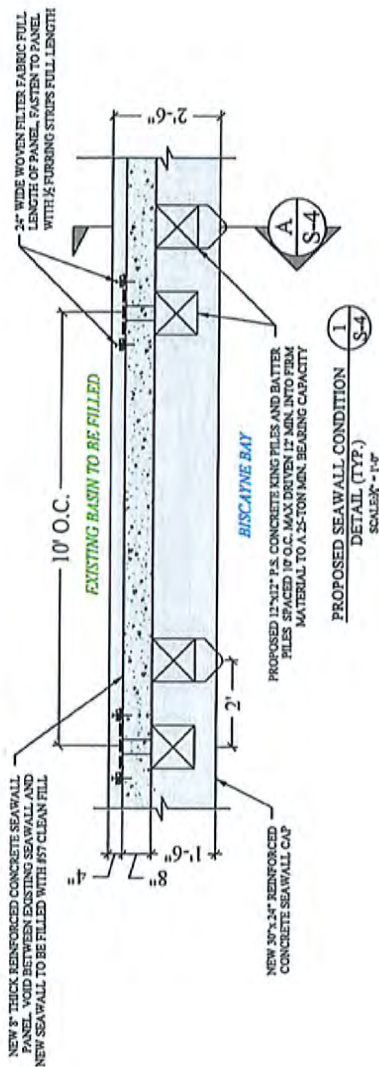
PERMIT DRAWING

Issue # Issue Date
① January 30, 2017
② May 14, 2019
③ January 13, 2020

PROJECT: 15-4915

PROPOSED SEAWALL &
CONCRETE CAP
SCALE: AS SHOWN
SHEET NO.

S-4



TURBIDITY CURTAIN NOTES.

1. A TURBIDITY CURTAIN WILL BE USED AS APPROPRIATE TO MAINTAIN AMBIENT WATER QUALITY DURING ALL CONSTRUCTION ACTIVITIES.
2. THE TURBIDITY CURTAIN WILL BE WEIGHTED AND EXTEND TO SUBMERGED WATERWAY BOTTOM (NOT TO REST ON BOTTOM).
3. THE TURBIDITY CURTAIN SHALL SURROUND ACTIVITY CONSTRUCTION AREAS ONLY.
4. THE TURBIDITY CURTAIN WILL BE OUTSIDE OF ANY NAVIGABLE CHANNEL.

17575 OLD CUTLER ROAD
SHORELINE
RESTORATION PROJECT

CLIENT:
MR. JEFFERY LEARY, ESQ.
18405 S. Dixie Hwy.
PMB 107
Miami, FL 33157

ENVIRONMENTAL CONSULTANT:
OCEAN
CONSULTING, LLC
3401 Miranda Avenue, Suite 7
Coral Gables, Florida 33134
Tel: (305) 921-9344
Fax: (305) 675-3234
CONTRACTOR:

PROJECT ENGINEER
MENDEZ PROFESSIONAL
ENGINEERING CORP
1385 Coral Way, Ste 203
Miami, FL 33145
Phone: (305) 450-4238

SEAL / SIGNATURE / DATE

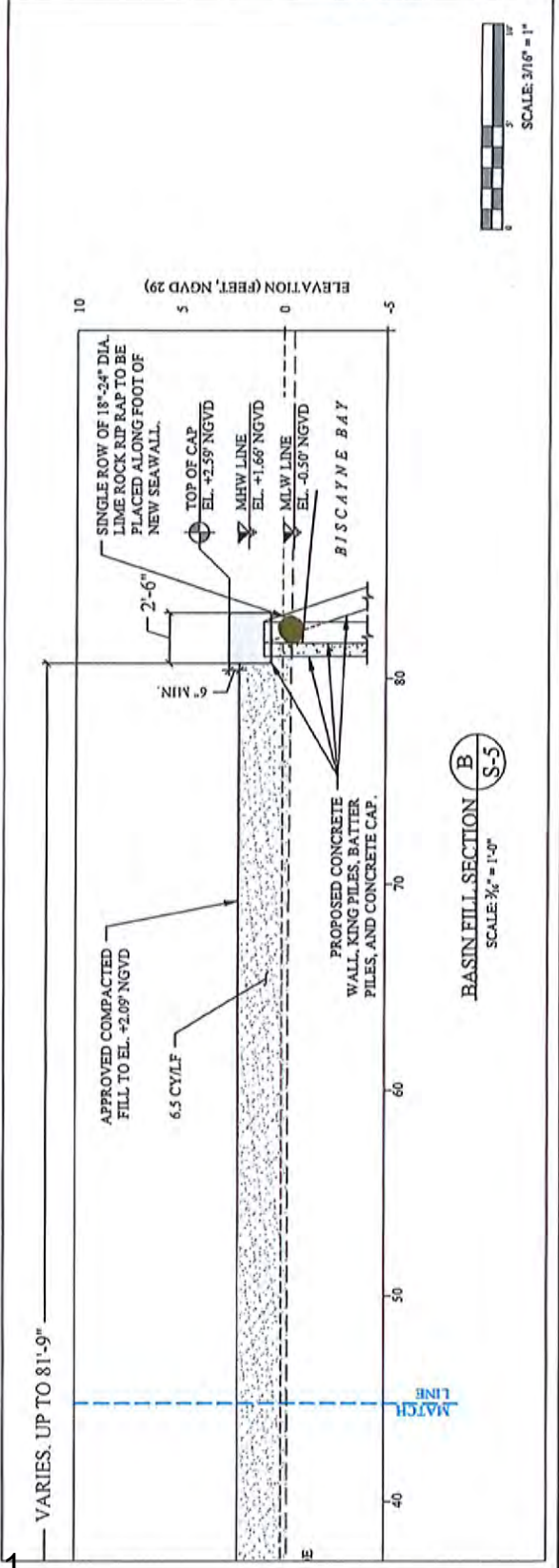
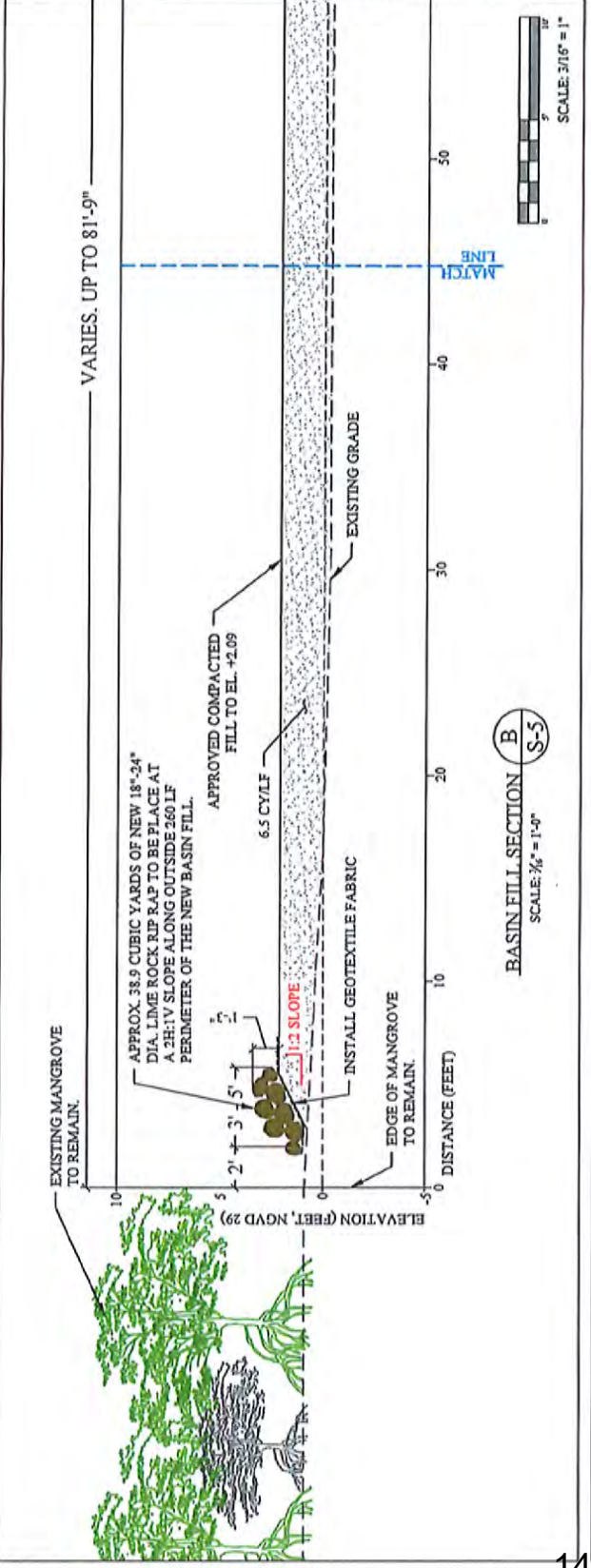
Jose A. Mendez
P.E. FL 52288, CA 08324

