The Impact of Drought on Coastal Ecosystems in the Carolinas

State of Knowledge Report January 2012

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About CISA

The Carolinas Integrated Sciences and Assessments (CISA) works with a variety of stakeholders across North Carolina and South Carolina to incorporate climate information into water and coastal management and related decision making processes. Efforts include working with decision makers on improving drought monitoring and management, linking climate variability to watershed/landuse planning, planning for coastal adaptation, and characterizing climate vulnerability in the region. The program is supported by NOAA's Regional Integrated Sciences and Assessments (RISA) program. Based at the University of South Carolina, CISA researchers collaborate with colleagues at the Southeast Regional Climate Center located at UNC-Chapel Hill, North and South Carolina Sea Grant, North Carolina State Climate Office, and South Carolina State Climate Office.

CISA's core activities encompass five general focus areas: drought, climate and watershed modeling, coastal climate, health, and adaptation. Within each of these areas, we are pursuing several cross-cutting activities that seek to advance scientific understanding of climate and hydrological processes in the Carolinas, improve the assessment of climate-related vulnerabilities and impacts, and provide timely and relevant information and tools for decision makers.

The Impact of Drought on Coastal Ecosystems in the Carolinas

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Photographs

Front cover:

Cypress bay during 2008 drought, Brunswick County, North Carolina (Dan Tufford).

Back cover:

Salt marsh along Leadenwah Creek, Wadmalaw Island, South Carolina, at low tide (Almeda Kelley).

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I. Introduction

This State of Knowledge Report provides a synthesis and analysis of the peer-reviewed literature regarding drought impacts on coastal ecosystems in the Carolinas. Its objective is to expand current, limited understanding of drought impacts on coastal ecosystems, to identify critical gaps, to inform future research efforts, and to suggest measures to facilitate drought adaptation for ecosystems. The report centers on biological, chemical, and physical impacts and is not meant to address social impacts. While we targeted literature from studies done in the Carolinas, some research from outside the region was included when deemed pertinent. The report focuses on the coastal ecosystems of North and South Carolina but includes portions of the southern Virginia and Georgia coasts with ecological continuity to the Carolinas.

Droughts are a normal part of climate variability that occurs at scales from days to years. Significant periods of widespread drought in the study area occurred in 1925-29, 1930-35, 1950-57, 1965-1971, 1980-82, 1985-88, 1998-2002, (USGS 2002, Weaver 2005) and 2007-2008. Many of the droughts of interest in this report are multi-year droughts or "supra-seasonal" droughts, typified by longer-term rainfall deficits and declines in water availability. Of lesser concern are seasonal droughts that occur as regular periods of low flow.

This executive summary presents the key themes identified from the literature. References are cited where a specific finding or observation is noted.

Section II provides brief syntheses of research findings, organized into the following coastal subsystems: Coastal Marine; Maritime; Estuarine; Tidal Freshwater Riverine, Forested Floodplain and Marsh; Coastal Impoundment; Non-Alluvial Wetlands; and Uplands.

Section III discusses drought-related research themes highlighted by researchers. These themes include the role of anthropogenic stressors, expected hydrological and ecological changes due to climate change, and recommendations for adaptation.

Section IV presents key literature gaps and research needs.

II. Drought Impacts on Coastal Ecosystems

The research reviewed for this report indicates that drought is discussed primarily in terms of the hydrology-related impacts that affect coastal ecosystems, such as changes to river discharge, freshwater inflows, water level, and water table depth. The severity of these effects depends upon the longevity and recurrence interval of drought event(s) and may be compounded by other anthropogenic stressors on the system. In addition, some drought-related research considers how sea level impacts these ecosystems. Sea level affects the quantity of saline waters delivered to estuarine systems, and, in concert with freshwater precipitation and runoff, influences the resultant salinity levels experienced by these systems. Salinity and flushing of pore water in estuarine marshes affects geochemical processes and productivity, and thereby the distribution and survival of these marshes.

While many aquatic and wetland coastal systems are particularly vulnerable to long-term and recurrent droughts, some potentially beneficial effects of drought were found through the literature review. Such benefits include temporary changes in nutrient input into aquatic systems that may reduce potential for algal blooms and increase water clarity thereby enhancing macrophyte productivity and reducing hypoxia. However, these effects may only be short-term and their beneficial nature is strongly site dependent.

A. Coastal Marine

Very little literature was found on coastal marine systems probably because most drought impacts to the near shore system are likely to be secondary in nature (e.g., drought induced change in estuarine discharges and estuarine habitat for marine species and anadromous fish).

B. Maritime

Maritime systems occur on barrier islands and contain numerous community types with very different hydrologies; hence the effects of drought will vary significantly depending on community type. Precipitation on barrier islands appears to be the only source of freshwater input and is therefore critical to several of these communities. Consequently, extended or repeated drought will likely cause significant community changes including potential successional community changes. Freshwater ponds, the community type perhaps most sensitive to the effects of drought, often are the only dependable source of water for animals on barrier islands and are critical to their presence there (Bellis, 1995). These animals include frogs, salamanders, water snakes, turtles, aquatic birds, and aquatic mammals.

C. Estuarine

The estuarine system has by far the greatest literature coverage relative to drought and related freshwater inflow. The report discusses drought effects associated with freshwater inflow, salt marsh dieback, community shifts and productivity, flushing and water quality, and fauna. The majority of ecological effects relate to changes in salinity, productivity, water quality, and circulation.

Documented effects from reduced inflow include: upstream shifts in salinity and the presence of hypersaline conditions, an increasing tidal influence on circulation patterns that may alter stratification and affect bottom water hypoxia, the alteration of freshwater transit time with consequences for the buildup of pollutants and pathogens as well as effects on biogeochemical transformations (e.g., denitrification), changes in estuarine geomorphology due to loss of sediments, and changes to water quality due to changes in the delivery of dissolved and particulate material and in their concentrations in the estuary itself (Alber 2002). During drought periods, lower freshwater inflow can dramatically lower nutrients and planktonic primary production, especially in eutrophic estuaries like the Neuse River in North Carolina (Wetz et al. 2011). Slow or reduced flushing times have been implicated in the outbreak of harmful algal blooms (Alber and Sheldon 1999; Paerl et al. 1998), and have a potential for higher sediment chemical contamination (Hyland et al. 1999).

Literature on drought effects on salt marshes concentrates in three areas: (1) the dieback of marsh that has occurred in the Southeast (and elsewhere); (2) community shifts in macrophyte composition; and (3) decreased marsh primary productivity as a result of higher pore water salinity. While the evidence for drought being the sole cause of marsh dieback events is still unclear, the literature suggests that high salinities acting in concert with other stressors (e.g., reduced pH, high interstitial salinity, sublethal bioavailable metals, drying) over an extended period could be responsible. Research shows that salt marsh primary production decreases with an increase in pore water salinity (Morris 2000). Marsh community shifts as a result of salinity changes and freshwater inflow manipulation are also reported, although drought is not discussed as the primary causative agent. White and Alber (2009) observed shifts in marsh composition (*Spartina alterniflora* densities increasing in brackish-dominated *Spartina cynosuroides* marsh zones) during a prolonged drought, and the effects were still evident in a two-year post drought survey. It is difficult to determine if an observed change in vegetation

structure and composition is part of natural variation of the system and temporary, or if the change is longer-term.

Estuarine organisms occur within focused salinity ranges, and different stages in their life histories have specific salinity requirements (Alber 2002). Investigations document how changes in the species composition, distribution, abundance, and health of fish and invertebrate assemblages are attributable to changes in freshwater flow. Changes in river flow are also linked to changes in migration patterns, spawning habitat, and fish recruitment. Prolonged drought can have long-term effects on estuarine trophic system dynamics and the dependent fish community.

D. Tidal Freshwater

Tidal freshwater systems[1] occupy the upper ends of estuaries and are composed of tidal marshes, tidal swamps, and associated riverine conveyances (rivers, streams, and headwaters). These wetlands generally receive sufficient freshwater flows to keep surface water salinities less than 0.5[2] (Cowardin et al. 1979). Few ecosystems exist in a more vulnerable location for shifts in salinity and flood regime than tidal freshwater wetlands (Doyle et al. 2007a). Differences in community type, swamp versus marsh, were clearly related to the penetration of saline water into swamps (Hackney et al. 2007). Many existing tidal freshwater marshes in the Carolinas and Georgia are remnants of rice culture clearing of coastal swamps in the 1700s where forest cover is either actively excluded or has not yet regenerated.

With the absence of the salt stressor, tidal freshwater marshes are significantly more floristically diverse than are downstream brackish or salt marshes. These marshes provide wildlife habitat for numerous species including a diverse assemblage of wading birds, feeding habitat for freshwater fish, and nursery habitat for estuarine organisms and juvenile anadromous species. Similar to salt marshes, tidal freshwater marshes depend on building soil volume to remain in equilibrium with sea level as these systems are sensitive to the accompanying saltwater intrusion.

Acute and chronic exposure to even low levels of salinity seriously impact tidal freshwater forests. The combined stress of flooding and salinity compound this threat. Low river discharges also can play a key role in elevating salinities in these systems and may act synergistically with rising sea level, leading to rapid loss and habitat conversion of tidal freshwater marshes (Neubauer and Craft 2009). Relative sea-level rise of any degree, large or small, in the absence of offsetting accretion or salinity diluting high river flows, will impact tidal freshwater swamps by increasing submergence and salinity over time. Microtopography plays a key role in the ecology of these systems with well-drained hummocks supporting woody forest vegetation and hollows (usually with saturated soils) having bare mud or herbaceous vegetation. Conversion from forest to marsh begins at a salinity of about 2 and is likely caused by a change in soil biochemistry with a switch from methanogenisis to sulfate reduction in the presence of saline pore space water (Hackney et al. 2007). Although drought alone (especially if not chronic) may not be solely responsible for the demise of this system, the cumulative impact of a subsiding coast, rising sea level, recurring hurricanes, and reduced freshwater flow could result in forest dieback and coastal retreat (Doyle et al. 2007b). Although little research was found on drought and tidal freshwater streams and rivers, non-tidal stream literature points to extended droughts having lasting adverse effects on riverine systems (Lake 2003) including destabilization of aquatic communities.

E. Coastal Impoundments

Drought impacts on functioning coastal impoundments were not covered to any extent in the literature. However, such impoundments may be able to ameliorate some drought effects (especially those dealing with salinity) by management schemes that selectively preclude or limit exchange during high salinity events in the estuary. These actions may protect, reduce, or simply delay conversion of sensitive tidal freshwater and/or brackish systems to more saline communities. Limiting exchange to manage salinity has its own set of biological and ecological consequences and trade-offs, the nature and detail of which are beyond the scope of this report.

F. Non-Alluvial Wetlands

The vulnerability of these wetlands to drought depends, in large part, on the sources of their water supply. Those fed by precipitation are the most vulnerable. Carolina Bays and Pocosins are dependent upon direct precipitation and evaporative water loss can result in the complete drying of shallow bays. Any deviation from normal precipitation will have an impact on their hydropatterns (Sun et al. 2006). Fire plays a large role in the ecology of these systems. The high organic and peat soils burn readily when dry and so the intensity of the fire is related to the depth of the water table (and hence influenced by droughts). Severe burns could result in a shift to a non-pocosin community or even create a lake in a former vegetated area. The widespread installation of drainage ditches for land management (e.g., agriculture, pine plantations, development) has altered the hydrology of these systems. Drainage ditches lower the water table and significantly affect surface runoff (Sharitz and Gibbons 1982), potentially increasing drought vulnerability.

G. Coastal Upland Forests

Several studies in upland pine forests indicate that drought reduces root production and photosynthetic rates. Drought may also affect mast production thereby reducing forage for numerous wildlife species.

III. Related Research Themes

A. Anthropogenic Impacts

Researchers regularly discuss the cumulative effects of drought in relation to other system stressors. The biggest "natural" stressor associated with drought in estuaries is salinity. Effects of drought, most often through the salinity stressor, are compounded by anthropogenic alterations. These alterations include dredging and maintaining navigation channels, development resulting in the proliferation of non-porous surfaces that increase the flashiness of urban runoff, ditching for agricultural and timber production that increases runoff of nutrients and eroded sediments, and conversion of coastal interior wetlands that decreases retention time of nutrients and other pollutants and reduces groundwater recharge opportunities. With most river systems in the Carolinas modified by dam projects, management and operation of these dams affect the timing, duration, and volume of water flows (Richter et al. 2003). During a drought, management of large reservoirs for water supply has the potential to compound instream flow and critical estuarine freshwater inflow problems. Increasing human population densities in the coastal zone are placing ever-greater demands on the coastal aquifer and thereby

[1]Although most literature about tidal freshwater systems includes them as part of the estuary, we have broken this zone out for separate discussion in this report due to its particular sensitivity to drought and concomitant salinity changes.

[2]The Practical Salinity Scale of 1978 defines salinity in terms of a conductivity ratio, so it is unitless.

contribute to larger interaction between saline and freshwater aquifers (Dame et al. 2000).

B. Drought and Climate Change

In the Southeast, changes in annual and seasonal precipitation patterns are expected to affect the timing and amount of water available and contribute to increased risks of flooding and drought (Karl et al. 2009; Scavia et al. 2002; Committee on Environment and Natural Resources 2008). Precipitation patterns are projected to be more episodic, with less frequent but more intense storms and longer inter-event droughts (Bates et al. 2008). Predicted warming air temperatures will exacerbate the effects of drought by increasing the rate of evapotranspiration. Drought may play a critical role in compounding the effects of sea level rise by reducing freshwater inputs to estuaries that deliver large quantities of suspended sediments critical to the accretion of wetland soils.

On the other hand, coastal processes are difficult to predict and many other natural factors affect streamflow and produce local and regional variations in drought severity. Factors such as physiographic characteristics and location of the watershed may affect evaporation rates, the influence of sea breezes, or the likelihood that tropical cyclones will provide drought relief.

C. Drought Adaptation

Discussions of adaptation measures are somewhat general in nature and intended to promote ecosystem resilience and adaptability. They fall primarily in the categories of water and/or land planning and management. The interaction of the two main stressors for drought impacts (altered precipitation patterns and water management) can act synergistically to create cumulative impacts or act "commensally" whereby water management can to some degree ameliorate the adverse effects of climate-related droughts. This capacity is a key resource for the development of drought adaptation measures for ecosystems. Among the key elements is water basin planning that includes developing knowledge of adequate flows to maintain river and stream system ecological functions with minimal degradation. This would enable informed decision-making for basinwide water allocation planning purposes during prolonged drought situations.

IV. Literature Gaps and Research Needs

The most critical needs include examining drought impacts in ecosystems not studied by existing research, implementing long-term studies to identify and examine causal relationships, and developing a set of indicators with which to monitor ecological change and impacts during drought.

There is relatively little literature on the following systems relative to drought impacts: coastal upland forests (particularly from an ecological impact perspective), freshwater stream tributaries of tidal rivers, pocosins and other non-riverine wetlands, savannas, maritime (barrier island) systems, nearshore ocean, and estuarine high marshes. Additional work on tidal freshwater marshes in the southeast appears warranted, as is research related to drought effects on seagrass beds, organism disease vectors (e.g., Perkinsus infection of oysters), and invasive floral and faunal species.

Long-term or time series studies capturing multi-year and repeated drought events are needed to understand drought impacts, especially for ecosystem processes. At the levels of populations and communities, short-term studies may enable some generalizations about the biotic responses to drought, but more long-term studies are required to understand impacts on ecosystem processes (Lake 2003). While many of these are covered in the report, long-term studies controlling for outside stressors to scientifically define cause-effect relationships would be valuable in impact analysis and remediation. It would be beneficial to take advantage of available sites with already established long-term data sets for different coastal ecosystem types. Long Term Ecological Research sites, National Estuarine Research Reserves, National Estuary Program Sites, State and Federal Refuges, etc. would be ideal places for such research. An intensive and continuous monitoring program that builds on these resources would require an organized and coordinated, multi-state, multi-institutional approach.

The literature revealed use of existing drought indexes (e.g. PDSI, PHDI), computer modeling, and integration of data sets as useful tools for investigating the effects of drought. However, innovation in tools for discerning conclusive cause and effect relationships were somewhat limited. There is a need to improve understanding of key monitoring variables and to develop a set of indicators of ecological change that may be caused by drought. A search for leading indicators, and including those dependent on a particular ecosystem service that could be degraded or destroyed by drought, will be most helpful to resource managers. Indicator species sensitive to drought-induced changes should be selected for coastal ecosystem communities and studied long-term to develop baseline conditions for comparison with continued sampling during long-term or repeated drought situations. Many data sets of great utility may already exist and can be used with new or continued sampling during droughts to better determine cause and effect relationships. More research and information is needed regarding:

- Groundwater resources in the study area and resulting drought impacts on groundwater dependent ecosystems and baseflow in streams.
- The importance of droughts during different seasons. While numerous studies mention the importance of seasonal flows to support aquatic organism needs in estuaries, more work is needed, particularly in tidal freshwater systems.
- The longevity of droughts in relation to long-term impacts and/or length of recovery. This will enable better decision making for water management during drought periods, allowing more informed impact analysis to balance instream flow with other water needs.
- Potential responses to salinity pulsing, given the expected changes in sea level rise, hydrologic changes, and more frequent and severe saltwater intrusion events due to climate change. The results of ongoing or new studies on coastal impoundment efforts at controlling salinity in tidal freshwater communities (if these exist) would be very informative. Such studies should focus on the long-term ability to provide a controlled salinity regime while minimizing the impacts of reduced connections and exchange with the estuary. Many impoundments, particularly those in National Wildlife Refuges, are managed for particular biotic communities with specific salinity tolerances. Maps that provide Refuge managers with horizontal and vertical detail about projected changes in salinity are needed to help them plan relocation or other adaptive measures. This also necessitates a habitat modeling capability as an essential planning tool.

I. Introduction

A. Background

Recent droughts in North and South Carolina have resulted in a broad array of impacts across many sectors and interests including water systems, hydropower generation, agriculture, forestry, and environmental resources. The 1998-2002 drought resulted in record low stream flows, ground water levels, and reservoir storage that severely impacted water users across the Carolinas (NCDENR 2004; SCDNR 2003). More recently, North Carolina experienced its driest year on record during 2007, and South Carolina experienced its fifth driest year on record during 2008 (NCDWR 2008). Many parts of the Carolinas continued to experience variable climate patterns throughout 2008 and 2009, and streamflow, reservoir, and groundwater levels recovered slowly (NCDWR 2009). During the writing of this report in 2011, parts of the region experienced extreme drought conditions, including southeastern North Carolina and southern and western South Carolina. Below-normal precipitation in winter 2010-2011 was attributed to La Ninã (NCDWR 2011). Similar conditions were expected to occur during winter 2011-2012 (South Carolina Drought Response Committee 2011).

National-level drought recommendations cite the importance of identifying, assessing, and integrating environmental concerns and needs into drought planning and response (NIDIS 2007; Subcommittee on Disaster Reduction 2005). However, studies have also indicated that "…environmental resources often receive inadequate attention during drought emergencies and in drought planning, not so much because of lack of concern but because of lack of expertise in this arena, lack of adequate financial resources, and sometimes lack of awareness"(NDPC 2000, 29). In the Carolinas, while impacts on agriculture and water supplies have been relatively well studied, drought effects on aquatic and terrestrial resources are not well understood but can be devastating (Lackstrom and Dow 2010). Drought affects water quantity and quality, which then affects ecological resources. Resource agencies in both states have documented how recent drought conditions contributed to water quality changes and increased salinities in coastal areas (Van Dolah et al. 2006; Deaton et al. 2010). In South Carolina, high salinity conditions produced by the 1998-2002 drought adversely impacted several species reliant on aquatic resources (e.g., shrimp, crabs, wood storks, wading birds, salamanders) (SCDNR 2003). Deaton et al. (2010) noted that below-normal river discharges in 2007-2008 contributed to high salinities in coastal rivers that pushed estuarine fish further upstream.

In order to advance regional understanding of the environmental impacts of drought, the Carolinas Integrated Sciences and Assessment (CISA)[1] team conducted a workshop in March 2010 with researchers, resource managers, and education and advocacy specialists engaged in coastal resource issues in South Carolina. The primary objectives of the workshop were to identify drought impacts and vulnerabilities related to coastal ecosystems and to identify ways in which drought planning and preparedness activities could better address coastal ecosystem needs and vulnerabilities.

Several broad themes related to coastal ecosystems and drought emerged from the workshop. First, concerns about water quality and quantity during drought events focused on 1) increased salinity and saltwater intrusion, 2) reduced ability to flush [disperse] or assimilate pollutants, thus exacerbating risks of impairment, and 3) overall water quality changes. Second, ecosystem impact concerns centered on habitat loss or conversion and consequent impacts to species, such as effects on recruitment, distribution and migration patterns, and primary and secondary production. Saltwater intrusion, as well as low streamflows and water levels, contribute to impacts and are attributed to both natural causes (drought) and human actions (e.g., changes in dam releases due to drought). Third, participants cited the lack of knowledge and understanding about drought and its impacts on coastal resources as a major concern. Part of this concern stems from the lack of availability of appropriate drought-related data and information. Such data includes not only hydrological and climatological information but also socioeconomic data regarding population growth, land use, and development patterns. Without suitable data and information, an overall lack of understanding about drought limits both the scientific and management realms (Lackstrom and Dow 2010).

B. Report Objectives

Workshop attendees identified a need for greater availability of appropriate drought data and information to manage coastal resources during drought. The current state of knowledge for this topic is seen as poor and minimal. Participants recommended that a "State of Knowledge Report" would be an essential first step to improving understanding. This report is intended to provide a synthesis and analysis of the peer-reviewed literature for the state of knowledge of drought impacts on coastal ecosystems in the Carolinas. Its objective is to expand current, limited understanding of drought impacts on coastal ecosystems, to identify critical gaps, to inform future research efforts, and to suggest measures to facilitate drought adaptation for ecosystems. The report is limited to biological, chemical, and physical impacts and is not meant to address social impacts. We focus on studies conducted directly in the area of interest, and in a few instances include work from outside this area that added particularly pertinent information.

[1]CISA, one of 11 NOAA-sponsored Regional Integrated Sciences and Assessments (RISA) teams, seeks to advance scientific understanding of climate variability and change and improve society's ability to respond to climatic events and stresses. See http://www.cisa.sc.edu/ for additional information.

C. Drought

Drought as a climatic event refers to the occurrence and/or persistence of below-normal rainfall. Droughts are a normal part of climate variability and may occur at a variety of temporal scales – from days, months, seasons, or years. While the Carolinas normally receive abundant, annual precipitation, the two states also experience interannual variability, as well as seasonal variations, in precipitation and evaporation. Significant periods of drought in the study area occurred in 1925-29, 1930-35, 1950-57, 1965-1971, 1980-82, 1985-88, 1998-2002 (USGS 2002; Weaver 2005), and 2007-08. Climatic causes of drought in the Carolinas have been attributed to shifts in the Bermuda High and the presence of La Niña conditions. Weather patterns in the Carolinas are influenced by the Bermuda High, a high pressure system normally located in the central North Atlantic Ocean and which enables the transport of moisture from the Gulf of Mexico into the Southeast. Drought occurs when the Bermuda High shifts westward, thereby enabling moisture transport to the central portions of the U.S. rather than to the Southeast. This shift creates conditions (e.g. lack of soil moisture) which may inhibit the development of thunderstorms, an inconsistent, albeit critical, source of rainfall during the summer (Weaver 2005). La Niña has been linked to below-normal winter and spring precipitation in the Southeastern U.S, the time of year when water supplies normally would be replenished (Weaver 2005; Seager et al. 2009). It should be noted that not all droughts demonstrate similar characteristics and trends. For example, the 1998-2002 drought resulted from several years of below-normal precipitation which produced gradual or cumulative impacts, while the 2007-2008 drought was characterized by a quick onset and rapidly deteriorating conditions exacerbated by above-average temperatures in Summer 2007. Finally, other natural factors may affect streamflow and produce local and regional variations in drought severity. Factors such as physiographic characteristics and location of the watershed may affect evaporation rates or the likelihood of benefiting from storms. Coastal areas, for example, may be more likely to receive relief from tropical cyclones.

Droughts are frequently characterized in four ways. Meteorological drought refers to a prolonged precipitation deficit. Agricultural drought refers to soil moisture deficits and concomitant crop failures and impacts. Hydrological drought refers to a precipitation deficit that results in lower than normal surface- or groundwater levels and supply. Declining inflows to reservoirs and falling streamflow, lake, and monitoring well levels may indicate deteriorating hydrological conditions. High temperatures and evaporation rates may also exacerbate conditions. Groundwater conditions are particularly important for surface water resources because groundwater sustains and replenishes streamflows, lakes, and wetlands. While streamflow and groundwater levels normally decline during the summer and fall, lower evaporative losses and decreased water demand in the winter months helps to recharge both groundwater systems and reservoirs (Weaver 2005; Badr et al. 2004). Finally, socioeconomic drought encompasses droughts that adversely affect some social or economic good, i.e. resources that are used or valued by humans and impacted by abnormal meteorological, agricultural, or hydrological conditions. Examples include livestock losses due to lack of feed or drinking water, forestry and timber damage through forest fires or pest infestation, and water deficits that disrupt normal operations of water-dependent industries or businesses (Wilhite and Glantz 1985; Kallis 2008).

With the exception of "hydrological drought," this classification system is of some, but limited, utility in discussing drought severity relative to ecological systems. Generally, the four categories are related to the sector or impact of interest; however, they are often portrayed as occurring in a somewhat linear progression, although not specifically hierarchical. Drought severity and recovery may be a product of many factors and will vary from drought-to-drought and place-to-place depending on the timing, extent, and duration of drought conditions. Many of the droughts of interest in this report are multi-year droughts or "supra-seasonal" droughts, typified by longer-term rainfall deficits and declines in water availability. Of lesser concern are seasonal droughts that occur as regular periods of low flow (Lake 2003). In the study area for example, October through November are generally the driest months while April through May are second driest (Southeast Regional Climate Center[2]). These periods would be considered seasonal dry periods.

While we do expect droughts to occur as a normal feature of climate variability, the "unpredictable" drought is the drought of interest for most ecosystem effects studies (although the hardest to plan for in terms of studies). Despite some meteorological predictability for La Niña events, an "unpredictable" drought would be one that is of long duration or large spatial extent, and/or one that exhibits severe or intense conditions (e.g. as measured by drought indices such as the Palmer Drought Severity Index[3] PDSI) or Standardized Precipitation Index[4] (SPI) (Humphries and Baldwin 2003).

[2]http://www.sercc.com/climateinfo/historical/historical.html

[3]The PDSI is an index of meteorological drought that compares the actual precipitation in an area to what would be expected based on the average climate of the area (climate division) and the meteorological conditions during and proceeding the month in question (Palmer 1965). It is calculated through measures of precipitation, evapotranspiration, runoff, and soil moisture ans is considered useful for agricultural or other water uses that are sensitive to soil moisture conditions (Guttman 1998).

[4]The SPI measures drought in terms of precipitation deficits and can be shown at different time scales (e.g. 1-month, 9-month, 12-month periods), depending on user needs. For example, the agriculture sector may be interested in short-term droughts. Water supply managers may be more interested in longer-term trends and conditions (Guttman 1998).

D. Coastal Carolina Ecosystems

This report focuses on coastal ecosystems of North and South Carolina. There is significant ecological continuity between extreme southeastern Virginia and northeastern North Carolina and between coastal Georgia and the south coast of South Carolina (Fig. 1, Appendix). To ensure complete coverage of relevant research, we include work that was done in those parts of Virginia and Georgia. Within these coastal ecoregions, the report concentrates on the portion bordered inland by the extent of tidal freshwater. The actual extent of tidal reach extends far inland in many locations (e.g., the Great Pee Dee River in South Carolina) and may vary temporally by several kilometers depending on tides and river discharge. Further, in light of sea level rise, the landward extent of tidal influence is in all probability moving inland. So the study area boundary is somewhat indistinct and in some cases it is not possible to be certain on which side of the boundary a

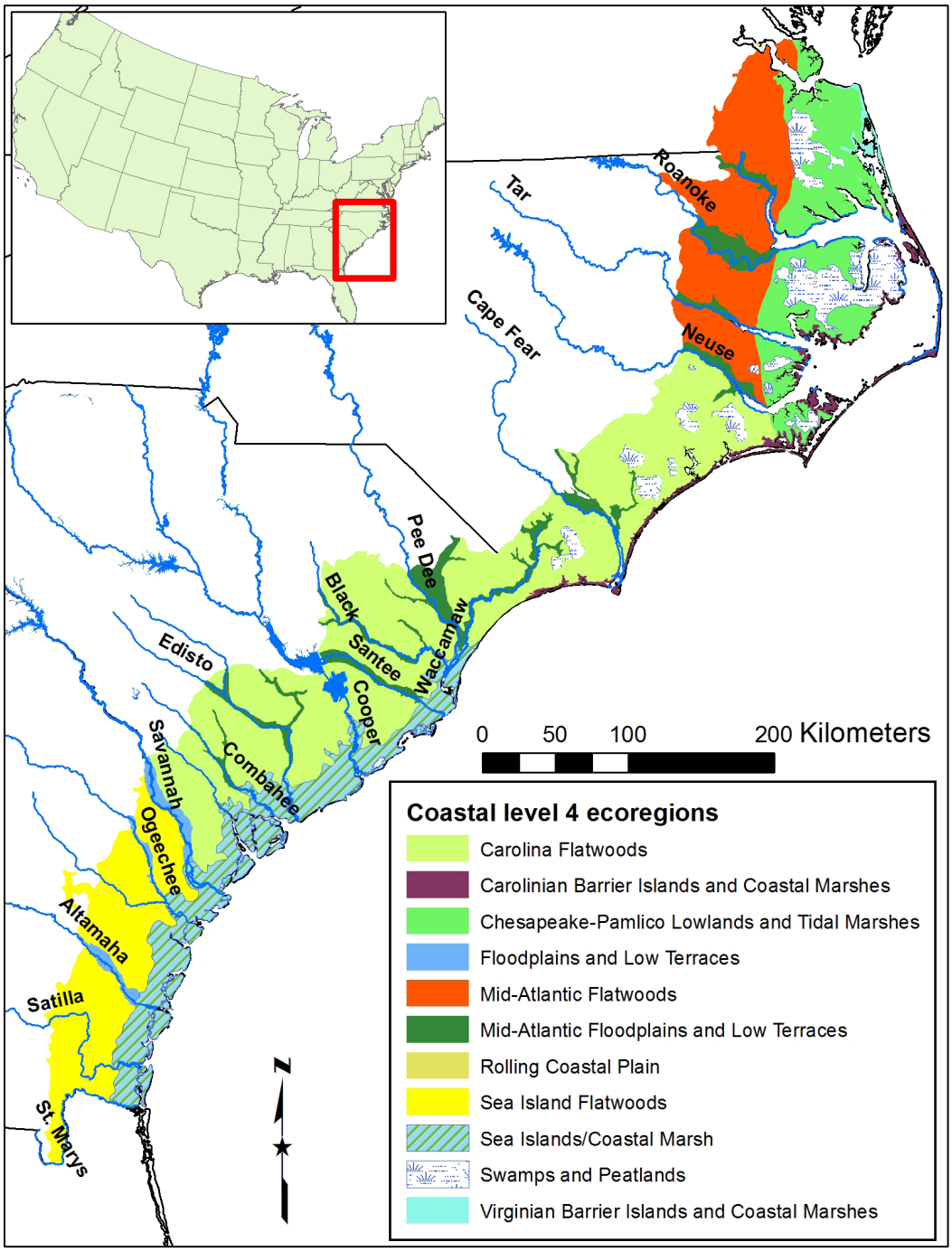

Figure 1. *Study Area. The study area includes parts of the Southeastern Coastal Plain, Southeastern Plains, and Southern Coastal Plain Level III ecoregions and the Level IV ecoregions indicated here. Also included are the major rivers that drain to the coast. Additional detail appears in the Appendix.*

study site was located. Additionally, many features such as some wetland and forest types occur on both sides of the boundary. There has been much work, for example, on Carolina Bays on the upper Coastal Plain and much less on those clearly within our study area. We chose to focus as much as possible on research within the designated study area because even though general characteristics of specific ecosystems may be similar to those further inland, the coastal instances are subject to special factors. These factors include occasional inundation with salt water during storm surges (e.g., Conner and Inabinette 2003) and differences in the chemical composition of precipitation (e.g., Peierls and Paerl 1997; Willey et al. 1988) that may affect their response to drought conditions.

Component Systems

The ecoregions along the US Southeast coast contain many natural subsystems and the literature is replete with classification systems that vary from fairly inclusive systems based on Cowardin et al. (1979) (e.g., McKenzie and Barkley 1980), or on botanical communities and compositions (Nelson 1986; Schafale and Weakley 1990) to simpler systems (e.g., National Ocean Survey 1979). For our purposes, a fairly simple system for report organization was chosen as follows.

Coastal Marine. This system is directly influenced by the Atlantic Ocean and generally has salinities exceeding 30. It includes the intertidal beach, surf zone and waters, sediments, and structures on the continental shelf to the 3-mile State territorial limit. Seaward of the turbulent surf zone, the shallow shelf is composed of soft, live (hard) bottom, and pelagic (water column) habitats.

Maritime. The maritime system is primarily an upland system located on barrier islands and includes bird key and bank[5], maritime dune, transitional shrub, and maritime forest subsystems. Components of this system are influenced to varying degrees by salt water and may contain internal wetlands and ponds.