PERMIT DRAWING
Issue # Issue Date
① January 30, 2017
② May 14, 2019
③ January 13, 2020

PROJECT: 15-4915

PROPOSED BASIN FILL
SECTION
SCALE: AS SHOWN
SHEET NO.

S-5



17575 OLD CUTLER ROAD
SHORELINE
RESTORATION PROJECT

Miami, Florida
 CLIENT: **MR. JEFFERY LEARY, ESQ.**
 18493 S. Dixie Hwy.
 PMB 107
 Miami, FL 33157

ENVIRONMENTAL CONSULTANT:
OCEAN
CONSULTING, LLC
340 Minorca Avenue, Suite 7
Coral Gables, Florida 33134
Tel: (305) 921-9344
Fax: (305) 677-3254

CONTRACTOR:

**PROJECT ENGINEER-
MENDEZ PROFESSIONAL
ENGINEERING CORP.**
1385 Coral Way, Ste 203
Miami, FL 33145
Phone: (305) 450-8238

SEAL / SIGNATURE / DATE

Jose A. Mendez
P.E. FL 53288, CA 08324

PERMIT DRAWING

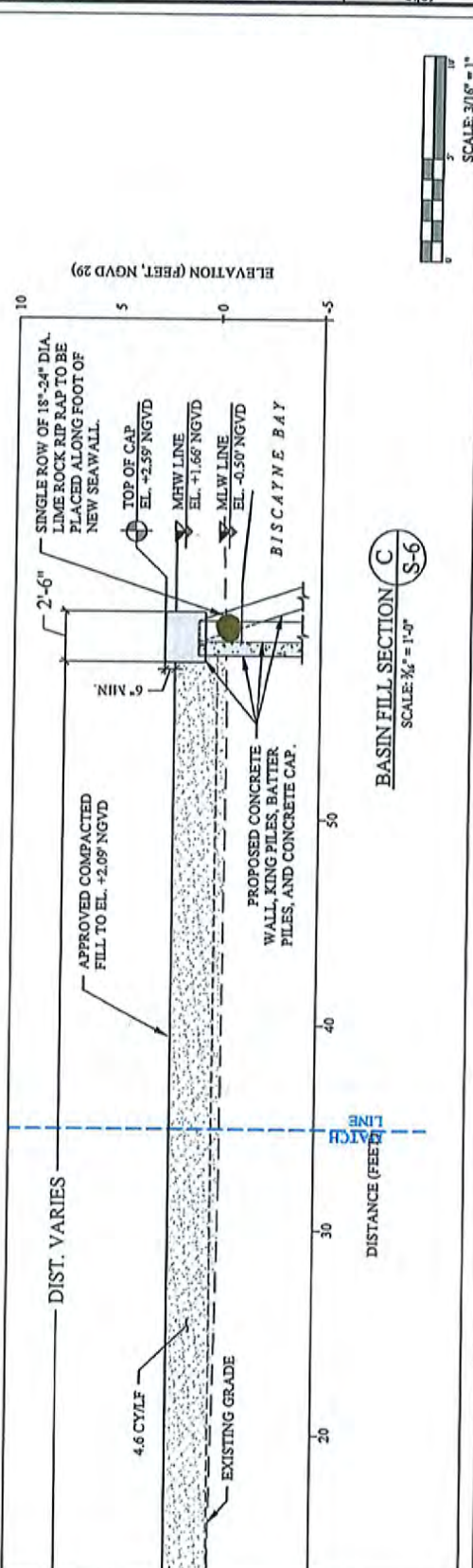
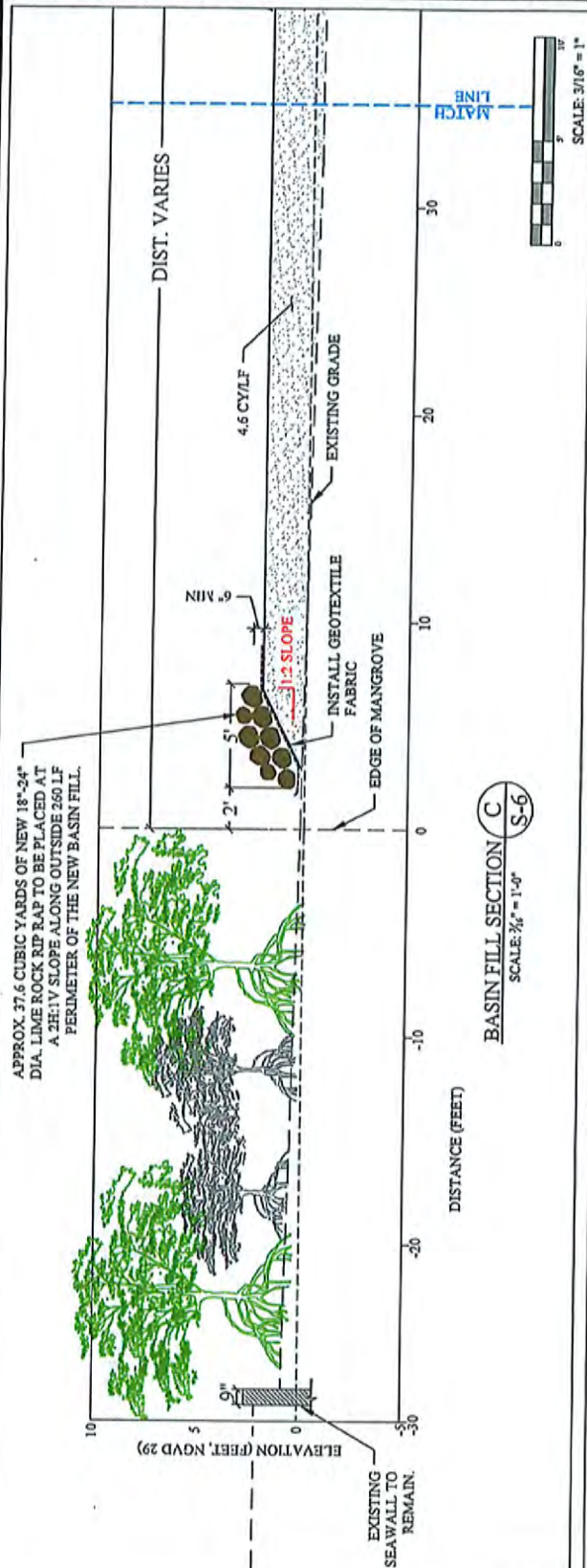
Issue #	Issue Date
①	January 30, 2017
②	May 14, 2019
③	January 13, 2020

PROJECT: 15-4915

PROPOSED BASIN FILL
SECTION

SCALE: AS SHOWN
SHEET NO.

5-6



**17575 OLD CUTLER ROAD
SHORELINE
RESTORATION PROJECT**

CLIENT:
Miami, Florida
MR. JEFFERY LEARY, ESQ.
18495 S. Dixie Hwy.
PMB 107
Miami, FL 33157

ENVIRONMENTAL CONSULTANT:
**OCEAN
CONSULTING, LLC**
3401 Miranda Avenue, Suite 7
Coral Gables, Florida 33134
Tel: (305) 921-9344
Fax: (305) 677-5254
CONTRACTOR:

PROJECT ENGINEER:
**MENDEZ PROFESSIONAL
ENGINEERING CORP**
1385 Coral Way, Ste 203
Miami, FL 33145
Phone: (305) 450-4238

SEAL / SIGNATURE / DATE

Jose A. Mendez
P.E. FL 57388, CA 08324

PERMIT DRAWING

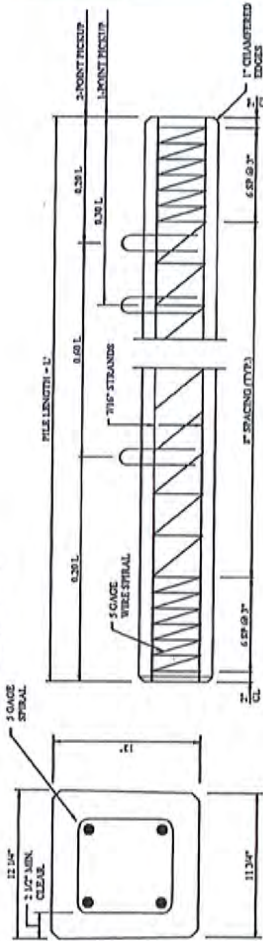
Issue # Issue Date
① January 30, 2017
② May 14, 2019
③ January 13, 2020

PROJECT: 15-4915
BASIN FILL SECTION
CONC. PILE & SEAWALL
PANEL DETAIL
SCALE: AS SHOWN
SHEET NO.

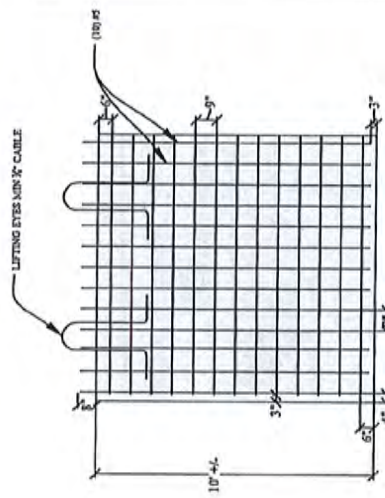
S-7

FILE NOTE:

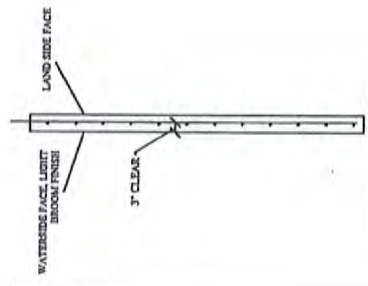
CONCRETE COMPLETIVE STRENGTH:
3000 PSI @ RELEASE
5000 PSI @ CURING
SPRALL TO BE ASTM A-42 TIGHT
HARD-DRAWN WIRE
STRANDS - USE EBAR 216 (GREEN)
LOW RELAXATION ASTM A-421
5-WIRE STRANDS TENSIONED TO 137 KIPS
EACH
1-POINT PICKUP MAXIMUM L - 40'
2-POINT PICKUP MAXIMUM L - 42'



CONCRETE PILE DETAIL

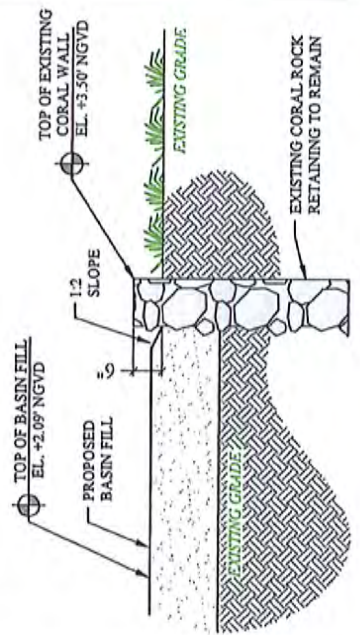


**PANEL DETAIL
SCALE: AS SHOWN**

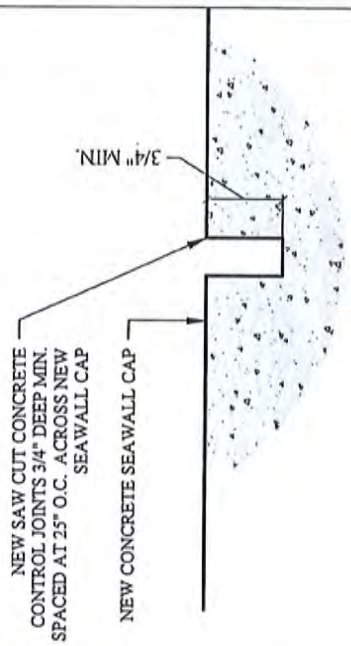


**TYP. SECTION OF
CONCRETE PANEL
SCALE: AS SHOWN**

STEEL DETAIL



**BASIN FILL CONNECTION SECTION
SCALE: 1/2\"/>**



**CONTROL JOINT DETAIL
N.T.S.**

Attachment J
Zoning Memorandum

Memorandum



Date: October 12, 2021

To: McKee Gray, Manager *McKee Gray*
Coastal Resources Section
Department of Regulatory and Economic Resources