Estuarine. The estuarine system contains intertidal and subtidal components and extends upstream from its mouth (connection with the ocean) to where waters have a salinity of less than 0.5. The subtidal component includes open water sounds, bays, tidal rivers, streams, and creeks with continuously submerged substrates and salinities greater than 0.5. Intertidal parts of the estuary are those areas intermittently flooded and exposed by tides and include salt and brackish marshes, intertidal creeks, mud and sand flats, and oyster reefs.

Tidal Freshwater Riverine, Forested Floodplain and Marsh. Tidal freshwater systems occur in sites where flooding occurs in response to lunar or wind tides and the salinity is less than 0.5. These areas manifest where river freshwater inflow and distance from seawater inlets are such that waters remain fresh yet are still tidal. Tidal cypress-gum swamps typically occur at the mouths of large rivers and smaller creeks and occasionally along sound shorelines. Tidal freshwater marshes occur in the lowermost parts of some tidal rivers and creeks and in large flats along the shorelines of freshwater sounds. Although we agree with the majority of literature that includes tidal freshwater systems as part of the estuary, we have broken this zone out for separate discussion in this report due to its particular sensitivity to drought and concomitant salinity changes.

Coastal Impoundments. This is a special category having derived from anthropogenic manipulation during the rice culture era in southern coastal areas. Large areas of swamp were diked and cleared primarily using slave labor. Interior cross-embankments and drains were emplaced to allow greater manipulation of water levels in the created compartments. Intact control of tidal waters varies among coastal impoundments. Many have reverted to fully tidal systems as embankments have eroded. Relict impoundments with breached or eroded nonfunctional dikes will be considered tidal marshes for this report (considered within the estuarine or tidal freshwater system depending on salinity). Others remain as managed wetlands, primarily for the attraction of waterfowl and other wading birds and have manipulated tidal exchanges. Coastal tidal impoundments occur in salt, brackish, and freshwater zones.

Non-Alluvial Wetlands. These areas are wetlands that do not have an obvious surface connection to flowing waters. They can be either depressions (e.g., Carolina bays or pocosins) or flats (e.g., savannas, wet pine flatwoods). Many systems are relict beach/dune and consist of ridge and swale topography with cypress/tupelo and/or hardwood associations. While vegetation may vary widely depending upon a number of factors, hydrology in these systems is maintained by precipitation and interaction with the groundwater table.

Uplands. This category includes those non-maritime areas with non-hydric soils. Vegetation is predominantly mesophytic or xerophytic. Natural uplands within the coastal zone defined by this report include mixed pine-hardwood forests, and longleaf and loblolly pine forests. Modified uplands include pine plantations, and developed areas.

[5]Bird keys and banks are small isolated islands that usually occur in tidal inlets and broad bays. They are very dynamic habitats because they are susceptible to over wash by storm action and spring tides and because they tend to migrate in response to inlet morphology (Sandifer et al. 1980). They are important habitat for colonial nesting birds.

II. Conceptual Ecological Model of Drought Impacts

We developed a simple conceptual ecological model as a framework to help visualize the flow from system drivers and stressors through ecological effects or causal links and eventual grounding in key attributes or values of the system (Fig. 2). We developed the model at the beginning of this project to help identify our current state of knowledge, organize the report, guide the literature search, and identify gaps in the literature. Based on the literature, we adjusted the model to incorporate additional causal effects uncovered during the literature review. Drawing from Ogden et al. (2005) we organized the model according to four major components: drivers, stressors, ecological effects, and attributes. These components are described in more detail below.

Drivers refer to the major, external factors or processes that have a large-scale influence on natural systems. These forces can be natural or anthropogenic. The two primary drivers of drought impacts in coastal ecosystems in the Carolinas are climate variability and water management. Because of its potential to significantly contribute to increased salinity in ecosystems discussed in this review, we also include sea level as a compounding driver.

Drought. Droughts are a normal part of climate variability and most ecosystems have evolved mechanisms to deal with prolonged precipitation deficits within non-extreme parameters. However, recurrent and persistent drought, or "an unpredictable low-flow period which is unusual in its duration, extent, severity or intensity" (Humphries and Baldwin 2003, 1142), may be a major ecosystem driver that has a large-scale influence on natural systems through influence on a number of stressors.

Water Management. Systems of water management can also create conditions of water shortage or scarcity. For coastal systems, upstream water use can affect the amount of water returned to a river system and the amount (and quality) of water available to downstream users. Dam and canal construction, river and stream channelization, and filling and draining of wetlands modify natural hydrologic systems. Most river systems in the Carolinas have been modified by large hydroelectric projects. Management and operation of these projects affect the timing, duration, and volume of water flows (Richter et al. 2003). The water management anthropogenic driver can work in concert with climate driven changes to either compound or reduce intermediary ecological effects and impacts on system attributes.

Sea Level. Sea level affects the quantity of saline waters delivered to estuarine systems and in concert with freshwater precipitation and runoff influences the resultant salinity levels experienced by these systems. Salinity and flushing of pore water in estuarine marshes affects geochemical processes and productivity, distribution and survival of these marshes. Relative Sea Level Rise (RSLR) encompasses land subsidence plus the worldwide change in sea level (Morris et al. 2002; Hughes et al. 2009; Kirwan et al. 2010). Among the scientific community, it is expected that RSLR due to climate change, resulting in warm water expansion and melting ice, will remain a persistent threat in the long-term.

Stressors represent physical or chemical changes internal to the natural system that are caused by the external driver(s) and that contribute to significant changes in biological components, patterns and relationships in the system. Whether caused by abnormal precipitation or water management decisions, coastal ecosystems are affected by lower than normal freshwater inflows, reduced groundwater levels, or soil moisture deficits. Precipitation shortfalls may act as a stressor on water quality by contributing to increased concentrations of pollutants or altered sediment delivery. However, other compounding stressors that would interact with drought to exacerbate effects, but perhaps not to the level of sea level rise (e.g., temperature, subsidence, and anthropogenic alterations), are not shown in the model to reduce its complexity.

Ecological Effects refers to the physical, chemical, and biological responses caused by stressors. Ecological effects represent the pathways through which the stressors alter or affect attributes so they may be viewed or termed as "causal linkages." The arrows between stressors and causal links, between the causal links themselves, and between causal links and attributes in theory set up "rebuttable presumptions" that can be explored in the literature or resolved through additional research. It should be noted that many additional causal linkages (e.g., altered marsh elevations, altered soil composition, increased freshwater transit times, altered dissolved oxygen levels) are either contained within a more general stressor or were excluded for diagrammatic clarity.

Attributes are the elements or indicators that represent the overall conditions of the natural system and the known or hypothesized effects of stressors. They usually represent important human values regarding, or uses of, the natural system. By following causal links in the diagram, the pathway from drivers through stressors to the attributes chosen to represent key ecological values of the coastal systems is diagrammed. We chose the attributes of Water Quality, Biodiversity and Fish and Wildlife Populations, and Estuarine Nursery Habitat And Biota to represent the Coastal Ecosystems in our study area.

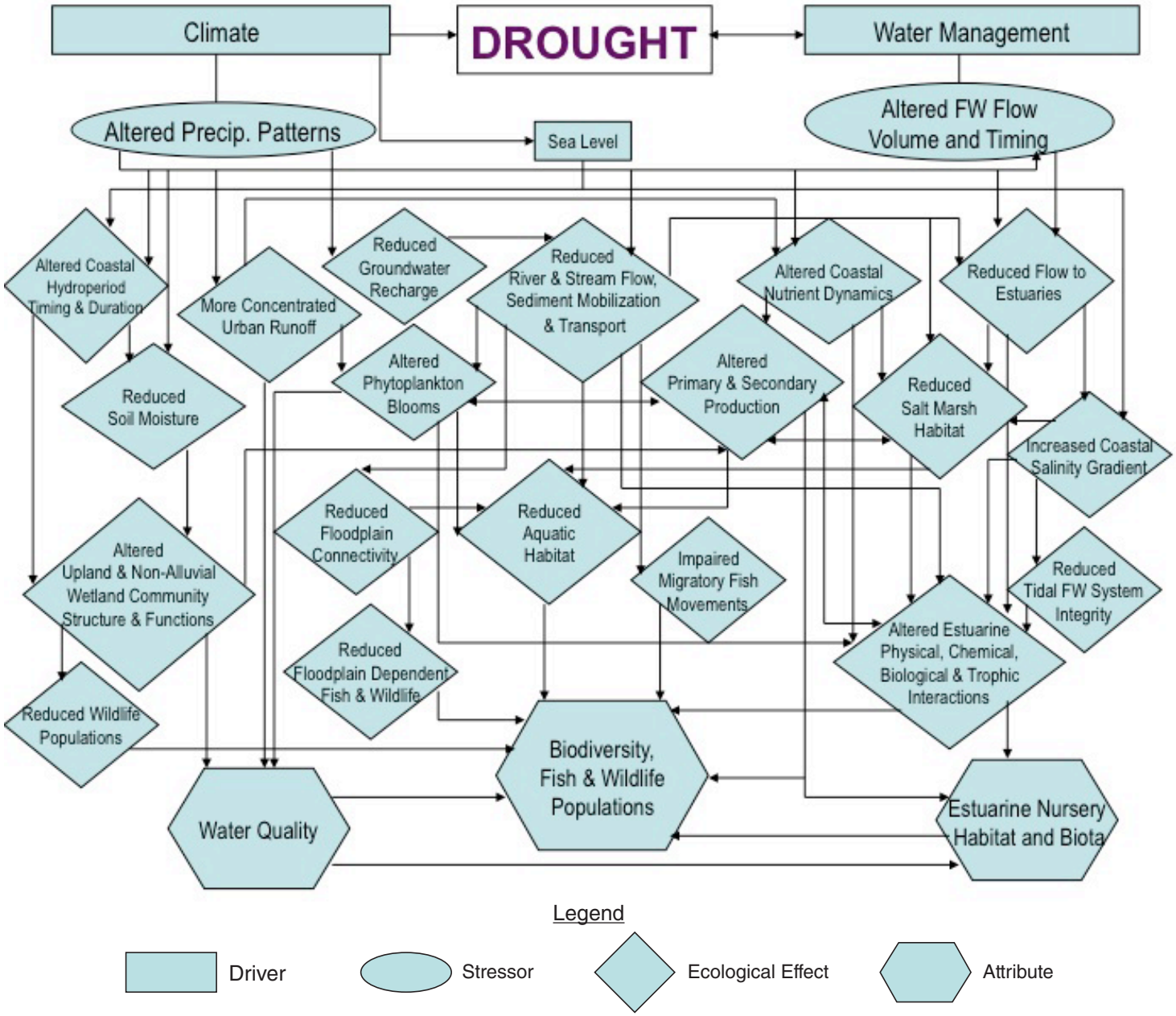

Figure 2. *Conceptual Ecological Model for Drought Impacts on Coastal Southeast Atlantic Ecosystems.*

III. Coastal Marine

We found only one report that addressed the direct effects of drought on near-shore ocean systems (Durako et al. 2010). Colored dissolved organic matter (CDOM) was studied in the outflow plume of the blackwater Cape Fear River, North Carolina during a wet and dry two year period to describe how river discharge affects carbon sources and transformations in the near-shore waters. During high flow periods, the discharge plume reflected riverine inputs (abundance of CDOM, detritus, and nutrients) while decreased phytoplankton concentrations suggested light limitation in the plume induced primarily by high CDOM. During low flow periods, CDOM played a much lesser role in light attenuation. Durako et al. (2010) concluded that CDOM may exert a major limiting influence over near-shore primary production during wet years with high discharge. In a logical extension of these findings, extended drought periods would reduce light attenuation caused by the plume resulting in higher near-shore primary production, although nutrient availability may mitigate this effect.

We are not surprised by the limited amount of research addressing direct impact of drought conditions on the near-shore system as any drought impacts to this system are likely to be secondary in nature, perhaps with the exception of direct point source discharge of urban drainage to the ocean. These impacts may include:

- Reduced river flow will lower nutrient inputs and urban runoff including associated pollutants transferring to this system through estuaries (see estuarine section, V). However, while changes in precipitation patterns predicted by climate change models include less frequent rain events, they also predict these less frequent events to have much heavier rainfall. This may counter the effects of drought-reduced inputs by increased flushing of built-up urban contaminants and increased soil erosion coupled with less nutrient transformation time in the estuaries.
- Most coastal marine species are dependent upon estuarine habitat for part of their life cycle. As described below in the estuarine section (V) of this report, sustained drought may affect estuarine nekton related habitats thereby impacting coastal marine biota.
- As referenced elsewhere in this report, drought impacts on estuaries and tidal freshwater streams (and inland rivers not covered in this report) will affect the ability of anadromous species to reach spawning grounds resulting in poor year classes. Since the young of these species are purported to support the marine food web on their out migration, there may be a negative impact on marine biota.

IV. Maritime

A community profile of the maritime ecosystem developed by Bellis (1995) may be considered the seminal work on this system. The system is described as "dominated by broadleaved evergreen trees and shrubs that occur in a discontinuous narrow band along the barrier islands and on the adjacent mainland from North Carolina to Florida" (Bellis 1995, 1). Flora and fauna typically consists of species that are well adapted to withstand the stresses of high salt spray exposure, wind damage, limited fresh water, soil erosion and dune migration, and periodic seawater inundation. This system covers the more stable undeveloped portions of barrier islands and coastal dune ridges and provides refugia for wildlife, storage capacity for groundwater, and help with soil stabilization.

Numerous community types fit into the maritime forest ecosystem (Bellis 1995). These include maritime shrub, maritime evergreen forest, maritime deciduous forest, coastal fringe evergreen forest, coastal fringe sandhill, maritime swamp forest, maritime shrub swamp, and interdune pond. As the names imply, these different communities may have very different hydrologies ranging from xeric to permanently flooded. As a result, the effects of drought will vary significantly in these different communities.

The literature does not directly cover drought effects on maritime communities to any great extent. What research does exist discusses the role of precipitation on barrier islands. Precipitation appears to be the only source of freshwater input and is critical to several of these communities. Consequently, extended or repeated drought is liable to result in significant community changes. Underneath the barrier island, fresh water tends to float as a lens over the underlying salt water. McPherson (1988) studied succession in a maritime forest system on Cumberland Island, Georgia. He concluded that fluctuations in water table depth controlled the wetter marsh/forest and drier scrub/forest community boundaries and soil moisture controlled succession of scrub to oak-palmetto forest. However, Bellis (1995) stated that biologists differ on the causes, direction, and rate of transformation among plant community types and that the relative contributions of factors such as climate, soil characteristics, pioneer species, and habitat disturbance have not been determined.

Freshwater ponds often are the only dependable source of water for animals on barrier islands. Such animals include frogs, salamanders, water snakes, turtles, aquatic birds, and aquatic mammals (Bellis 1995). Associated freshwater wetlands expand habitat diversity. Based on observations on Cumberland Island, Georgia, Hillestad et al. (1975) indicated that when these ponds are present, many of the above aquatic fauna also provide a varied and more dependable food source for nonaquatic inhabitants.

Mayes and List (1988) indicated concern over possible damaging effects of periodic drought conditions on maritime ponds. However, Hillestad et al. (1975) suggested that these ponds are "pulse-stabilized" aquatic systems and that water-table oscillations as well as other perturbations (e.g., drought and fire) are necessary to maintain them. Without these perturbations, shallow water wetlands would fill with organic matter and convert to shrub or swamp forest. The exposure and drying of the shallow pond bottoms accelerates aerobic decomposition, releasing nutrients that support later wet-season productivity. In a study of Nags Head Pond,North Carolina, Davison (1988) found that prolonged lowering of the water level permitted establishment of loblolly pine (*Pinus taeda*), redbay *(Persea borbonia*), Carolina willow (*Salix caroliniana*), black gum (*Nyssa sylvatica*), and red maple (*Acer rubrum*) seedlings. The latter three species can survive seasonal flooding and may become established around the pond margins after the water table returns to surficial levels.

V. Estuarine

For the purposes of this report, we consider the estuarine system to start at the ocean/land margin and extend upstream to the limit of tidal waters. This includes tidal freshwater systems as part of the estuary in concert with the majority of estuarine research. However, since the effects of drought on the tidal freshwater system differ somewhat from more saline parts of the estuary, we have opted to address them in a separate section. Dame et al. (2000, 793) provided a good description of southeastern estuaries: "Estuaries of the southeastern Atlantic coastal plain are dominated by shallow meso-tidal bar-built systems interspersed with shallow sounds and both low flow coastal plain and high flow piedmont riverine systems." In the study area of this report, two general geographical areas can be discriminated: the sounds of North Carolina, and the alternating series of riverine and ocean dominated bar-built systems of South Carolina and Georgia. Dame et al. (2000) is an excellent source for further characterization of these systems.