From: Rockell Alhale, ERPS *R. Alhale*
Coastal Resources Section
Department of Regulatory and Economic Resources

Subject: Class I Permit Application by John Dubois for the Filling of Tidal Waters in Association with the Installation of a New Seawall in Biscayne Bay at 17575 Old Cutler Road, Palmetto Bay, Miami-Dade County, Florida

Pursuant to Section 24-48.2(II)(B)(7), of the Code of Miami-Dade County, Florida, a substantiating letter or plan approval shall be submitted by the applicant stating that the proposed project does not violate any zoning laws. Said letter or plan approval will be submitted after approval by the Miami-Dade County Board of County Commissioners and prior to issuance of the Class I permit.

Attachment K

Names and Addresses of Owners of All Riparian or Wetland Property within Three Hundred (300) Feet of the Proposed Work

Attachment K:

**Names and Addresses of Owners of All Riparian or Wetland Property within Three
Hundred (300) Feet of the Proposed Work**

Yardena Taitz Trust
17401 Old Cutler Road
Miami, Florida 33157

Bette Burch
17601 Old Cutler Road
Miami, Florida 33157

United States of America
Biscayne National Park
2975 Horseshoe Drive South
Suite 800
Naples, Florida 34104

Village of Palmetto Bay
9705 East Hibiscus Street
Palmetto Bay, Florida 33157

CLASS I PERMIT APPLICATION NO. CLI-2018-0467

Class I Permit Application by John Dubois for the Filling of Tidal Waters in Association with the Installation of a New Seawall in Biscayne Bay at 17575 Old Cutler Road, Palmetto Bay, Miami-Dade County, Florida

DATE: October 12, 2021

Staff's recommendation of denial for the above-referenced permit application is based on the applicable evaluation factors under Section 24-48.3 of the Code of Miami-Dade County, Florida (Code). The following is a summary of the proposed project and staff's opinion to its conformance with each with respect to each applicable evaluation factor:

1. **Potential Adverse Environmental Impact** – The subject 8-acre property is the site of a single-family residence which fronts Biscayne Bay in the Village of Palmetto Bay. The property abuts the Biscayne Bay Aquatic Preserve and is situated less than one hundred yards north of the boundary of Biscayne National Park. The shoreline of the subject property is primarily unconsolidated and stabilized by halophytic (i.e. salt tolerant) plant species, predominantly mangrove trees, consistent with the neighboring properties. The project area is not unlike other unconsolidated shorelines or tidal flat habitats throughout Biscayne Bay. A small portion of the shoreline located south of the single-family residence is stabilized by an existing concrete and limerock seawall (see Aerial of Subject Property, Attachment A). The proposed project area is shown on Attachment B.

On January 31, 2019, DERM staff biologists conducted a biological assessment (Attachment C) of the proposed project area. The project area is shallow with portions of the submerged lands exposed at low tide, which is typical of intertidal areas associated with mangrove fringes. The shoreline landward and to the south of the proposed fill footprint is predominantly vegetated with red mangroves (*Rhizophora mangle*), white mangroves (*Laguncularia racemosa*), and black mangroves (*Avicennia germinans*). Within the submerged portions of the proposed project areas, sparse shoal grass (*Halodule wrightii*) and macroalgae consisting of brown algae (*Sargassum spp.*) and green algae (*Batophora spp.*) were observed. The project area also contained organic detritus, which is decomposing organic matter produced by the natural breakdown of such things as marine vegetation.

The seagrass and macroalgae documented during the biological assessment represent important components of benthic habitat that are typical of the tidal system. Shallow tidal areas such as these are used by various wildlife including avian species. During the biological assessment at the subject property, DERM staff observed several avian species foraging in the project area including a green heron, a juvenile little blue heron (a State designated threatened species), and white ibis.

Mangroves and their root systems, including those at the subject property, are integral to shoreline stability in coastal areas and are an important component of the Biscayne Bay ecosystem. Mangroves and their root systems help minimize erosion through the stabilization of sediment, provide habitat for marine and avian fauna, increase water quality and clarity by filtering nutrients and other pollutants, and provide organic detritus to the marine trophic system as recognized in scientific literature (see three scientific articles related to mangroves, Attachment D). In addition, shorelines vegetated with mangroves, similar to those at the subject property and throughout Biscayne Bay, act as a form of natural protection to storm surge during damaging storm events, bearing the initial brunt against powerful winds and waves, as further supported by scientific literature (see scientific article related to storm surge and mangroves, Attachment E).

The organic detritus documented during the biological assessment is consistent with the natural accumulation of detritus within mangrove fringes and shallow tidal areas. The decomposition of organic detritus provides essential nutrients that form the foundation of the food web. These nutrients play a vital role in not only the ecology of the mangrove system itself but also adjacent seagrass and coral reef communities and the overall health of Biscayne Bay (see scientific articles related to mangrove detritus, Attachment D). In addition, as the organic detritus breaks down, the resulting deposits settle to the substrate and create new habitat. This new habitat can support the growth of submerged aquatic vegetation such as seagrasses or emergent vegetation in the form of mangroves.

The proposed scope of work for this Application involves the filling of approximately 10,500 square feet of tidal waters within Biscayne Bay to an elevation of +2.09 feet National Geodetic Vertical Datum (NGVD). The Applicant proposes to contain the fill by installing a concrete panel seawall along the east side of the project area (project area shown on Attachment B) and a three-foot-wide limerock riprap boulder revetment along the north, west, and south sides of the fill area, which would be situated between the proposed fill area and the existing

mangrove fringe. The proposed seawall is proposed to have concrete king piles, concrete batter piles, and a row of limerock riprap boulders to be installed waterward of the new seawall. The northern portion of the new seawall is intended to connect to an existing seawall and the southern extent of the new seawall would terminate at the mangrove fringe. The Applicant has stated that the purpose of the request to fill tidal waters is to restore a segment of the property's shoreline to what the Applicant asserts existed in 1938. The proposed project would result in filling approximately 0.25 acres of submerged bay bottom lands of Biscayne Bay. The Applicant further asserts that the proposed project will protect the property from hurricanes, eliminate stagnant water, and enhance the adjacent waters.