A. Freshwater Inflow

A great deal of drought-pertinent estuarine research focuses on the volume of freshwater inflow into estuaries. This research was largely triggered by concerns of water management (i.e., upstream manipulation of freshwater flow) for competing uses (e.g., dams, hydropower, water supply, etc.). We included this work because we consider impacts from low inflow a surrogate for drought conditions. As stated in the literature, the observed ecological changes resulting from drought may reflect changes occurring in response to manmade reductions in freshwater delivery to the coastal zone, such as from damming, irrigation, and human consumption (Wetz et al. 2011). However, it is likely that this parallel is more convincing relative to long-term drought conditions. Scavia et al. (2002) reported that changes in freshwater delivery alter estuarine stratification, residence time, eutrophication, and primary production. Moreover, Dame et al. (2000, 798) expressed: "Understanding the impacts of freshwater input to estuaries and the resulting changes in salinity regimes may be one of the most important challenges facing coastal scientists and managers. Salinity is a primary indicator of estuarine circulation because of its conservative character; it is also a significant determiner of biological productivity, faunal distributions, and habitat structure." It is important to note that impacts in regions experiencing drought may be exacerbated because water managers will likely be storing more freshwater within the watershed in response to drought (Scavia et al. 2002).

Literature on freshwater flow into estuaries became prevalent in the 1960's. Copeland noted (1966, 1839) "…freshwater input to estuaries is an important factor. Without it, estuaries become hypersaline and species composition can be altered drastically." A thorough review and overview of this issue is presented by Alber (2002). The schematic diagram below (Fig. 3) indicates the major effects of freshwater inflow on estuarine condition and resources. Quantity, as well as timing and quality, are all major influences. Drought conditions can greatly alter these influential freshwater inflow factors.

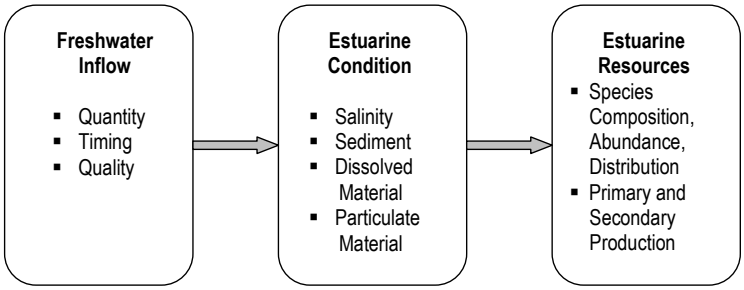

Figure 3. *Schematic diagram of the effects of freshwater inflow on estuaries (from Alber 2002).*

In addition to an upstream shift in salinity and hypersaline conditions, Alber (2002) identified a number of other effects on estuarine condition including:

- Decreased outflow can lead to expansion of the zone of transition from zero salinity to full seawater, hence lengthening the saline portion of the estuary. The example given to illustrate this point was the Altamaha and nearby Satilla Rivers in Georgia that have similar tidal regimes but different median flows (the Satilla's flow is one-tenth of the Altamaha's). Freshwater was encountered only 20 km upstream in the Altamaha as compared to 50 km upstream in the Satilla (Smith 2001).
- Alterations in freshwater inflow can change the hydrodynamic regime of an estuary. Decreases in discharge will serve to increase the influence of the tide on circulation patterns, such that a stratified system with well-developed gravitational circulation can shift to a well-mixed system where tidal exchange increases in importance. Such inflow-induced changes in stratification can in turn affect bottom water hypoxia.

- Decreased freshwater inflow can result in an increase in flushing or freshwater transit time with consequences for the buildup of pollutants and pathogens as well as effects on biogeochemical transformations (e.g., denitrification).
- Changes in inflow can lead to changes in estuarine geomorphology because freshwater is generally a source of sediments. This can result in losses for tidal deltas, benthic communities, and intertidal habitat. Upstream dams that trap sediment can exacerbate these effects. Decreased flood discharge or increased intervals between flooding events, can have the opposite effect as the erosive capacity of river floods diminishes. Under these conditions, the estuary tends to shrink its channel dimensions due to sediment accumulation.
- Changes in either the timing or quantity of freshwater input can have important effects on both the delivery of dissolved and particulate material and on their concentrations in the estuary itself, thus linking to profound effects on water quality.

Although the section above focuses on inflow reductions from both climatic and water management drivers, prolonged droughts alone may bring about ecosystem changes in estuarine conditions. A prolonged drought may shift estuaries into different salinity categories. For example, South Carolina experienced a prolonged drought from the end of the 1998 water year through the 2001 water year (Kiuchi 2002). Subsequent sampling of estuarine conditions in 2003-2004 by Van Dolah et al. (2006) indicated that salinity means in both tidal creeks and open water habitats were lower than those observed in surveys conducted in 1999-2000 and 2001-2002 (during the extended drought). Of greater note was the shift in the percentage of the state's estuarine waters that were considered to be oligohaline (salinity < 5) or mesohaline (salinity > 5 to < 18). This was 28% and 29% for tidal creeks and open water habitat, respectively, compared to < 11% for either habitat in the previous two surveys (during extended drought).

Ecosystem-level changes that result directly from changes in freshwater inflow are also addressed by Alber (2002). These changes include:

- Shifting isohalines caused by variations in freshwater inflow can affect the distribution of both rooted vegetation (e.g., upstream movement of *Spartina* species) and sessile organisms.
- As a given isohaline moves upstream, the channel width and the extent of intertidal habitat are often different, with consequent effects on the suitability of the new location for benthic organisms.
- Motile organisms can be affected as well. Different life stages of ichthyofauna and invertebrates have specific salinity requirements. Changes in the species composition, distribution, abundance, and health of fish and invertebrates have been attributed to changes in freshwater inflow. The longitudinal salinity gradient has been identified as the single most important factor linked to successful recruitment of larval and juvenile marine fish. Abundances of anadromous species (e.g., shad, herring and striped bass) have been correlated with inflow. Alteration of inflow timing can affect spawning and nursery cycles as many fish and shellfish life histories are cued to high spring runoff.
- Inflow-related changes in delivery of nutrients, organic matter, and sediment can affect estuarine productivity rates and trophic structure. A positive correlation between nitrogen loading with higher inflow and phytoplankton production appears numerous times in the literature. Conversely, decreased inflow can often be linked to decreased rates of both primary and secondary production. Decreased inflow and the consequent reduction in sediment delivery and turbidity have been shown to actually switch an estuary from a pelagic to a benthic food web. A two-year drought event in the Apalachicola River estuary in Florida (approximately a 50% reduction in river flow) had dramatic effects on trophic structure (see Livingston 1997).

Other work, with more specific reference to drought, reinforces and expands upon the estuarine and ecosystem-level effects from freshwater inflow changes presented by Alber (2002). This research is discussed in the following sections.

B. Nutrients and Planktonic Primary Production

In one of the few long-term studies, Wetz et al. (2011) described a 10-year time-series (2000-2009) investigation of the effects of drought on nutrients, water quality, plankton and biogeochemical measurements within the eutrophic Neuse River estuary in North Carolina. This period encompassed two severe droughts. During the droughts, dissolved inorganic nitrogen concentrations were 46–68% lower than the long-term mean due to reduced riverine input. Phytoplankton productivity and biomass were slightly below average for most of the estuary during a spring–autumn drought in 2002, but were dramatically lower than average throughout the estuary during an autumn–winter drought in 2007–2008. Mesozooplankton were one-eighth the abundance during the 2008 drought period compared to non-drought years. Below-average abundances persisted for a year after drought cessation with implications for secondary productivity and food availability for higher trophic levels.

The Wetz et al. (2011) study also addressed the seasonal differences in drought impacts. They found that the timing of the most severe stages of drought might determine its subsequent effect on phytoplankton communities and ultimately water quality. The 2002 drought was most severe during spring-summer and ended abruptly when an extended wet period began in the watershed. No long-term effects were detected. In contrast, the 2007–2008 drought was most severe during autumn-winter. Although Neuse River flow rates returned to normal at the cessation of the drought, they were not substantially higher than average as following the 2002 drought, and it took well over a year for system productivity to recover. Also, there was a small to moderate effect of the 2002 summer drought on phytoplankton, but a more substantial effect with the 2007–2008 autumn-winter drought. These differences may be a result of Neuse River estuary phytoplankton–nutrient relationships in that winter phytoplankton populations are much more reliant on riverine nutrients than those during summer (Paerl et al. 1998; Twomey et al. 2005).

Particularly in a eutrophic estuary, the question arises as to whether the reduction of nutrient input may be a positive effect of reduced freshwater inflow and lead to improved estuarine water quality. Paleoecological records showed a trend of more oxygenated estuarine waters during prolonged droughts (Cronin and Vann 2003). Following the 2007 drought, Wetz et al. (2011) found that the downstream region of the Neuse River estuary remained mostly devoid of blooms into 2009. However, relatively large phytoplankton blooms were observed in the mid- to upper-estuary during summer 2008 and winter–spring of 2009 as normal river flow resumed. This, along with intensifying stratification due to freshwater inflow, inevitably led to the development of hypoxic conditions in this region and consequently, significant fish kills were observed. From an estuarine habitat-management perspective, water quality may improve in some areas of an estuary while deteriorating in others as a result of droughts (Wetz et al. 2011).

Wetz et al. (2011) also raised an interesting question regarding the shift from phytoplankton production to benthic microalgal production. Most studies agree that drought and reduced nutrient input lowers phytoplankton production in the water column. However, in the Neuse River estuary, Fear et al. (2004) found that a 15% reduction in phytoplankton biomass would lead to a 20% increase in sediment surface area that was in the euphotic zone and therefore capable of microalgal production. This highlights the connectivity between benthic and pelagic processes in estuaries and the need to address both when assessing the effects of drought.

Similar nutrient reductions were noted in a ten-year study on the Neuse (Burkholder et al. 2006). This study was conducted from 1993-2003 and captured a three-year severe drought (2000-2002). Their analysis found that total nitrogen, total phosphorus, and dissolved oxygen were all significantly lowered when drought data was included in the ten-year study. However, concentrations of ammonium (NH_4^+) were not responsive to dry or wet conditions during the ten-year study. In North Carolina estuaries, Mallin (1994) found that wet winters were characterized by higher nitrate concentrations that reached the lower estuaries and resulted in increased phytoplankton primary production. There was lower phytoplankton primary production during dry winters.

Paerl and colleagues (Paerl et al. 2003; Paerl et al. 2007; Paerl et al. 2010) also promoted the idea of phytoplankton as a good indicator of ecological change. Community composition changes substantially between high and low inflow scenarios. With low flow, there is a dominance of dinoflagellates and cyanobacteria. These changes impact eutrophication, food web, biogeochemical (hypoxia), and habitat conditions. In data collected in the Neuse, Pamlico, Pungo, and New River estuaries of North Carolina, Fensin (2006) found that populations of harmful dinoflagellates increase with the beginning of a drought but decrease as droughts continue. Davis (2010) found that drought had significant effects on the assemblage of nitrogen fixing organisms associated with the roots of *Spartina*, indicating further potential nutrient cycling impacts. Because of the strong correlation between hydrological conditions and nutrient loading, Paerl et al. (2004) recommended that basin scale reduction strategies for nitrogen and phosphorus should include hydrologic variability (hurricanes, floods, droughts).

In an example of secondary drought impacts, estuarine nutrient inputs from colonial nesting wading birds (e.g., white ibis) are reported as much greater than riverine inputs (Bildstein et al. 1990; Bildstein et al. 1992). White ibis feed primarily on crayfish in bottomland hardwood swamps. White ibis populations were affected by a pre-breeding season drought. Nesting numbers and per-pair breeding success were significantly lower than normal in 1985. Nutrient import to the estuary was eleven times greater during the previous wet year (1984) than during the dry year (Bildstein et al. 1990; Bildstein et al. 1992).

Salt marsh along Leadenwah Creek, Wadmalaw Island, South Carolina (Almeda Kelley).

C. Salt Marsh

The literature on drought effects on tidal salt marshes concentrated in three areas: (1) the dieback of marsh that has occurred in the Southeast (and elsewhere); (2) community shifts in macrophyte composition; and (3) decreased marsh primary productivity as a result of higher pore water salinity.

1. Marsh Dieback

Marsh dieback has occurred at several locations within our study area as well as in Gulf Coast marshes. The term "sudden salt marsh dieback" was introduced to distinguish these events from those that can be explained by other types of disturbances, such as those caused by wrack damage or herbivory. The term is defined as: "a phenomenon that causes plants to undergo rapid senescence, resulting in mortality", or more specifically, "a progressive decline that begins with thinning and browning of the above-ground material and leads to plant death over a period of months" (Alber et al. 2008, 2). Studies implicate drought, perhaps in combination with other stressors, as a potentially strong factor in this phenomenon. In this section we review research that has investigated the linkages between drought and marsh dieback.

Alber et al. (2008) presented an overview of recent decadal marsh dieback events. Along the Southeast Atlantic coast, a 2001 event in coastal Georgia affected both *Spartina alterniflora* and *J. roemerianus*. Effects include noticeably thinning vegetation that deteriorated to bare mud with rhizome stubble. By 2002, dieback sites were reported throughout the Georgia coast and into South Carolina. This phenomenon is not at all limited to the southeast Atlantic coast and the overview also includes a major dieback (affecting over 100,000 ha) in Louisiana in 2000 and another less severe dieback in 2007. Schrift et al. (2008) attributed the large loss of *S. alterniflora* marshes within the Mississippi River Deltaic Plain in 2000 to drought disturbance. Although some areas recovered after this expansive disturbance, in many cases, salt marshes were converted to unvegetated mudflats that eventually subsided. The magnitude and severity of this and other Louisiana marsh dieback prompted restoration experimentation with hydraulically pumped dredged sediment slurry additions to the marsh surface (see Stagg and Mendelssohn 2010). Dieback has also occurred in the Florida panhandle, eastern Long Island, New England, and California (Alber et al. 2008).

Earlier instances of marsh dieback generally had many common characteristics that distinguish them from the most recent decadal events (Alber et al. 2008). Most recorded events found that marsh dieback occurred in low-lying areas of the marsh where soils tended to be waterlogged. Sometimes, but not always, a reduced soil environment with high sulfide concentrations was observed. Dieback also usually occurred in the interior of the marsh. When transplant studies were conducted, survival was low to moderate in dieback areas.

It has been suggested that sudden dieback in the most recent cases is associated with drought (Alber et al. 2008). The brown marsh[6] (Louisiana description equivalent to dieback) event in Louisiana coincided with one of the most intense droughts in 100 years and the time leading up to the dieback in Georgia, from 1999 to 2001, was the driest three-year period on record. Investigators at a joint Louisiana–Georgia workshop identified drought as a likely factor involved in dieback although there were differences between the dieback phenomena at the Gulf and southeast sites (Georgia Coastal Research Council 2004).

Alber et al. (2008) used two lines of evidence to explore this hypothesis: (1) use of the monthly values of the PDSI for the climate division where each event occurred; and (2) a similar analysis of long-term trends in temperature and precipitation (1950–2007) using data from weather stations located near each reported dieback site. Precipitation during the reported dieback tended to be low in the southeast but not in most other areas, supporting the results of the PDSI analyses. There were no clear linkages between dieback and temperature. However, the evidence for drought being the cause of dieback events is still unclear. The data in Alber et al. (2008) reinforced the notion that recent dieback events in the Gulf and the southeast were associated with drought, but it did not support this idea for the mid-Atlantic and northeast, although drought could not be completely ruled out in these areas. Ogburn and Alber (2006) found that the causes of salt marsh dieback are likely varied and remain unclear in most cases, and the causes of dieback events in North Carolina, South Carolina, Massachusetts, and Virginia have not been determined.

If drought can be linked to dieback, it is important to determine the mechanisms that cause plant dieback. Water column salinity stress acting alone is dubious as *S. alterniflora* can tolerate fairly high salinities – work in Georgia marshes indicated that salinities of up to 60 were encountered in *S. alterniflora* dominated marsh zones (Pennings et al. 2005) even though a salinity of 60 coincides with the upper limit of laboratory determined salinity tolerance for this species (Linthurst 1980). Also, salinities sampled during dieback were significantly less than 60. There is much discussion in the literature on the synergism of effects or what may be called cumulative impacts. It is conceivable that drought or other

[6] Louisiana equivalent term to dieback

stressors could make plants more susceptible to conditions that are normally tolerated. McKee et al. (2004) hypothesized that soil desiccation, increase in soil salinity, and the uptake of toxic metals (i.e., iron or aluminum) caused acute salt marsh dieback in the Mississippi River deltaic plain. Greenhouse studies in Louisiana demonstrated that *S. alterniflora* is susceptible to the combined effects of drought, reduced pH, and bioavailable metals – all of which were sublethal when taken individually (Mendelssohn et al. 2005). Brown et al. (2006) showed that the effects of salinity and drying are not additive, but that both impacted root and shoot dry weights and reduced nutrients. Survival was 100% with all individual stressor treatments; survival was reduced to 71% with high salinity and drought acting synergistically. Survival was further reduced to 29% in a longer-term, 24-day combined stressor exposure (Brown and Pezeshki 2007). They concluded that these combined stressors over an extended period could cause marsh browning[6] as noted in the field.