DERM staff completed an evaluation of the proposed work including a review of historic aerial photographs of the project location and previous Class I permit application records, along with the Applicant's submittals. In addition, DERM considered both historical and existing site conditions when evaluating the project. Although the Applicant asserts that the purpose of the project is to restore the shoreline, a review of historic aerials (see historical aerials, Attachment F) revealed that the shoreline at the project location has been stabilized by the mangrove fringe and associated root systems in approximately the same configuration for over 50 years. Accordingly, the Applicant's assertion regarding the configuration of the shoreline in 1938 is irrelevant and insufficient to overcome the Code criteria for the filling of tidal waters. Moreover, DERM staff notes that the 1938 aerial photograph that the Applicant relies on is inconclusive as to the location of the shoreline at that time. In addition, property records show that the Applicant purchased the property in the year 2000 with the shoreline in the current configuration.

Section 24-48.3(1) of the Code provides that DERM shall base its recommendation on this Application, and the Board shall make its decision, based on certain evaluation factors, when applicable. These evaluation factors are listed in Section 24-48.3(1) of the Code and include but are not limited to, the potential adverse environmental impact and cumulative adverse environmental impacts of the proposed project including but not limited to the effect upon hydrology, water quality, water supply, wellfields, aquifer recharge, aesthetics, navigation, public health, historic values, air quality, marine and wildlife habitats, archeological values, wetlands soils suitable for habitat, floral and faunal values, rare, threatened, and endangered species, natural flood damage protection, wetland values, land use classification, recreation, and any other environmental values affecting the public interest.

Based on a review of these evaluation factors, this Application should be denied, as explained in more detail herein. As proposed, adverse environmental impacts would likely occur as a result of the filling of the project area and construction of a new seawall and riprap revetment at the subject property. The proposed filling would alter a functioning intertidal system by eliminating the existence of seagrass and wetland habitat, thereby diminishing the wetland value of this area. It is likely that the proposed filling would also result in depleted floral and faunal values for various species by eliminating a high functioning intertidal system. The installation of the proposed seawall and the filling of the project area is not likely to provide additional protection during storm events or provide flood protection of the property as the elevation of the seawall cap would likely not be sufficient to prevent the continual erosion of fill as a result of tidal events (see excerpts of the Public Works Manual, Attachment G).

Although the construction of the project does not include the trimming or removal of mangroves, the project is expected, over time, to negatively impact the existing mangroves onsite for the following reasons. The project is likely to result in minimized tidal exchange to the mangroves growing along this section of the shoreline. This reduced tidal flushing is reasonably expected to have negative impacts to the mangroves and mangrove ecosystem. In addition, the filling of the project area, including the placement of riprap immediately adjacent to the mangroves, is reasonably expected to accelerate sedimentation within the mangrove fringe landward of the fill footprint and would likely result in the degradation of the existing mangrove habitat by minimizing the area for tidal exchange to the mangroves growing along this section of the shoreline.

In addition to the negative environmental impacts discussed above, there is also the potential for several secondary environmental impacts due to the proposed project design and construction methodology. The project as designed does not adequately address the containment or grading of the proposed fill. Based on the proposed seawall cap elevation, it is reasonably expected that tidal waters will wash over the seawall during high tides and storm events. As a result of these tidal events, it is reasonably expected that the proposed seawall configuration will result in the continual erosion of fill into tidal waters during an outgoing tide, or during extreme marine weather, thus negatively affecting water quality and disrupting the hydrology of the project area and possibly beyond. Long term erosion may affect the benthic resources documented waterward of the proposed seawall and the mangrove wetlands that border the proposed riprap revetment. Furthermore, the construction methodology submitted raises several concerns including the feasibility of conducting the proposed work by water, how the construction operations will avoid and minimize potential direct and secondary impacts to benthic resources, and whether adequate water depth exists for the barge and any associated vessels to navigate through the area. Although if approved, the Class I permit would require standard turbidity controls during all phases of construction, the above mentioned concerns relate to the long term and continued adverse environmental impacts associated with the project, beyond the construction phase of the project.

In addition, Section 24-48.3(2) of the Code provides that dredging and filling work proposed in Class I permit applications shall comply with at least one of the criteria listed below:

- a) Minimum dredging and spoiling for public navigation or public necessity.
- b) An alteration of physical conditions as may be necessary to enhance the quality or utility of adjacent waters
- c) Minimum dredging and filling for the creation and maintenance of marinas, piers, docks and attendant navigational channels.
- d) Minimum dredging and filling as is necessary for the elimination of conditions hazardous to the public health or for the elimination of stagnant waters.
- e) Minimum dredging and filling as is necessary to enhance the biological, chemical or physical characteristics of adjacent waters.
- f) A physical modification necessary to protect public or private property.

For the reasons discussed herein, this Application is not consistent with any of the dredge and fill criteria listed in Section 24-48.3(2)(a)-(f) of the Code as referenced above, and therefore this Application should be denied. Dredge and fill criteria (a) does not apply as the project does not involve work relating to public navigation or public necessity. The project is not consistent with dredge and fill criteria (b) because the filling and alteration of the shoreline would not enhance the quality or utility of adjacent waters. The proposed work would likely result in the discharge of fill material into the adjacent mangrove and benthic habitats, thereby decreasing overall water quality. Dredge and fill criteria (c) does not apply as the project does not involve the creation and maintenance of marinas, piers, docks and attendant navigational channels. The project is not consistent with dredge and fill criteria (d) because the proposed filling is not necessary for the elimination of conditions hazardous to the public health or for the elimination of stagnant waters. In fact, the project area does not contain hazardous conditions or stagnant waters and is comprised of organic detritus that is consistent with the natural accumulation of detritus within mangrove fringes and shallow tidal areas. The nutrients produced from the decomposition of organic detritus provides essential nutrients that form the foundation of the food web and play a vital role in the ecology of the mangrove system, the adjacent seagrass and coral reef communities, and the overall health of Biscayne Bay. The project is not consistent with dredge and fill criteria (e) because the project will have a negative effect on the biological, chemical, and physical characteristics of the adjacent waters as the proposed fill material will eliminate the benthic habitat within the fill footprint and it is likely that the fill material will discharge into the adjacent mangrove and benthic habitats, decreasing water quality, habitat, and biodiversity. The project is not consistent with dredge and fill criteria (f) because the filling is not necessary to protect private property. As previously discussed in this memorandum, the shoreline of the property has been stable for over 50 years. In addition, based on the elevation of the proposed seawall it is reasonably expected that tidal waters will wash over the seawall during high tides or extreme marine weather events. The existing mangroves at the property serve as a form of natural protection to storm surge during damaging storm events and the proposed work is likely to have a negative effect on the mangroves as detailed above. Based on DERM's evaluation, there are less impacting

options for the Applicant to protect the property, including the installation of a retaining wall landward of the existing mangrove fringe. DERM concludes that the project as proposed is not consistent with any of the dredge and fill criteria.