Other studies have considered trophic interactions. Dietl (2006) presented drought induced high soil salinities and acidity as the primary cause of salt marsh die-off. However, the author also postulated that overfishing of crabs, resulting in reduced predation on the marsh grazing snail *Littoraria irrorata*, may be the primary factor. Silliman et al. (2005) indicated that induced stress (drought) and grazers (snails) acted synergistically to cause die-off. Silliman et al. (2009) also suggested that dieback is consumer-controlled and target anthropogenic disturbances (i.e., intensification of drought events via climate change and overharvesting of snail predators, e.g., blue crabs). However, Alber et al. (2008) dismissed *L. irrorata* as the primary cause of dieback. While *L. irrorata* are capable of denuding an area of marsh of aboveground biomass if in sufficient abundance, they do not graze plant roots that were destroyed in typical sudden dieback scenarios.

Cahoon et al. (2011) conducted a study in Texas and Louisiana marshes during and after extreme drought. They found that changes in local hydrology and groundwater conditions can cause short-term perturbations to marsh elevation trends through shrink–swell of marsh soils. In Texas, a rapid decline in marsh elevation was related to severe drought conditions, which lowered local groundwater levels. “It appears that extended draw-downs lead to increased substrate consolidation making it less resilient to respond to future favorable conditions” (Cahoon et al. 2011, 463). (Also see work by Morris et al. (2002) in the marsh productivity section (V.C.3) below for further discussion of the criticality of marsh elevation to long-term marsh stability.) Overall, water storage changes led to rapid and large short-term, marsh-elevation impacts that were as much as five times greater than the long-term elevation trend. These findings indicate the importance of long-term, high-resolution elevation data sets to understand the prolonged effects of water deficits on marsh elevation change. This drought induced a change in marsh elevation, which could, when coupled with sea level rise and/or other stressors, create another synergistic dieback scenario.

In support of the cumulative stressor theory for marsh dieback, Alber et al. (2008) stated that it could be that a combination of changes in soil chemistry and pathogens interact with plants in different stages of vulnerability. Or, ongoing stressors such as eutrophication and/or sea level rise may have made marsh plants susceptible to drought.

Alber et al. (2008, 9) concluded the overview of marsh dieback with the following: “There have been increased reports of dieback in recent years, but it is not clear if this is because dieback is happening more frequently or that people are more attuned to the phenomenon. However, the fact that sudden dieback occurs in multiple marshes over large areas makes it of considerable ecological and societal importance. Although many areas affected by dieback are recovering, some areas persist, particularly sites that have subsided/eroded so that they are now mudflat rather than marsh habitat.” Brown and Pezeshki (2007) found that the critical time for recovery from the combined effects of increased salinity and soil drying may greatly diminish after two weeks from the onset of stress conditions. Mendelssohn and Morris (2000) detailed further cumulative abiotic factors that create stress on marsh production. This information is discussed in the salt marsh productivity section (V.C.3) below.

2. Marsh Community Shifts

While community shifts as a result of salinity changes and freshwater inflow manipulation are reported in the literature, discussion of drought as the primary causative agent is not prevalent. However, it is intuitive that drought and resultant reductions in freshwater inflow are directly linked to salinity change. Moreover, it is difficult to determine if an observed change in vegetation structure and composition is part of natural variation of the system and temporary, or if the change is longer-term. Strong and highly variable environmental gradients (i.e., hydrology and salinity in the case of tidal marshes) further confuse this determination (Wetzel and Kitchens 2007).

The Savannah River, running along the border of Georgia and South Carolina, has perhaps the southeast Atlantic coast's longest-term study of community shifts due to salinity changes. Wetzel and Kitchens (2007) reported on vegetation changes in a Savannah River tidal freshwater, oligohaline, and mesohaline marsh plant communities over an 18 year period, especially as driven by two events: (1) the removal of a tide gate after 14 years of operation and (2) a four-year regional drought. The tide gate removal had a much greater effect than the drought on interstitial salinity levels and was most intense in the oligohaline marsh where long-term conversions occurred. Community shifts in freshwater and oligohaline marsh induced by the drought were temporary, lasting from 1 to 3+ years. Neither event resulted in long-term alterations of the mesohaline plant communities indicating some resilience of estuarine vegetative communities to long-term drought.

A reciprocal transplant study between different saline zones in Savannah River marshes demonstrated that salinity gradient shifts can alter plant community structure and composition (Wetzel et al. 2004). The rate of community change of the transplanted sod depended on the direction of displacement along the salinity gradient. Community change of donor sod from brackish areas to freshwater areas took about 18 months and appeared to be controlled by biotic factors such as vegetative expansion and interspecific competition. Community changes for the reciprocal transplant (freshwater donor sod to more saline areas) were controlled primarily by environmental factors (i.e., salinity stress). Change was fairly rapid (6-10 months) and caused by the decline or death of the freshwater species. Species loss increase in this reciprocal transplant was proportional to the salinity difference between the donor and receiver plots (5% of species lost with a mean salinity difference of 0.14 and 71% species loss with a mean salinity change of 5.56). Wetzel et al. (2004, 88) concluded "…sustained fluctuations in the salinity gradient can alter plant communities in as little as six months but 18 to more than 30 months were needed for the transplanted sod to resemble the surrounding plant community."

In order to document the effect of a recent drought and the resulting marine intrusion event on plant communities, Visser et al. (2002) collected data sets in Barataria estuary, Louisiana in 1997 and 2000. Community shifts were compared to surface salinity changes at four points along the estuarine gradient within the study area. Surface water salinity increases were reflected by a change in species composition in the mesohaline to fresh marshes. The largest species composition shift observed was the shift from oligohaline wiregrass (species rich vegetation type dominated by *Spartina patens*) to mesohaline wiregrass (vegetation type dominated by *S. patens* with few other species).

White and Alber (2009) reported on a study in the Altamaha River Estuary in Georgia. This is a well-mixed estuary where salinity is approximately 20 at the mouth and decreases to zero approximately 20 km upstream. There are typically three marsh zones (locations set as a pre-drought baseline for the purposes of this study). Moving upstream from the mouth, the salt marsh zone extends upstream 6 km and is dominated by *S. alterniflora* extending upstream 6 km. A brackish zone occurs 6-16 km upstream of the mouth with mixed *S. alterniflora* and *S. cynosuroides*. *Zizania* and *Zizaniopsis* dominate a tidal freshwater marsh zone. They found the salt/brackish marsh division to have a salinity of approximately 15. A large decrease in freshwater flow and encroachment of saltier water into brackish areas accompanied the prolonged drought between May 1998 and September 2002. Three bankside vegetation surveys were conducted along the length of the estuary over a period of four years. An experimental removal program was conducted in the mixed Spartina brackish zone.

The restrictive 14-15 salinity line for *S. cynosuroides* moved upstream during the drought and *S. alterniflora* densities increased in the brackish mixed *Spartina* communities. Also, the downstream extent of *S. cynosuroides* shifted upstream about 3 km. Post-drought surveys in 2004 when salinity levels returned to normal indicated that *S. alterniflora* dominance continued in the mixed brackish marsh community, outcompeting *S. cynosuroides*. White and Alber (2009) concluded that *S. alterniflora* is a strong invader into new territory under higher-salinity, low-flow stress. The experimental removal program verified this. The two-year rapid upstream shift of *S. cynosuroides* suggests that its downstream limit may be a potential indicator of changes in flow conditions in the system. These studies supported the concept that the distribution of brackish/salt marsh plants is controlled on the downstream (saltier) end by the ability to tolerate physical stress and on the upstream (fresher) end by interspecies competition (Pennings et al. 2005).

Latham et al. (1991) explored another possible indicator of salinity changes, *Scirpus validus*, in the lower Savannah River. The spatial pattern and relative importance of *S. validus*, as well as the composition of co-occurring species, changed significantly with changing salinity. However, other factors, including differences in soil organic matter, elevation, and competition with other species, were also significant. *S. validus* is a generalist species and grows well over a wide range of salinities. As salinities increased, fewer species are physiologically tolerant, and *S. validus* occurred as a dominant species. As salinities continued to increase across the gradient, *S. validus* decreased in dominance and was replaced by *S. alterniflora*.

Tidal freshwater swamp converted to salt marsh, Huspa Creek, South Carolina (April Taylor).

3. Salt Marsh Productivity

Salinity is a major factor in *S. alterniflora* production and growth of *S. alterniflora* has been found to be weak in areas with high pore water salinity (Nestler 1977; Smart and Barko 1980). Accumulations of salt in intertidal sediments occur regularly in high marsh areas and also in lower marsh areas when low sea level reduces flushing. Salt marsh primary production decreases because of an increase in pore water salinity (Morris 2000). Salinity is a factor in the spatial and temporal variations in *S. alterniflora* productivity. Differences in soil anoxia and soluble sulfide affecting plant nitrogen uptake and assimilation affect spatial variability. Temporal differences are influenced by climatic and tidal controls on inundation (Mendelssohn and Morris 2000). The mechanisms by which this occurs are complex but basically result from interference with uptake of ammonium (the prevailing form of inorganic nitrogen in salt marsh sediments) and greater energy requirements for plant osmoregulation (Mendelssohn and Morris 2000). Sulfide toxicity and a resultant plant decrease in its ability to metabolize anaerobically via alcoholic fermentation further reduce productivity in waterlogged soils (Mendelssohn and Morris 2000). Mendelssohn and Morris (2000, 75) concluded their treatise on ecophysiological controls on *Spartina* productivity with the following: "During summers of unusually low mean sea level, soil salinities rise and primary production declines. It is during such events when rainfall will have the greatest impact on salinity and production. Thus, tidal, meteorological and climatic events have significant effects on the physical and chemical properties of salt marsh sediments, and these properties directly affect the physiology and productivity of marsh plants."

There is roughly a two-fold variation in salt-marsh primary production from year to year that is, in large part, driven by interannual changes in sea level and salinity (Morris 2000). Such losses in primary productivity are ecologically significant due to their role in trophic dynamics, nutrient cycling, and sediment accretion (Morris 2000). A rise in estuarine salinity levels due to drought conditions could manifest in similar high pore water levels of salinity and productivity losses. Maximum marsh production occurs in the summer months, coinciding with months of maximum evapotranspiration and pore water salinity (Morris 2000). Droughts (typical of summer months) would serve to further aggravate pore water salinity conditions. Indeed, in the North Inlet estuary, South Carolina, creekside *S. alterniflora* productivity (based on both above and belowground biomass measurement) seems to be correlated with rainfall and the concurrent depression of salinity (Dame and Kenny 1986). These authors further reported that the implications of grass biomass variation in wet years compared to dry years are striking.

The above scenarios can lead to a compounding long-term stability impact on salt marshes. Accretion of both sediment and organic matter is necessary to maintain intertidal marsh elevation and to compensate for RSLR as well as seasonal and annual variations in sea level. The long-term stability of these ecosystems depends upon interactions among sea level, land elevation, primary production, and sediment accretion that adjust the sediment surface elevation towards equilibrium with mean sea level (Morris et al. 2002). Productivity is important as an increase in production and biomass density will enhance sediment deposition by increasing the efficiency of sediment trapping (Morris et al. 2002). Should extended droughts decrease macrophyte production and related sediment accumulation, marsh stability relative to RSLR could be compromised.

In an example of the importance of rainfall in flushing marsh sediments, work by Walters et al. (2010) suggested that productivity in the high marsh may be positively affected by development runoff. Analyses of sediment characteristics of the Murrells Inlet Estuary, South Carolina suggested development of the upland boundary affected physical conditions within the high-marsh. For example, pore water salinities were greater along undeveloped shorelines during a non-drought period, even after rain events. While *Juncus roemerianus* is known as a generalist species, biomass production is greater when grown at relatively low soil salinities (Pennings et al. 2005). Development and concomitant increased freshwater runoff may be responsible for decreased pore water salinity and improved growing conditions for *J. roemerianus*. Conversely, it would seem that drought reduction of this runoff would result in higher pore water salinity, thus reducing high marsh production (Walters et al. 2010).

D. Estuarine Flushing/Water Quality

In general, estuaries are large filters or traps for materials that can be transformed by resident processes (Dame et al. 2000). In riverine estuaries, transport of dissolved and particulate material is generally seaward due to the net flux of water down slope due to gravity. The amount of time river water spends in the estuary before being discharged to the coastal ocean will influence the amount of material processing that occurs within the estuary.

Decreased freshwater inflow results in an increase in flushing time or freshwater transit time (Alber and Sheldon 1999; Sheldon and Alber 2002). Transit time provides a measure of the time it takes river water to travel through the system and has consequences for the ability of an estuary to flush out materials (Alber 2002). As transit times increase, the concentrations of pollutants and pathogens can increase as well. Transit time also sets the context for many of the biological and chemical processes that occur within estuaries, including nutrient transformations (Alber and Sheldon 1999). The major materials of carbon, nitrogen, and phosphorus are rapidly translocated, transformed, and remineralized within estuarine ecosystems (Dame et al. 2000). River flow, tides, and wind generated waves and currents are major components in translocating materials (Dame et al. 2000).

Freshwater transit time has been positively correlated with the fractional export of nitrogen from estuaries and negatively correlated with the amount of denitrification (Dettmann 2001). Slow flushing times have been implicated in the outbreak of harmful algal blooms (Alber and Sheldon 1999; Paerl et al. 1998), and reduced estuarine flushing has a potential for higher sediment chemical contamination (Hyland et al. 1999).

Southeastern riverine systems carry large quantities of dissolved and particulate materials to the coastal ocean (Dame et al. 2000). The extent of impact from altered freshwater inflow may depend on the type of estuary. Tidally dominated estuaries (bar-built and lagoonal) may export large amounts of inorganic and organic materials. Vogel et al. (1996) computed a first approximation of the relative contribution of tidal water exchange by bar-built and lagoonal systems and the discharge by riverine systems in South Carolina. They estimated that tidal water exchange was over an order of magnitude higher than river discharge and that nutrient exports by bar-built systems were at least an order of magnitude greater than riverine delivery to the coastal ocean. Water fluxes in all estuarine types appear to play a major role in the accumulation or lack of accumulation of anthropogenic materials, nuisance algae blooms, and marine diseases (Dame et al. 2000).

While flushing time is specific to freshwater (or materials dissolved in it) and represents the transit time through the entire system (i.e., from head of tide to the mouth), residence time is the average time particles take to escape the estuary (Sheldon and Alber 2002). While it is important to understand this difference, reduced freshwater flow into the estuary from drought events will affect both processes. According to Paerl et al. (2011), drought leads to low flow that leads to low mixing and longer residence time. Scavia et al. (2002) reported that the effects of altered residence times can be considerable. In estuaries with water residence times of less than a day, phytoplankton are generally flushed from the system as fast as they can grow, reducing the estuary's susceptibility to eutrophication. As residence times also will increase as a result of reduced freshwater inflow, susceptibility to eutrophication will increase.

Droughts may positively influence dissolved oxygen levels in tidal creeks. In a study of factors affecting dissolved oxygen in North Carolina tidal creeks, MacPherson et al. (2007) compared and contrasted the two major factors influencing drops in DO – water-column biochemical oxygen demand (BOD5) and sediment oxygen flux (SOF). BOD5 was driven by a set of parameters linked to warm season storm water inputs that stimulated organic seston loads, especially chlorophyll a, while SOF behaved less strongly so. Runoff processes that increase loads of organic material and nutrients and ground water discharges low in dissolved oxygen contribute to occurrences of low dissolved oxygen in tidal creeks.

The North Carolina coast is dominated by extensive and poorly flushed sounds (Dame et al. 2000). These estuaries are characterized by modest primary production, and they support the largest estuarine fishery in the southeastern Atlantic region. Industrial animal production contributes to high nutrient loading. This area includes several estuaries with persistent problems with pollution. In contrast, well-flushed bar-built and riverine estuaries are common in South Carolina and Georgia.

While reports of harmful algal blooms in South Carolina and Georgia estuaries are rare, blooms of cyanobacteria (blue-green algae) and dinoflagellates have resulted in significant ecological and economic damage in some rivers and estuaries of North Carolina. One recognized species of toxic dinoflagellate, *Pfiesteria piscicida*, was identified as the causative agent of approximately 50% of the fish kills in the Pamlico, Neuse, and New River estuaries of North Carolina in the past decade (Burkholder and Glasgow 1997; Burkholder et al. 1995; Burkholder 1998). However, some controversy, particularly around the life cycle of the organism and the contribution of hypoxia to the fish kills, currently exists. Paerl (1988) indicated that studies have emphasized the roles vertical and horizontal water column stability play in the buildup and persistence of bloom populations. Strong vertical stratification favors the accumulation of cyanobacteria and dinoflagellates at depths optimal for growth.

Cyanobacteria appear to be well adapted to eutrophic waters characterized by periods of thermal stratification and stagnation when climatic and nutritive factors favor growth.

On a slightly different note, Wetz et al. (2009) examined distributions of, and controls upon, formation of transparent exopolymer particles (TEP) (facilitated by phytoplankton blooms) in North Carolina's Neuse River Estuary over a 1-year period. TEP represent an important fate for phytoplankton production and play a critical role in sinking fluxes and food webs in aquatic ecosystems. TEP could be quite important in estuarine food webs by facilitating aggregation of small planktonic biomass (i.e., pico- and nanoplankton), thereby increasing their availability to higher trophic levels. TEP may also act as a vector for pathogenic microorganisms.

Based on their experiments, Wetz et al. (2009) speculated that the residence time of TEP was enhanced when drought causes low river flow, although decreasing water temperatures may have also played a role. At the end of the drought, increased flushing of the estuary with low cation river water created unfavorable conditions for TEP retention and reformation. Cation presence (linked to salinity) is also presented as a factor in formation and retention of TEP. Wetz et al. (2009, 453) concluded that "This would leave open the possibility that TEP may vary on timescales similar to those influencing freshwater delivery to estuaries/watersheds, including decadal (i.e., in relation to large-scale atmospheric fluctuations), interannual, seasonal, and event scale (i.e., storms), not to mention in response to longer term changes related to proposed climate change scenarios."

Anthropogenic alterations and water management may exacerbate the effects of low discharge in riverine estuaries. Goni et al. (2009) studied the characteristics and transport of particulate organic material in Winyah Bay, South Carolina under contrasting river discharge, tidal amplitude, and wind regimes. A persistent regional decadal drought (1997-2007) resulted in overall decreased levels of river discharge throughout all the months of the year.

Results showed a negative correlation between salinity and discharge, which was expected. However, total suspended solids also had a negative correlation with discharge, which was unusual, as this would not be the case for most estuaries. Low river discharge and a southwest wind resulted in a resuspension of bed sediments from shallows, highly altered organic matter, a northeast landward flux of salt, and sediment and particulate organic matter promoting efficient entrapment of materials in the estuary. The results indicated that Winyah Bay is an efficient trap for particulate materials under both low and moderately high discharge conditions and variable winds. Because of the dredging of the navigation channel, which artificially deepens the thalweg, only very high river discharges are likely to result in net export of particles at depth. Also, because of the highly engineered regulation of the rivers draining into Winyah Bay and the considerable water diversion rates in this area, such high discharge events are much less likely than in the past. Hence, they concluded that estuaries such as Winyah Bay have probably undergone major changes in their overall flushing rates and biogeochemical cycling. The end result of these anthropogenic modifications is the enhanced trapping and increase of residence time of most materials within the estuary, making it more sensitive to the effects of pollution, eutrophication, and salt intrusion. Droughts decrease the likelihood that the high discharge events needed to reduce estuarine entrapment will occur. However, at least along the South Carolina and Georgia coasts in systems of moderate to high tidal range, tidal flushing appears to play an important role in mitigating anthropogenic inputs (Dame et al. 2000).

E. Estuarine Fauna

Based on previous discussions relating to drought alteration of estuarine biochemical and physical processes, numerous paths for drought and freshwater inflow reductions to impact estuarine fauna exist. These vary in their level of directness. They include undesirable salinity change affecting cues for life cycle movements in, out, and through the estuary; reduction of dissolved oxygen; reductions in primary and secondary productivity limiting food availability; facilitating conditions for harmful or toxic blooms of cyanobacteria and dinoflagellates; and enhancing entrapment of contaminants in estuarine waters. Adequate freshwater delivery is vital to support the biodiversity, productivity, and fishery habitat of estuaries (Nixon and Buckley 2002).

Southeastern estuaries sustain an abundant diversity of recreationally and commercially important species. Commercial landings data support the idea that South Carolina and Georgia fisheries are similar in terms of total catch and proportion of shellfish and estuarine species. North Carolina's fisheries are dominated by finfish, have greater annual yields compared to South Carolina and Georgia, and are primarily based on estuarine species (Dame et al. 2000). In a review of flow regulation impacts, Drinkwater and Frank (1994) found changes in species composition, distribution, abundance, and health of fish and invertebrates attributable to changes in freshwater flow. They also linked changes in river flow to changes in migration patterns, spawning habitat, and fish recruitment.

Estuarine nekton use the shallow tidal creeks, marshes, mangrove swamps, and vascular plant grass beds (North Carolina) as nursery grounds and habitat (Hettler 1989; Gilmore 1995; McIvor and Rozas 1996). Different species cycle in and out of the estuary year round (temporal niche partitioning), effectively utilizing these critical nursery habitats. Both resident and transient nekton can potentially distribute production from these shallow habitats to the subtidal estuary and coastal ocean. According to a survey by Nelson et al. (1991), Penaeid shrimp and blue crabs dominate the invertebrate component of commercially and recreationally important estuarine nekton. The most abundant finfish (the bay anchovy, spot and striped mullet) occur across a wide range of salinities and dominate most southeastern estuaries.

Changes in salinity structure due to droughts can affect the distribution of motile organisms in the estuary. For the most part, estuarine organisms occur within focused salinity ranges, and different stages in their life histories have specific salinity requirements (Alber 2002). For example, post larval white (*Penaeus setiferus*) and pink (*Penaeus duorarum*) shrimp are most abundant in the polyhaline and mesohaline portions of the estuary. Post larval brown shrimp (*Penaeus aztecus*) utilize the mesohaline zone and juvenile brown and white shrimp are most abundant in the mesohaline and oligohaline zone (Wenner and Beatty 1993). In a study of early life stages of the blue crab (*Callinectes sapidus*) in Charleston, South Carolina estuaries, Mense and Wenner (1989) found that life history stages of both post larval and juveniles collected appear to be distributed on the basis of salinity. Childress (2010) found a strong correlation between the PDSI, raised salinity, and a decrease in the blue crab population. Bulger et al. (1993) found nonrandom discontinuities in fish distributions along the estuarine gradients in Chesapeake and Delaware Bays. Abundances of anadromous striped bass on the Roanoke River in North Carolina were correlated with inflow (Rulifson and Manooch 1990). These examples demonstrate the life cycle strategies of estuarine organisms in niche partitioning and the criticality of maintaining multiple haline zones in estuaries.

A 20-year record of blue crab catches in the Georgia fishery suggests that lower salinity in estuaries seems to be associated with blue crab population increases. Between 1970 and 1990, large catches were positively correlated with a high river flow, suggesting that the availability of fresh water was beneficial to the health of the crab population (Lee and Frischer 2004). Blue crab populations in South Carolina declined from 1995-2007 with annual landings down 30% below the 20-year average catch (Childress 2010). Work by Childress (2010) suggested that even a species like blue crabs with a wide salinity tolerance may be negatively impacted, and decreased by changes in annual freshwater discharge rates. Changes in salinity were negatively correlated with crab density. Decreased freshwater flow during prolonged periods of drought resulted in higher salinities that negatively impacted blue crab densities (Fig. 4).

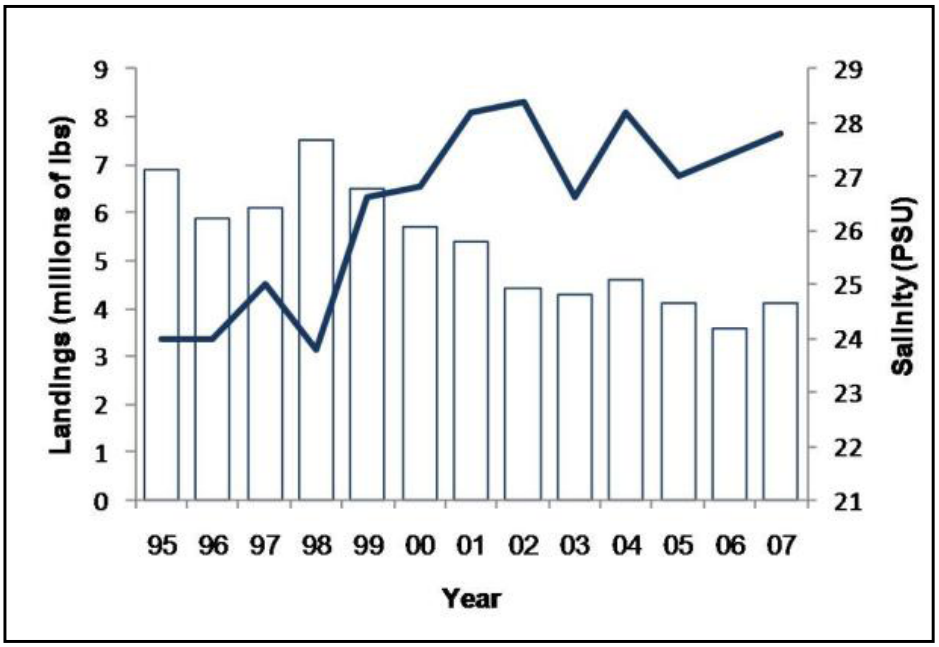

Figure 4. *Annual landings of South Carolina blue crabs (bars) and average annual salinity (line). Salinity is significantly and negatively correlated with blue crab landings (r^2 = 0.539, df = 12, p = 0.003). (from Childress 2010; reprinted with permission)*

Adult crab density decreased in higher salinities due to a combination of decreased growth and higher mortality in upper river regions. These habitat patches have lower food availability and higher densities of crabs, which led to increased mortality due to increased cannibalism. Childress' rationale for higher salinities impacting blue crab populations points out the complexity of the evolved life history strategies of estuarine organisms and the importance of salinity in these strategies.

> Blue crabs clearly use salinity cues to select appropriate habitat for settlement, mating, and larval release. First, higher salinity at the mouths of rivers means that post larval crabs may not perceive freshwater cues that would normally signal the saltmarsh nursery, thus, delaying settlement and increasing mortality. Second, males migrate upriver until they reach an optimal salinity of approximately 15. During drought years, these males must travel further upriver to reach their preferred salinity into areas with few saltmarsh prey species. Third, immature females must travel upriver until they reach males during the mating season. This extra distance travelled, coupled with lower food resources is likely to reduce the energy stores available for egg production. Finally, mated females must return to high salinity waters in order to release their larvae. If mature females reach the high salinity cue while still in the river they will potentially release their larvae too soon, leading to a reduced probability the larvae will reach the open ocean to complete the life cycle (Childress 2010, 3-4).

Many other important estuarine fauna share these complex life history cycles including Penaied shrimp. DeLancey et al. (2008) conducted a long-term trawl monitoring of white shrimp stocks within the ACE Basin National Estuarine Research Reserve in South Carolina. This work could serve as a good baseline for future similar (to Childress' blue crab work) drought effects analysis on white shrimp. They note that many previous authors have stated that populations of the species seem to be positively affected by relatively higher amounts of freshwater inflow (Gunter and Hildebrand 1954; Lam et al. 1989). DeLancey et al. (2008, 196) concluded with the remark that "…if the trend toward drier, more saline conditions persist, it may well impact the productivity of white shrimp stocks in the future and will warrant continued monitoring."

Changes in the timing of water delivery can also affect estuarine resources. The life histories of many fish and shellfish are cued to high spring runoff, such that changes in timing can affect spawning and nursery cycles (Alber 2002). In Sabine Lake, Texas, the presence of a dam shifted peak flows from spring to summer, reducing the availability of both low-salinity nursery habitat for brown shrimp in the spring and high-salinity nursery habitat for white shrimp in the summer (White and Perret 1974). In a study of the impact of salinity variability on estuarine organisms in Northeastern Florida Bay, Montague and Ley (1993) found a negative correlation between the standard deviation of salinity and the density of plants and benthic animals and suggested that frequent salinity fluctuations result in increased physiological stress.

Drought-induced changes toward higher salinities can also facilitate introduction of harmful marine parasites. Such parasites include the proliferation of a marine parasitic dinoflagellate *(Hematodinium perezi)* lethal to the blue crab and *Perkinsus marinus*, a parasitic protozoan infecting oysters (Power et al. 2006). Observations suggested that *Hematodinium* disease outbreaks were more likely when the flow of fresh water to estuarine systems was reduced and the salinity was high – greater than 28 (Lee and Frischer 2004).

Power et al. (2006) studied *Perkinsus* infection of oysters in coastal Georgia following a prolonged drought. Unfortunately, all sampling was conducted during a post-drought period and was unable to offer direct cause and effect evidence. However, the work could provide a baseline facilitating future study regarding *Perkinsus* in southeastern estuaries during a drought.

Prolonged drought can have long-term modification effects on estuarine trophic system dynamics and the dependent fish community. Livingston (1997) conducted a 13-year study on the effects of reduced river flow on nekton and trophic dynamics in the East Bay estuary portion of Apalachicola Bay in Florida. Prolonged drought led to reduced fish species richness and trophic diversity. Habitat stress was related to enhanced instability of the biological components of the estuary as a function of changes in nutrient cycling. The normally turbid estuary cleared with resultant changes in primary production and associated changes in system trophic response. The food web was simplified while overall fish biomass and individual species populations were numerically reduced. The trophic response times of fish assemblages were measured in years, beginning with the initiation of the drought. Such responses cannot be defined without truly long-term analyses. Recovery with resumption of increased river flows was likewise a multi-year event.

Variations in freshwater flow over drought/non-drought periods can affect the presence and/or severity of estuarine fish kills by influencing phytoplankton production, salinity stratification, and bottom-water hypoxia (Wetz et al. 2011). In the Neuse River Estuary (NRE), North Carolina, extensive bottom-water hypoxia is one of the main correlates of fish kills in the system. During both the 2002 and 2007 droughts, reduced freshwater inflow led to lower phytoplankton productivity and a slight reduction in salinity stratification, which increased bottom dissolved oxygen concentrations, and fish kills were greatly reduced relative to the long-term average (Wetz et al. 2011). However, they include the caveat that "it is difficult to establish causality due to the complexity of factors controlling NRE fish kills and overall fishery response to hypoxia (Wetz et al. 2011, 235)." The effects of the 2007-2008 drought lasted well into 2009. Although the downstream region of the NRE remained mostly devoid of blooms into 2009, relatively large phytoplankton blooms were observed in the mid to upper estuary during summer 2008 and winter-spring of 2009 as normal river flow resumed. This, along with intensifying stratification due to freshwater inflow, inevitably led to development of hypoxic conditions in this region, and significant fish kills were observed.

During their study, Wetz et al. (2011) found mesozooplankton abundance was 8–13-fold higher in summer 2007 prior to the peak of the drought as compared to summer 2008 or 2009, indicating that the downstream reduction in primary production may have had a multiyear negative effect on these important intermediaries between phytoplankton and fish. Although there were relatively minor fish kills during both 2002 and 2007, significant fish kills were noted during summer 2008, suggesting that lack of food availability (i.e., zooplankton) coupled with seasonal hypoxia were major causative factors. Since the estuarine-dwelling phase of many commercial fish species may be critically dependent on one particular season (Able and Fahay 1998), a single drought could have prolonged negative consequences on estuarine-dependent fish species if larval or juvenile growth is hampered (Baptista et al. 2010; Martino and Houde 2010).

VI. Tidal Freshwater

Tidal freshwater systems occupy the upper ends of estuaries and are composed of tidal marshes, tidal swamps, and associated riverine conveyances (rivers, streams, and headwaters). Tidal freshwater wetlands are found worldwide at the outlets of coastal rivers with low gradient and low topographic relief at or near sea level (Conner et al. 2007a). In the United States, they commonly occur in the lower Coastal Plain along the Atlantic coast and Gulf of Mexico coast stretching from Maryland to Texas (Odum and Hoover 1998; Mitsch and Gosselink 1993). These wetlands receive sufficient freshwater flows to generally keep surface water salinities less than 0.5 (Cowardin et al. 1979). Tidal freshwater systems typically share unique characteristics and biota of both riverine and estuarine systems and act as a transition habitat, occupying a substantial area of the coastal zone (NWRC 2011) (see Fig. 5). Few ecosystems exist in a more vulnerable location for shifts in salinity and flood regime than tidal freshwater wetlands (Doyle et al. 2007b).