Pursuant to Section 24-48.4 of the Code, potential adverse environmental impacts and cumulative adverse environmental impacts for a proposed project must be avoided and minimized. Mitigation is required to compensate for the loss of habitat and for water quality impacts associated with the filling of tidal waters and is calculated based on the functional loss of the habitat. In addition, mitigation is required to compensate for water quality impacts associated with the installation of a vertical seawall. Section 24-48.4 of the Code further states that mitigation should not be used to make an otherwise non-permittable project permittable, and must maximize preservation of existing natural resources including avoiding the impact altogether by not taking certain action or parts of an action, as well as minimizing impacts by limiting the degree or magnitude of the action or its implementation. As proposed, the project has not been designed to avoid and minimize impacts and will result in adverse environmental impacts and cumulative adverse impacts to the existing benthic and wetland habitat. To mitigate for the proposed project, the Applicant proposes to use credits purchased from the Florida Power & Light Everglades Mitigation Bank (EMB). Should the Board approve the Application as proposed, the amount of compensatory mitigation required would be determined based on the W.A.T.E.R. - Wetland Assessment Technique for Environmental Reviews and would therefore require the purchase of 0.17 EMB saltwater credits.

2. **Potential Cumulative Adverse Environmental Impact** – The proposed project is reasonably expected to result in cumulative adverse environmental impacts as set forth in Number 1 above.
3. **Hydrology** – The project is reasonably expected to adversely affect surface water drainage as the fill will disrupt the natural flushing of the mangrove wetlands as set forth in Number 1 above. Furthermore, based on the proposed final seawall cap elevation of 2.59 feet NGVD, it is reasonably expected that tidal waters will flow over the seawall cap and inundate the fill area during seasonal high tides and storm events and result in the long term erosion of fill into tidal waters and the adjacent mangrove wetlands.
4. **Water Quality** – The proposed project is reasonably expected to adversely affect water quality. The filling of the project area is expected to accelerate sedimentation within the mangrove fringe landward of the proposed fill footprint and would likely result in the degradation of the water quality of the existing mangrove habitat by minimizing the area for tidal exchange to said mangroves. In addition, based on the proposed elevation of the seawall cap (2.59 feet NGVD), it is reasonably expected that tidal waters will go over the seawall cap during seasonal high tides and storm events and result in the long term erosion of sediment into tidal waters, thereby degrading the water quality of adjacent benthic and wetland habitats.
5. **Wellfields** – The proposed project is not reasonably expected to adversely affect wellfields.
6. **Water Supply** – The proposed project is not reasonably expected to adversely affect water supply.
7. **Aquifer Recharge** – The proposed project is not reasonably expected to adversely affect aquifer recharge.
8. **Aesthetics** – The proposed project is reasonably expected to result in negative aesthetic impacts; the shorelines of the adjacent properties to the north and south are unconsolidated and comprised of mangrove fringes. Therefore, the installation of a seawall would not be consistent with the surrounding area.
9. **Navigation** – The proposed project is not reasonably expected to adversely affect navigation.
10. **Public Health** - The proposed project is not reasonably expected to adversely affect public health.
11. **Historic Values** - The proposed project is not reasonably expected to adversely affect historic values.
12. **Archaeological Values** - The proposed project is not reasonably expected to adversely affect archaeological values.

13. **Air Quality** – The proposed project is not reasonably expected to adversely affect air quality.
14. **Marine and Wildlife Habitats** – The proposed project is reasonably expected to adversely affect marine and wildlife habitats, as set forth in Number 1 above.
15. **Wetland Soils Suitable for Habitat** – The proposed project involves work in wetland soils. The proposed project will disrupt the natural surface water flow to the intertidal area located adjacent to the mangrove shoreline, which contain wetland soils suitable for the establishment of wetland vegetation. Additionally, the project as proposed is reasonably expected to adversely affect the mangrove wetland habitat through the long term degradation of the surrounding areas as a result of sedimentation from erosion of the proposed fill.
16. **Floral Values** – The proposed project is reasonably expected to adversely affect floral values as set forth in Number 1 above.
17. **Fauna Values** – The proposed project is reasonably expected to adversely affect fauna values as set forth in Number 1 above.
18. **Rare, Threatened and Endangered Species** – Please see Objective 9/Policy 9B of the Conservation, Aquifer Recharges, and Drainage Element IV as discussed in this project report.
19. **Natural Flood Damage Protection** - The proposed project is reasonably expected to adversely affect surface water drainage or retention of stormwater. The project is reasonably expected to adversely affect surface water drainage based on the proposed final elevation of the seawall cap to 2.59 feet NGVD. Based on this elevation, it is reasonably expected that tidal waters will go over the seawall cap during seasonal high tides and storm events and result in the long term erosion of sediment into tidal waters and the adjacent mangrove wetlands as set forth in Number 1 above.
20. **Wetland Values** – The proposed project is reasonably expected to adversely affect wetland values as set forth in Number 1 above.
21. **Land Use Classification** – Pursuant to Section 24-48.2(II)(B)(7) of the Code of Miami-Dade County, Florida, a substantiating letter or plan approval shall be submitted stating that the proposed project does not violate any zoning laws. Said letter or plan approval will be submitted after the approval by the Board of County Commissioners and prior to the issuance of a Class I permit.
22. **Recreation** - Not applicable.
23. **Other Environmental Values Affecting the Public Interest** – The proposed project is not reasonably expected to adversely affect other environmental values affecting the public interest. The proposed project will occur on lands owned by the applicant.
24. **Conformance with Standard Construction Procedures and Practices and Design and Performance Standards** – The project does not comply with the standard construction procedures and practices and design and performance standards of the applicable portions of the Miami-Dade County Public Works Manual, including but not limited to Sections C3 and D5. Chapter 33B of the Code of Miami-Dade County is not applicable.
25. **Comprehensive Environmental Impact Statement (CEIS)** - In the opinion of the Director, the proposed project is reasonably expected to result in significant adverse environmental impacts and cumulative adverse environmental impacts. However, a CEIS was not necessary for DERM to complete an evaluation of the proposed project.

26. **Conformance with All Applicable Federal, State and Local Laws and Regulations** - The Applicant has obtained authorization from the Florida Department of Environmental Protection; however, the proposed project is not in conformance with all applicable regulations of the Code of Miami-Dade County (as discussed in the memo) (Chapter 24 of the Code of Miami-Dade County) and the Applicant will be advised to obtain Federal authorization pursuant to the United States Clean Water Act.
27. **Conformance with the Miami-Dade County Comprehensive Development Master Plan (CDMP)** - In the opinion of the Director, the proposed project is not in conformance with all relevant Objectives in the CDMP and therefore is not consistent with the goals and objectives of Policy 3A of the Land Use Element. The following is a summary of the proposed project as it relates to the CDMP:

LAND USE ELEMENT I:

Objective 3/Policy 3B - Protection of significant natural resources and systems from incompatible land uses. The Applicant has not demonstrated that the proposed project is a compatible land use.

Objective 3/Policy 3C - Protection of natural resources and systems. The proposed project does not involve development in the Big Cypress area of Critical State Concern or the East Everglades.

TRANSPORTATION ELEMENT II

Aviation Subelement/Objective AV-5A - Aviation System Expansion - There is no aviation element to the proposed project.

Port of Miami River Subelement/Objective 3 - Minimization of impacts to estuarine water quality and marine resources. The project is not located within the Miami River.