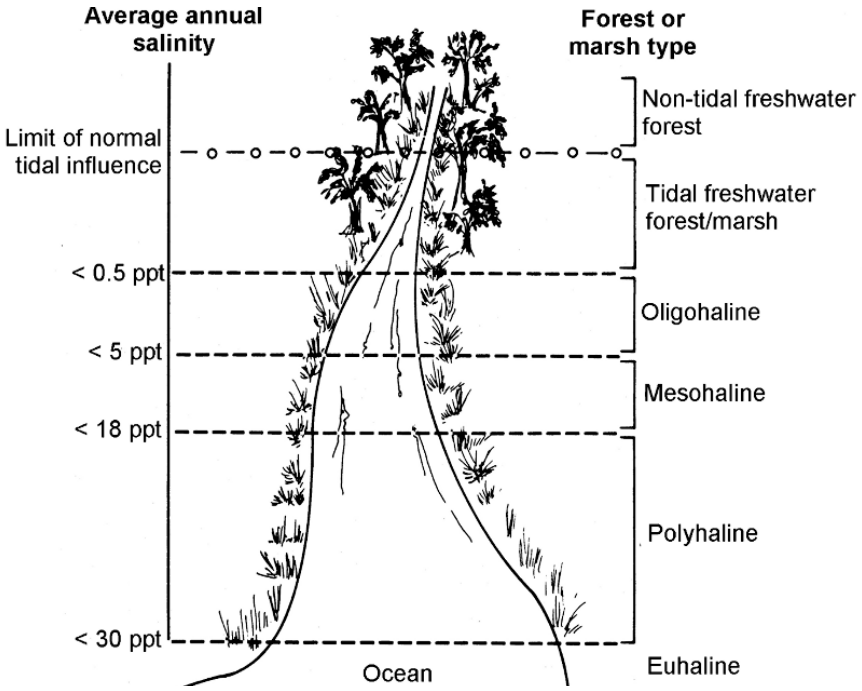

Figure 5. *The location of tidal freshwater wetlands along an idealized riverine landscape (Odum et al. 1984).*

Tidal freshwater marsh and forested swamp systems are maintained by similar hydrological conditions. Freshwater input (rainfall, river flow, and ground water) and the physical forcing of the tide act together to control water levels in them. River discharge (in the absence of upstream water control) depends on rainfall, evapotranspiration over the entire drainage basin, and antecedent groundwater conditions (Day et al. 2007). Poff et al. (1997) identified five critical components of river flow regime that regulate ecological processes in river ecosystems: the magnitude, frequency, duration, timing, and rate of change of hydrologic conditions. The interaction of these five river flow components with coastal tides regulates ecological processes in tidal freshwater wetlands. The spatial distribution of freshwater marshes or forests in riverine tidal settings depends on the interaction of river discharge rate with tide range. Daily tide range decreases as river discharge increases. At high discharge, the force of water flowing downstream impedes the progression of the tidal wave upstream (Day et al. 2007).

By definition, tidal freshwater systems have a mean annual salinity of less than 0.5. However, occasionally, they may experience higher salinities during droughts and periods of low freshwater flow (Odum and Hoover 1998). Hackney et al. (2007), through their work on the Cape Fear River, North Carolina, indicated that differences in community type, swamp versus marsh, were clearly related to the penetration of saline water into swamps. However, historically, many existing tidal freshwater marshes in the Carolinas and Georgia were created as remnants of rice culture clearing of coastal swamps in the 1700s where forest cover is either actively excluded or has not yet regenerated. Conner et al. (2007a) indicated that tidal freshwater marshes tend to be lower in elevation (inundated with greater frequency, depth, and duration) than tidal freshwater forests. Herbaceous vegetation, because of its ability to change with the season and from year to year depending on the degree of flooding and salinity of the floodwater, is less sensitive to these stressors than tree species that have to tolerate changing conditions to survive (Day et al. 2007). This is largely due to the salinity tolerance diversity and shorter lifespans in the herbaceous vs. forested communities.

A. Tidal Freshwater Marsh

Odum et al. (1984) described the tidal freshwater marsh system of the southeast in detail. The physical conditions necessary for tidal freshwater marsh development (and maintenance) include adequate rainfall or river flow to maintain fresh conditions, a flat gradient from the ocean inland, and a significant tide range (Mitsch and Gosselink 1993). This environment is more strongly influenced by the effects of inflowing riverine freshwater than the lower part of the estuary, which, in most cases, is strongly influenced by oceanic tides. With the absence of the salt stressor, tidal freshwater marshes are significantly more floristically diverse than are downstream brackish or salt marshes. These systems are capable of trapping and processing suspended sediments from upstream sources, thereby serving to reduce downstream loadings and improve the quality of water bound for the estuary (Odum et al. 1984). However, the extent to which they do this is dependent upon the age and ecological maturity of the marsh, the magnitude of upland runoff, anthropogenic effects such as sewage loading, and magnitude of tidal action (Stevenson et al. 1976). These marshes provide habitat for numerous species including a diverse assemblage of wading birds, feeding habitat for freshwater fish, and nursery habitat for estuarine organisms and juvenile anadromous species. Of all wetland habitats, coastal freshwater marshes may support the largest and most diverse populations of birds (Mitsch and Gosselink 1993). Odum et al. (1984) compiled a list of 280 species of birds that have been reported from tidal freshwater marshes.

Baldwin et al. (2001) investigated the effects of hydrology on tidal freshwater marsh vegetation along the Patuxent River in Maryland through a series of field and greenhouse studies. Taken together, the results of the studies indicated that 3–10 cm of flooding can significantly reduce seedling recruitment and growth in many plant species of tidal freshwater marshes and result in lower plant diversity. The greenhouse study further indicated that shallow flooding early in the growing season can reduce the abundance of certain species, primarily annuals, for the remainder of the growing season, resulting in a less diverse community. These findings suggested that hydrology is a dominant environmental variable controlling interannual variation in plant species composition of tidal freshwater marshes. Additionally, this study suggested that small increases in frequency and duration of inundation, which might occur due to watershed land-use changes, sea-level rise, or land subsidence, will reduce the diversity of these plant communities.

Similar to salt marsh systems described earlier in the report, tidal freshwater marshes depend on building soil volume to remain in equilibrium with sea level as these systems are sensitive to the accompanying saltwater intrusion. Low river discharges can also play a key role in elevating salinities in these systems and may act synergistically with rising sea level, leading to rapid loss and habitat conversion of tidal freshwater marshes (Neubauer and Craft 2009).

Neubauer (2011) performed controlled in situ experiments in tidal freshwater marshes of a tributary of the Waccamaw River, South Carolina. By physically adding diluted creek saltwater and freshwater to different treatment plots, he separated the effects of increased salinity from increased flooding. The creek water treatment plots had salinities representative of oligohaline conditions (salinity >5) and porewater salinities ranging from 1.6-3.3. Resultant salinity stress on the system significantly decreased production and species composition (species richness was approximately twice as high in control and freshwater treated plots).

In comparison to other areas, relatively little literature exists on the impact of drought on tidal freshwater marshes in the Carolinas. The bulk of references to hydrological and salinity impacts (which drought may influence) to tidal freshwater systems is contained in Conner et al. (2007a) which focuses on tidal forested wetlands. Since these two communities (tidal marshes and forests) are shaped and driven by the same hydrological forces, the recent work on tidal freshwater forest systems (see next section, VI. B) should apply to tidal freshwater marshes. The biggest noted concern is the loss or conversion of these communities due to salt intrusion. Drought can only act to exacerbate this concern.

B. Tidal Freshwater Forested Wetlands

Tidal freshwater forested wetlands are among the most sensitive ecosystems to sea-level rise and increased drought or changing flood frequency (Conner et al. 2007a). These systems are readily impacted by acute and chronic exposure to even low levels of salinity. The combined stress of flooding and salinity may compound the threat in these systems such that the margin for survival and compensation to changing hydrology and salinity is much less than for other coastal habitats. These systems are very responsive to subtle changes in climatic extremes (e.g., drought, storm surge, chronic salinization) that may influence site condition (Krauss et al. 2009). As mentioned above for tidal freshwater marshes, these wetlands can be periodically exposed to salinity either by large tides, droughts, or storm events, but their general function and species assemblage are those of a freshwater ecosystem. The hydroperiods of tidal freshwater forests range from rarely flooded to permanently flooded, which influence forest tree species composition and growth (Doyle et al. 2007a).

Seasonal patterns of tide and river stage in coastal rivers of the southeast play a key role in maintenance of these systems. According to Doyle et al. (2007a), these patterns generally reflect low riverflow and above normal tides in the summer and fall, and higher riverflow and lower tides in winter and spring. This seasonal effect of inversely correlated streamflow and tide stage provides more freshwater flow when trees are mostly dormant and greater saltwater exchange during the active growing season. The seasonally driven effects of salinity on tree growth may play a greater role in controlling the growth and succession of tidal freshwater forests than what has been documented to date.

As mentioned previously, the greatest threat to these systems appears to be salinity and flooding. It is at the lower edge of tidal freshwater systems that salinity becomes a controlling, sometimes chronic, factor (Day et al. 2007). Relative sea-level rise of any degree, large or small, in the absence of offsetting accretion or salinity diluting high river flows, will impact tidal freshwater swamps by increasing submergence and salinity over time. As sea level advances upslope, tidal freshwater swamps at lower elevations near the estuary interface will experience salinity conditions that will compromise growth and survival and initiate forest dieback (Doyle et al. 2007a). Chronic exposure to inundation by water with salinity of approximately 2 led to conversion of tidal forested wetland to oligohaline or brackish marsh on the Cape Fear River (Hackney et al. 2007). This observation was reinforced by work in Florida hydric hammocks where Williams et al. (2003) documented the loss of a dominant species, southern red cedar (*Juniperus virginiana*) through the interaction of sea level rise and drought, not drought alone.

The tidal portion of the hydrology differentiates these communities from other forested riverine wetlands. On the Cape Fear River in North Carolina, Hackney et al. (2007) reported that soils are nearly always saturated (at or near the surface) in tidal freshwater swamps distinguishing them from seasonally or episodically flooded bottomland hardwood swamps located further upstream or at higher elevations. This occurs because the time between each tidal inundation is generally short. Duberstein and Kitchens (2007) found a similar situation in Savannah River tidal freshwater forests where hydrologic conditions resulting from the range and consistency of the semidiurnal tides kept soils saturated for the entire year in most areas of the tidal forest, even during the extended drought conditions of 1999-2002. This reinforces salinity and not drying as the primary stressor during drought and creates a very different environment for both plants and animals compared to typical riverine-wetlands, where soils dry to some depth for extended periods, especially during the growing season (Hackney et al. 2007).

Microtopography (a series of hummocks and hollows) affects ecosystem function in these communities by affecting oxygen penetration, nutrient availability, rates of decomposition, and herbaceous plant species distributions (Courtwright and Findlay 2011). Microtopography plays a key role in vegetative distribution in these systems with the hummocks (approximately 15 cm in elevation above the hollows) supporting woody forest vegetation, and the hollows (saturated) being bare mud or herbaceous vegetation similar to that found in tidal freshwater marshes (Hackney et al. 2007). The hollows function as the upper extreme of tidal creeks, providing the benefits of regular flooding and draining to the tidal freshwater ecosystem without the erosive force of stronger tidal forces downstream. The hummock tops provide a well-watered/well-drained environment with more aeration than forested wetlands with continuously saturated soils. Duberstein and Kitchens (2007) found that freshwater flushing was a key factor in the different communities of two comparative study areas in tidal freshwater swamps along the Savannah River. As opposed to a tree-dominated forest, the shrub-dominated area was located further from a tidal source and as a result had reduced microtopography and two-fold higher values for soil electrical conductivity, Sodium, Chlorine, and Calcium.

When river discharge in the Cape Fear River is low and/or astronomical or wind forces cause water levels to lower, the forest floor dries for an extended period (Hackney et al. 2007). The distribution of surface water salinity between sites and across specific sites is mediated by the effects of localized rainfall and upland runoff, river discharge, and regional climatology (i.e., drought or flood). During drought conditions, the tide influences the groundwater level, which is never far

below the surface. During the summer drought of 2001, saline water penetrated far upriver and salinities at an upstream station were near 1, roughly twice the values typical of this site. In the summer of 2002, sites with salinities typically in the 0.2-0.5 range experienced an order of magnitude increase in salinity.

Doyle et al. (2007a) investigated the effects of tidal flooding, river source, and salinity on the long-term growth response of baldcypress (*Taxodium distichum*) trees and forests in the tidal freshwater zone of the Pee Dee and Waccamaw Rivers near Georgetown, South Carolina. They reported that the timing and duration of drought or flood events is important to the level of impact. During extended droughts, riverflows can remain near base flow, allowing greater tidal pulsing and salinity intrusions further upstream than normal. During extended wet seasons and with active tropical storm seasons, riverflows can remain high most of the growing season and abate to some degree salinity incursions and even refresh zones that experience saltwater intrusion during drought periods.

These systems appear to be able to resist conversion to tidal marsh if flooding by saline water is infrequent. However, even low levels of saline water, less than 10% seawater, led to major community change in tidal freshwater swamps of the Cape Fear River if such conditions were chronic (Hackney et al. 2007). While individual or sporadic intrusions of saline water did not immediately lead to a change in vegetation (i.e., tidal swamp to tidal marsh), chronic exposure did. Marsh sub-sites in the study area (hollows) were flooded by water with a salinity of <1 almost every tide, whereas stations dominated by large trees (hummocks) were exposed to low concentration saline water (salinity <1) only during periods of drought. The concentration of saline water necessary to begin the process of community change seems to be approximately 2 based on a long-term data set (Hackney et al. 2007). Growth of baldcypress in a four-year study of tidal swamp transects in South Carolina (Waccamaw River), Georgia (Savannah River), and Louisiana was reduced considerably at mean annual salinities above 2 (Krauss et al. 2009). The Lower Waccamaw (salinity 2.1) and Lower Savannah (salinity 3.4) sites were both actively converting to marsh at the time of the study.

Periodic pulses of floodwater containing low levels of seawater can dramatically affect soil biogeochemistry through the generation of hydrogen sulfide (Hackney et al. 2007). Tidal freshwater swamp soils typically generate methane as a byproduct of anaerobic mineralization (methanogenesis) but produce hydrogen sulfide as the dominant microbially mediated byproduct (via sulfate reduction) when saline water is introduced into soil pore space. Sulfate, the building block of sulfides (a toxic substance to many plants and animals), is abundant in seawater but relatively scarce in freshwater. This phenomenon was observed during periods of regional drought within the Cape Fear watershed. It is likely that both saline water and hydrogen sulfide (toxic to many plant and animal species) in the soils following prolonged or chronic exposure to saline water will result in a change of vegetation within tidal freshwater swamps (Hackney et al. 2007). The Cape Fear River data indicated that even short term pulses of salinity (such as those induced during a drought) can result in temporary plant community changes such as the shift from perennial to annual species. These changes, however, are short-lived and are typically followed by rapid post-drought recovery. Only when the frequency of saline water inundation becomes chronic do tidal freshwater swamps convert to brackish marsh.

Daily flooding and draining can theoretically increase productivity in tidal freshwater forests (Day et al. 2007). It has been shown experimentally that seedlings of baldcypress and black willow (*Salix nigra*) have increased growth rates with hydroperiods that simulate tidal frequency (Conner 1994; Day et al. 2006). However, this could only occur in an ideal situation where enough freshwater flow blocks the introduction of salt water through tidal action (Day et al. 2007). Besides salinity, soil total nitrogen, flood duration, and flood frequency affect forest structure and growth (Kraus et al. 2009). More frequent flooding (apparently in excess of the tidal flooding frequency in the baldcypress and black willow seedling experiments above) along strongly tidal rivers tended to decrease growth, while increased flood durations associated with longer-term ponding of non-tidal sites tended to increase growth. Productivity values of tidal freshwater forested wetlands on Bull's Island, South Carolina were at the lower end of values reported previously for forested wetlands in the southeastern United States (Ozalp et al. 2007). The main reason for the lower productivity of forests on Bull Island may have been due to drought conditions the study area experienced from late 1998 until mid-2002.

Doyle et al. (2007a) conducted studies on the growth of the dominant tree species, baldcypress, in the tidal freshwater wetlands around Sandy Island in the Waccamaw and Pee Dee Rivers, South Carolina. They found that drought years, in concert with low riverflow and high tides, increase saltwater exposure sufficiently to cause relatively poor growth performance compared to that of the trees and sites upstream. However, periods of high riverflow and rainfall may be adequate to flush or ameliorate saltwater concentrations and stimulate substantial rebound of growth potential. Tree growth during the non-drought 1970s (and concomitant higher-than-normal river flooding) exceeded all other decades and years at all other forest sites. Furthermore, Doyle et al. (2007a) reported that any extended periods of chronic or elevated exposure to saltwater intrusion without freshwater recharge will ultimately lead to tree death and forest dieback. In all cases, the presence of saltwater has been shown to compromise tree health and growth potential such

that exposed tidal sites and forests near the estuary interface are expected to grow more slowly than are tidal forests without salinity pulsing. Tree growth suppression during drought years and above normal tides may be an indicator of an eventual threat to tree health and potential for forest dieback from salinity intrusion especially in light of rising sea levels.

Conversion of swamps to marsh, due primarily to salinity incursion, can take over a decade with individual trees overcoming short salinity exposure episodes by adopting different rooting, water-use, or ion translocation strategies (Yanosky et al. 1995; Williams et al. 2003). Krauss and Duberstein (2010) confirmed some of these mechanisms in baldcypress at a salinity stressed transition area (active conversion from swamps to marsh) near Georgetown, South Carolina.

Vegetative components of the tidal freshwater wetland communities have varying tolerances to inundation (depth, frequency, and duration). A flood and salinity sensitive tolerance for species typically found in tidal freshwater forests was created by Conner et al. (2007c). Their study area included portions of both brown water rivers (Santee, Great Pee Dee, Savannah) and blackwater rivers (Ashepoo, Combahee, Cooper, and Waccamaw) in our geographical area of interest. Baldcypress exhibits the greatest degree of salt tolerance both as seedlings (Pezeshki 1990; Conner and Askew 1992; McLeod et al. 1996) and as mature canopy trees (Conner et al. 2007c). Hence, along with flood regime, climate-induced salinization has the effect of driving tidal and nontidal swamp stands to a baldcypress monoculture as a stage of degradation. In the Krauss et al. (2009) study, the distribution of water tupelo (*Nyssa aquatica*) in the overstory was limited to salinities below 1, while swamp tupelo (*Nyssa biflora*) occurred at salinities as high as 2.1 in the lower Waccamaw River site.