CONSERVATION, AQUIFER RECHARGES AND DRAINAGE ELEMENT IV:

Objective 3/Policies 3A, 3B, 3D - Wellfield protection area protection. - The proposed project is not located within a wellfield protection area.

Objective 3/Policy 3E - Limestone mining within the area bounded by the Florida Turnpike, the Miami-Dade/Broward Levee, N.W. 12 Street and Okeechobee Road. - The proposed project is not located within this area.

Objective 4/Policies 4A, 4B, 4C - Water storage, aquifer recharge potential and maintenance of natural surface water drainage. - The proposed project is not reasonably expected to adversely affect water storage, aquifer recharge potential or natural surface water drainage as the project area is not located within the areas identified by this objective.

Objective 5/Policies 5A, 5B, 5F - Flood protection and cut and fill criteria – The proposed project is not located within a cut and fill basin.

Objective 6/Policy 6A - Areas of highest suitability for mineral extraction. - The proposed project is not located in an area proposed or suitable for mineral extraction.

Objective 6/Policy 6B - Guidelines for rock quarries for the re-establishment of native flora and fauna. - The proposed project is not located in a rock quarry.

Objective 7/Policy 7A - Wetland protection and restoration. The proposed project is reasonably expected to adversely affect the wetlands adjacent to the project footprint and restrict the water flow into and through the existing mangroves on-site. The Applicant has not demonstrated that the project is consistent with any of the activities listed under Policy 7A: the project is not necessary to prevent or eliminate a threat to public health,

safety or welfare, there are other reasonable alternatives that exist, and the project area is not characterized as a disturbed or degraded area.

Objective 7/Policy 7J – Consistency with Comprehensive Everglades Restoration Program (CERP) objectives- The proposed project is not inconsistent with CERP.

Objective 9/Policy 9A- Protection of habitat critical to Federal or State-designated threatened or endangered species. The project is not reasonably expected to adversely affect habitat that is critical to federal or State designated, endangered or threatened species.

Objective 9/Policy 9B- Protection of habitat critical to Federal or State-designated threatened or endangered species. The Applicant has not demonstrated that feeding habitats used by federal or State designated threatened or endangered species will be protected and buffered from surrounding development and will not be further degraded or destroyed. Please note as referenced Number 1 above, DERM staff observed a juvenile little blue heron (a State designated threatened species) foraging within the project area.

Objective 9/Policy 9C- Protection of habitat critical to Federal or State-designated threatened or endangered species. DERM staff have not observed rookeries or nesting sites within the project area.

COASTAL MANAGEMENT ELEMENT VII:

Objective 1/Policy 1A – Mangrove wetlands within Mangrove Protection Areas – The proposed project is not located within a designated “Mangrove Protection Area.”

Objective 1/ Policy 1B - Natural surface flow into and through coastal wetlands. – The proposed project will affect natural surface flow into and through the existing coastal wetland system as set forth in Number 1 above. The existing tidal connection between the mangroves located landward of the project area and Bay is reasonably expected to be reduced and not maintained to the maximum extent possible.

Objective 1/ Policy 1C - Elevated boardwalk access through mangroves. – The proposed project does not involve the construction of an elevated walkway through mangroves.

Objective 1/Policy 1D - Protection and maintenance of mangrove forests and related natural vegetational communities. – Although the construction of the project does not include the trimming or removal of mangroves, the project is expected, over time, to adversely affect mangrove forests and related natural vegetational communities as set forth in Number 1 above. The existing tidal connection between the mangroves located landward of the project area and the Bay is reasonably expected to be reduced and not maintained to the maximum extent possible.

Objective 1/Policy 1E - Mitigation for the degradation and destruction of coastal wetlands. Monitoring and maintenance of mitigation areas. – To mitigate for the proposed impacts, the Applicant has proposed to purchase credits from the Florida Power & Light Everglades Mitigation Bank (EMB).

Objective 1/Policy 1G - Prohibition on dredging or filling of grass/algal flats, hard bottom or other viable benthic communities, except as provided for in Chapter 24 of the Code of Miami-Dade County, Florida. – The project proposes filling of an area characterized as a viable benthic community that is not otherwise consistent with Chapter 24 of the Code for the reasons set forth in Number 1 above.

Objective 2/Policies 2A, 2B - Beach restoration and renourishment objectives. - The proposed project does not involve beach restoration or renourishment.

Objective 3/Policies 3E, 3F - Location of new cut and spoil areas for proper stabilization and minimization of damages. - The proposed project does not involve the development or identification of new cut or spoil areas.

Objective 4/Policy 4A– Protection of endangered or threatened animal species. Although a State threatened species was documented onsite as identified in Number 1 above, it is not reasonably expected to imperil endangered or threatened species.

Objective 4/Policy 4C– Protection of endangered or threatened animal species. Although a State threatened species was documented onsite as identified in Number 1 above, it is not reasonably expected to imperil endangered or threatened species.

Objective 4/Policy 4E– Protection of endangered or threatened animal species. The proposed project is not within an area identified by the Miami-Dade County Manatee Protection Plan as essential habitat. However, in order to minimize any potential impacts to manatees, any Class I permit issued for any in-water work at the property would require that all standard manatee conditions be followed during all in-water operations.

Objective 4/Policy 4F– Protection of endangered or threatened animal species. The project does not involve a new or expanded marine facility.

Objective 5/Policy 5B - Existing and new areas for water-dependent uses. - The proposed project does not involve water-dependent use.

Objective 5/Policy 5D - Consistency with Chapter 33D, Miami-Dade County Code (shoreline access, environmental compatibility of shoreline development) – The thresholds for review under the Shoreline Ordinance do not apply and the proposed project is not subject to shoreline development review.

Objective 5/Policy 5F - The siting of water dependent facilities. - The proposed project does not involve the creation of new water dependent facilities.

28. **Conformance with Chapter 33B, Code of Miami-Dade County** (East Everglades Zoning Overlay Ordinance) – The proposed project is not located within the East Everglades Area.
29. **Conformance with Miami-Dade County Ordinance 81-19** (Biscayne Bay Management Plan Sections 33D-1 through 33D-4 of the Code of Miami-Dade County) – The applicant has not demonstrated consistency with the applicable portions of the Biscayne Bay Management Plan as it relates to development impacts for proposed coastal construction.
30. **Conformance with the Miami-Dade County Manatee Protection Plan** – The proposed project is not within an area identified by the Miami-Dade County Manatee Protection Plan as essential habitat. However, in order to minimize any potential impacts to manatees, any Class I permit issued for any in-water work at the property will require that all standard manatee conditions be followed during all in-water operations.
31. **Consistency with Miami-Dade County Criteria for Lake Excavation** – The proposed project does not involve lake excavation.
32. **Zoning Recommendation** – Pursuant to Section 24-48.2(II)(B)(7) of the Code of Miami-Dade County, Florida, a substantiating letter or plan approval shall be submitted stating that the proposed project does not violate any zoning laws. Said letter or plan approval would be submitted after the approval by the Board of County Commissioners and prior to the issuance of the Class I permit.
33. **Coastal Resources Management Line** - A coastal resources management line was not required for the proposed project, pursuant to Section 24-48.2(II)(B)(10)(b) of the Code of Miami-Dade County.
34. **Maximum Protection of a Wetland's Hydrological and Biological Functions** – The project as proposed is reasonably expected to adversely affect the long term hydrological and biological functions of the mangrove wetlands adjacent to the fill footprint as set forth in Number 1 above.

35. **Class I Permit Applications Proposing to Exceed the Boundaries Described in Section D-5.03(2)(a) of the Miami-Dade County Public Works Manual** – Not applicable.

The proposed project was also evaluated for compliance with the standards contained in Sections 24-48.3(2), (3), and (4) of the Code of Miami-Dade County, Florida. The following is a summary of how the standards relate to the proposed project:

24-48.3 (2) Dredging and Filling for Class I Permit –DERM evaluated the project for consistency with the dredge and fill criteria as it relates to navigation, water quality, public health, and protection of private property; the project does not comply with any of the criteria listed below:

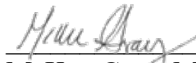
- a) Minimum dredging and spoiling for public navigation or public necessity.
- b) An alteration of physical conditions as may be necessary to enhance the quality or utility of adjacent waters.
- c) Minimum dredging and filling for the creation and maintenance of marinas, piers, docks and attendant navigational channels.
- d) Minimum dredging and filling as is necessary for the elimination of conditions hazardous to the public health or for the elimination of stagnant waters.
- e) Minimum dredging and filling as is necessary to enhance the biological, chemical or physical characteristics of adjacent waters.
- f) A physical modification necessary to protect public or private property.

The project as proposed is not consistent with any of the dredge and fill criteria listed in the Section 24-48.3(2)(a)-(f) of the Code for the reasons set forth in Number 1 above.

24-48.3 (3) Minimum Water Depth Required for Boat Slips Created by the Construction or Placement of Fixed or Floating Docks and Piers, Piles and Other Structures Requiring a Permit Under Article IV, Division 1 of Chapter 24 of the Code of Miami-Dade County – The proposed project does not involve the creation of a boat slip.

24-48.3 (4) Clean Fill in Wetlands – The proposed project involves the placement of approximately 10,500 square feet of fill in tidal waters immediately adjacent to mangrove wetlands. Any fill authorized to be placed at the property should meet the definition of clean fill as defined in Section 24-5 of the Code of Miami-Dade County.

BASED ON THE FOREGOING, IT IS RECOMMENDED THAT A CLASS I PERMIT BE DENIED.



McKee Gray, Manager
Coastal Resources Section



Rockell Alhale, ERPS
Coastal Resources Section



MEMORANDUM

(Revised)

TO: Honorable Chairman Jose "Pepe" Diaz
and Members, Board of County Commissioners

DATE: February 1, 2022

FROM: 
Gen Bonzon-Keenan
County Attorney

SUBJECT: Agenda Item No. 5(G)

Please note any items checked.

- ☐ "3-Day Rule" for committees applicable if raised
- ☐ 6 weeks required between first reading and public hearing
- ☐ 4 weeks notification to municipal officials required prior to public hearing
- ☐ Decreases revenues or increases expenditures without balancing budget
- ☐ Budget required
- ☐ Statement of fiscal impact required
- ☐ Statement of social equity required
- ☐ Ordinance creating a new board requires detailed County Mayor's report for public hearing
- ☒ No committee review
- ☐ Applicable legislation requires more than a majority vote (i.e., 2/3's present ____, 2/3 membership ____, 3/5's ____, unanimous ____, CDMP 7 vote requirement per 2-116.1(3)(h) or (4)(c) ____, CDMP 2/3 vote requirement per 2-116.1(3)(h) or (4)(c) ____, or CDMP 9 vote requirement per 2-116.1(4)(c)(2) ____ to approve
- ☐ Current information regarding funding source, index code and available balance, and available capacity (if debt is contemplated) required

Approved _____ Mayor
Veto _____
Override _____

Agenda Item No. 5(G)
2-1-22

RESOLUTION NO. _____ R-93-22

RESOLUTION TAKING ACTION ON A CLASS I PERMIT APPLICATION BY JOHN DUBOIS FOR THE FILLING OF TIDAL WATERS IN ASSOCIATION WITH THE INSTALLATION OF A NEW SEAWALL IN BISCAYNE BAY AT 17575 OLD CUTLER ROAD, PALMETTO BAY, MIAMI-DADE COUNTY, FLORIDA

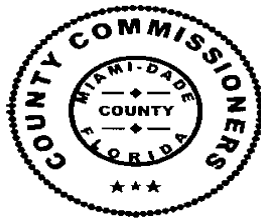
WHEREAS, this Board desires to accomplish the purposes outlined in the accompanying memorandum, a copy of which is incorporated herein by reference,

NOW, THEREFORE, BE IT RESOLVED BY THE BOARD OF COUNTY COMMISSIONERS OF MIAMI-DADE COUNTY, FLORIDA, that this Board having considered all the applicable factors contained within Section 24-48.3 of the Code of Miami-Dade County, hereby denies the application by John Dubois for the filling of tidal waters in association with the installation of a new seawall in Biscayne Bay at 17575 Old Cutler Road, Palmetto Bay, Miami-Dade County, Florida, as set forth in the memorandum from the Miami-Dade County Department of Regulatory and Economic Resources.

The foregoing resolution was offered by Commissioner **Danielle Cohen Higgins** , who moved its adoption. The motion was seconded by Commissioner **Oliver G. Gilbert, III** and upon being put to a vote, the vote was as follows:

	Jose "Pepe" Diaz, Chairman	aye	
	Oliver G. Gilbert, III, Vice-Chairman	aye	
Sen. René García	nay	Keon Hardemon	aye
Sally A. Heyman	absent	Danielle Cohen Higgins	aye
Eileen Higgins	aye	Joe A. Martinez	absent
Kionne L. McGhee	nay	Jean Monestime	aye
Raquel A. Regalado	absent	Rebeca Sosa	absent
Sen. Javier D. Souto	absent		

The Chairperson thereupon declared this resolution duly passed and adopted this 1st day of February, 2021. This resolution shall become effective upon the earlier of (1) 10 days after the date of its adoption unless vetoed by the County Mayor, and if vetoed, shall become effective only upon an override by this Board, or (2) approval by the County Mayor of this resolution and the filing of this approval with the Clerk of the Board.



MIAMI-DADE COUNTY, FLORIDA
BY ITS BOARD OF
COUNTY COMMISSIONERS

HARVEY RUVIN, CLERK

Melissa Adames

By: _____
Deputy Clerk

Approved by County Attorney as
to form and legal sufficiency.

CJW

Abbie Schwaderer-Raurell
Christopher J. Wahl