Herbaceous vegetation with shallow roots (e.g., pickerelweed [*Pontederia cordata*]) appeared to be a more sensitive indicator of salinity intrusion than woody vegetation (Hackney et al. 2007). This may be due to higher tolerance of woody species or their having deeper roots and growing on hummocks less exposed to saline inundation. In the upper Cape Fear River estuary, baldcypress has been shown to survive, albeit with lower growth (based on tree rings), even after soil salinity had increased above 5 (Harmon 1993; Yanosky et al. 1995).

Another important factor to be considered in these coastal forests is the recruitment of new individuals into the forest (Conner et al. 2007c). Buttonbush (*Cephalanthus occidentalis*), black willow (*Salix niger*), and cottonwood (*Populus deltoites*) can germinate in standing water, while baldcypress and water tupelo must have dry periods for the seed to germinate and establish (DeBell and Naylor 1972; Hook 1984; Kozlowski 1997). In many cases, adequate dry periods for cypress/water tupelo regeneration are not occurring (Conner et al. 1986; Keim et al. 2006), and if water levels continue to rise, coastal forested areas will eventually be replaced by scrub-shrub stands, marsh, or open water. Droughts may actually benefit baldcypress regeneration because seedlings usually cannot withstand more than a month of total submergence (Williston et al. 1980). Typically, previously logged baldcypress/water tupelo stands in non-tidal swamps areas will exhibit water tupelo dominance because of this baldcypress recruitment restriction and water tupelo stump sprouting (S. Gilbert, personal observation). Work on the Savannah River suggested that the lack of baldcypress as a codominant in the canopy of the tidal freshwater study areas supports the notion that baldcypress logging had occurred in the past (Duberstein and Kitchens 2007).

These systems, dependent upon the balance of freshwater discharge and tidal pulse to maintain hydrological conditions conducive to their survival, exist in a realm where large anthropogenic modifications greatly influence this balance. Every southeast system reported in the literature on this subject has experienced these modifications. For example, over the last 100 years the lower Cape Fear River was modified by the widening and deepening of channels and extensions to the Intracoastal Waterway (Hackney et al. 2007). In addition to sea-level rise, these activities have significantly altered the intrusion of both tides and saline water upstream to such an extent that areas once occupied by tidal freshwater swamps have been converted into brackish and salt marsh. Upstream wetlands that occupy floodplains and were once bottomland hardwood/softwood swamps, and non-tidal riverine wetlands are now tidal freshwater swamps. In the Cape Fear River estuary, historic dredging operations have increased the tidal range, converted freshwater swamp forest to estuarine salt marsh, and augmented sea-level rise.

Duberstein and Kitchens (2007, 326) reported similar cumulative impact factors resulting from anthropogenic changes in the Savannah River including hydropower stations, tide gate, and past and continuing navigation channel dredging and deepening. Key effects of these cumulative impacts include decreased sediment deposition at the harbor and measurable interstitial salinity increases (up to two-fold) in the freshwater marshes (Latham and Kitchens 1995; Dusek 2003). These modifications have also resulted in conversion of tidal freshwater marshes to brackish marshes (Pearlstine et al. 1993) and caused significant declines in the number of striped bass eggs found at historic spawning grounds (Van Den Avyle et al. 1990; Van Den Avyle and Maynard 1994).

Coastal ecosystems along the Gulf of Mexico and south Atlantic are currently undergoing forest dieback and decline from increasing tidal inundation, saltwater intrusion, and altered freshwater flow – changes attributed to global climate change, climate variability, and anthropogenic activities (DeLaune et al. 1987; Pezeshki et al. 1990; Allen 1992; Krauss et al. 2000; Chambers et al. 2005; Conner et al. 2007b). By virtue of their sensitivity to both flooding and salinity, tidal freshwater wetlands occupy a dynamic niche and are extremely vulnerable. Drought alone (especially if not chronic) may have only minor impacts on these systems. However, based on the cumulative impact of anthropogenic changes to the delivery of both fresh and saline waters, coupled with sea level rise and predicted changes in precipitation patterns and temperature rise (increased evapotranspiration), the outlook for these systems is somewhat bleak. As stated by Doyle et al. (2007a, 412) based on work in the relatively natural systems of the Waccamaw and Pee Dee Rivers: "The cumulative impact of a subsiding coast, rising sea level, recurring hurricanes, and reduced freshwater flow will result in forest dieback and coastal retreat as the intertidal zone migrates upslope."

Tidal freshwater swamp, Waccamaw River, South Carolina (William Conner).

C. Tidal Freshwater Streams and Rivers

Tidal conveyances (rivers, streams and associated headwaters) of the tidal freshwater zone experience the same primary drought stressors as covered above for wetland systems: inflow and salinity. A literature search for drought impacts related specifically to tidal freshwater riverine habitats revealed little information and consequently this section draws from other references.

The riverine system that supports tidal freshwater wetlands differs from non-tidal rivers and streams in fluvial (tidal pulsing), chemical (salinity intrusion and nutrients) and biological (faunal communities) contexts. Stranko et al. (2007) reported that although the fish assemblage of freshwater tidal streams is very different from non-tidal systems, species numbers appear to be related to stream size, with larger systems having more species. Exceptions to this relationship occur at sites with relatively high proportions of impervious land cover within 10 km upstream. Therefore, while discussion of the copious studies pertaining to drought impacts on non-tidal freshwater streams and rivers could be informative, differences related to tidal energy, adjacent tidal wetland organic export, the important salinity stress factor and the potential differences in faunal communities should not be minimized. Nevertheless, several defining works for non-tidal coastal freshwater streams and rivers relative to drought impacts exist. Lake (2003) compared impacts for periodic seasonal droughts vs. longer, unpredictable, aseasonal or supra-seasonal droughts. As with most systems described elsewhere in this report, it is the longer supra-seasonal droughts that have long-lasting adverse effects.

Griswold et al. (2008) sampled macroinvertebrates, flow, and water chemistry in small headwater streams from 2001 to 2007 in two adjacent coastal plain streams of contrasting headwater type (wetland and groundwater-fed seep) in southwestern Georgia following resumption of flow after the extreme 1998-2000 drought. Based on this work, they stated that unpredictable, extreme climatic events (e.g., droughts) can lead to loss of surface flow in many small streams potentially destabilizing aquatic communities.

Finally, Humphries and Baldwin (2003) summarized key messages from papers presented at the Symposium *Role of Drought in the Ecology of Aquatic Systems* held in Australia (2001). These papers provide a plethora of information on drought effects and recovery of these systems. Among other findings, they reported the following:

- Physical, morphological, physiological or behavioral refugia confer resistance or resilience to riverine populations and communities that experience drought conditions. The physical and chemical parameters associated with refugia habitats and their formation, influence population parameters within, and interactions among, species and can have protracted reproductive consequences, even well after the cessation of the drought.
- Fish, invertebrate and plant populations and assemblages seem to recover rapidly from drought. Most studies of the effects of drought, however, have arisen fortuitously and have involved relatively short temporal and small spatial, scales. Innovative approaches, such as microsatellite DNA analyses, can reveal that the effects of drought may be profound and long-lasting, resulting in population bottlenecks and altering the course of the evolution of species.
- During periods of drought, decreases in inputs of dissolved organic carbon, nitrogen and phosphorus may lead to carbon limitation to microbial metabolism, resulting in autotrophic production being favored over heterotrophic production.

D. Associated Headwaters

Headwater streams located immediately adjacent to the coast of the southeast typically have low discharge and are subjected to occasional drought conditions. The year-to-year intermittency of flow is unpredictable as is the length of the dry period (Smock et al. 1994). The different flow regimes can have significant effects on organic matter dynamics and macroinvertebrate communities. Chadwick and Huryn (2007) found that the timing and duration of stream flow of intermittent streams was more important than habitat in affecting macroinvertebrate production. These systems are already stressed by flashier hydroperiods from watershed development and its accompanying significant increase in impervious surface. Flashier hydroperiods result in fluvial impacts to stream channel stability. Further hydroregime changes may accompany precipitation pattern changes as a result of climate change. The effects of climate change precipitation pattern changes on ephemeral streams should be reduced hydroperiods and changes in hydroregimes, characterized by a more 'flashy' pattern of repeating cycles of inundation and drying (Brooks 2009).

Dai et al. (2010) ran a physically based distributed hydrological model, MIKE SHE, to evaluate the effects of altered temperature and precipitation regimes on the streamflow and water table in a headwater forested watershed of Huger Creek which drains into the East Branch of the Cooper River near Charleston, South Carolina. The model simulated an annual average streamflow increase or decrease by 2.4% with one percentage increase or decrease in precipitation. The model also demonstrated a relationship between changes in water table depth (cm) and precipitation. Also, the annual average water table depth and annual average streamflow linearly decreased with an increase in temperature within the range of temperature change scenarios (0-6°C) (modeled changes in evapotranspiration). However, the model did have problems in simulating flow in intermittent streams and so probably overestimated streamflow and other related parameters. They concluded that simulation results from the potential climate change scenarios indicate that future climate change (precipitation and temperature changes) will substantially impact the hydrological regime of upland and wetland forests on the coastal plain with corresponding implications to altered ecosystem functions that are dependent on water. They promoted MIKE SHE as a potentially valuable tool for such predictions but encourage further work to improve intermittent streamflow simulation. Similar findings were made in a MIKE SHE modeling study in a mixed pine-hardwood wetland in the Santee Experimental Forest (Lu et al. 2006).

Smock et al. (1994) studied a coastal headwater stream that drains into the North Inlet estuary, South Carolina. The year of the study was a drought year (1985-86) and the headwater experienced low flow and sometimes no flow during the study period. They reported that streambed drying probably affected storage of particulate organic matter by decreasing the amount processed during the dry months. The macroinvertebrate community was nearly completely eliminated by the drying. However, summertime drying is not untypical in this stream and the macroinvertebrate community may be adapted to intermittent flow and therefore somewhat resilient to drought. However, rates of recovery were difficult to determine without a longer-term study to determine what constitutes normal conditions. An interesting side note is that this study also included the effects of Hurricane Hugo in 1989 that scoured sediments out of the streambed. They concluded that the effects of the hurricane were much more severe than the drought.

VII. Functioning Coastal Impoundments

The literature search did not reveal work relative to the effects of drought on coastal impoundments. Impoundments that are essentially fully open to the estuarine system would experience impacts noted above for their respective estuarine salt, brackish, and tidal freshwater locations. Functioning coastal impoundments (i.e., those maintaining full water management capabilities) may be able to ameliorate some drought effects (especially those dealing with salinity) by management schemes that selectively preclude or limit exchange during high salinity events in the estuary. These actions may protect, reduce, or simply delay conversion of sensitive tidal freshwater and/or brackish systems to more saline communities. However, limiting exchange towards these ends has its own set of biological and ecological consequences and trade-offs, the nature and detail of which are beyond the scope of this report.

Managed impoundment along Cooper River, South Carolina (Dan Tufford).

VIII. Non-Alluvial Wetlands

In general, non-alluvial wetlands are defined as wetland communities with variable hydroperiods occurring in basins or depressions, or on slopes, with no connection to above-ground stream or river systems (Sutter and Kral 1994). The four major environmental variables that control the vegetation in these communities are hydroperiod, fire frequency, the presence of organic matter, and the source of water (Ewel 1990).

The vulnerability of wetlands to drought depends, in large part, on the sources of their water supply. The dominant sources of water to wetlands are precipitation, surface flows, and groundwater discharge (Brinson 1993). Generally, wetlands that are fed mainly by precipitation are the most vulnerable to losing wetland characteristics, while groundwater-driven sites that have large volumes of available water stored in aquifers will have the greatest resistance (Winter 2000). The responses of surface-flow wetlands will be somewhere between those types.

According to Poff et al. (2002), wetlands dominated by precipitation are already at the "dry end" of the spectrum of wetland types. They tend to occur in flat landscapes with low soil drainage in humid climates, such as parts of the eastern U.S. Depending on the frequency and duration of drought, long-term impacts may be that plant species less tolerant of flooding would dominate, and, over time, soils would dry to the point that they would no longer support the nutrient cycling processes unique to wetland soils. Peat-based wetlands would be especially hard-hit as the highly organic soils undergo oxidation and subsidence, thus altering drainage patterns, topography, and exposure to fire.

A. Carolina Bays and Pocosins

Shrub bogs, Carolina bays, and pocosins are restricted to the Atlantic Coastal Plain and abundant from southern Virginia to northern Florida, with the largest area of pocosins occurring in the outer coastal plain of North Carolina (Sharitz and Gibbons 1982). They are systems with a low nutrient status, in most instances have no permanent standing water and are strongly influenced by the hydrologic regime. Pocosin water storage is highly seasonal, and it is dominated by the balance between water inflows of precipitation and outflows by evapotranspiration, surface flow, and groundwater flow (Mitsch and Gosselink 1993). The hydrology of both systems is largely dependent upon direct precipitation, evapotranspiration and run-off to maintain water volume. Evaporative water loss can result in the complete drying of shallow bays.

Carolina bays and pocosins differ in geological formation and geomorphology. Carolina bays are characterized by unique elliptical depressions (shape, alignment, and surrounding rim) and do not have a single characteristic vegetation type (Sharitz and Gibbons 1982). In contrast, pocosins are freshwater wetlands with broadleaved evergreen shrubs or low trees including pond pine (*Pinus serotina*). They commonly grow on highly organic soils with poor drainage.

Some of the literature cited below was not clear in discerning whether the study locations were in the upper or lower coastal plain. However, since these ecosystems are essentially isolated from tidal waters even when near the coast, we include here research found within the northern and southern boundaries of the study area with the caveat mentioned earlier about different precipitation chemistry near the coast and the potential for impact by storm surge.

Lide et al. (1995) in a study of Thunder Bay (upper coastal plain of South Carolina) found that ponded water is a surface expression of the water table. Although fluctuation of pond stage is controlled by precipitation and evaporation, there are continuous seepage losses and periodic groundwater inflow. The hydrology of southern forested wetlands such as forested Carolina bays is extremely dynamic and is affected by the balance between precipitation and evapotranspiration. Any deviation of normal precipitation will have an impact on Carolina bay hydropatterns (Sun et al. 2006).

Many smaller Carolina bay depressions contain temporary aquatic habitats that may dry up seasonally or for longer intervals under local conditions of low precipitation or regional drought, while some of the larger and deeper ones may contain permanent water. The majority of Carolina bays are essentially intermittent lakes that contain water during certain seasons or wet periods and remain dry during other periods (Sharitz and Gibbons 1982). Reardon et al. (2009) reported that high levels of water storage occur in the winter and early spring when precipitation exceeds evapotranspiration. Lower levels are common in the late spring and summer when evapotranspiration is high and rainfall is less consistent.

A particularly notable aspect of pocosins is that the frequency and intensity of fire can influence vegetation. The highly organic and peat soils burn readily when dry and so the intensity of the fire is related to the depth of the water table (and hence influenced by droughts). Severe burns could result in a shift to a non-pocosin community or even create a lake in a former vegetated area. Many pocosin shrubs have evergreen sclerophyllous leaves (leathery leaves that resist moisture loss and fire), usually considered to be an adaptation to drought (Sharitz and Gibbons 1982). Reardon et al. (2009) studied factors influencing organic soil smoldering potential (fire danger) in North Carolina pocosin communities. Interestingly, they found that during the period of highest fire frequency in March and April, the water table levels were closest to the surface and soil moistures during that time were at levels with low estimated probabilities of sustained smoldering.

As is typical for coastal systems, anthropogenic disturbances compound any potential effects of drought on these systems. The widespread installation of drainage ditches for land management (e.g., agriculture, pine plantations, development) has altered the hydrology of these systems. Drainage ditches lower the water table and significantly affect surface runoff (Sharitz and Gibbons 1982).

Research on drought effects on Carolina bays focuses almost exclusively on vegetative changes. Mulhouse et al. (2005) found that wetland vegetation is strongly dependent upon climate-influenced hydrologic conditions, and plant composition responds in generally consistent ways to droughts. Their study showed that aquatic species decreased during the drought in all wetlands, regardless of vegetation group. However, the extent of species composition change during drought may be influenced by the pre-existing structure of wetland vegetation. They also found a difference in drought response by type of bay. Prior to drought conditions, all wetlands were ponded almost continuously, but during drought pond/meadow wetlands had shorter and more variable hydroperiods than the grass/sedge marshes. Consequently, vegetation change was greater in pond/meadow wetlands than in grass/sedge marshes. They concluded that vegetation change may be partly confounded with hydrologic conditions that provide greater opportunities for species recruitment in pond/meadow bays. The results suggested that Carolina bay vegetation dynamics may differ as a function of dominant vegetation and climate-driven variation in wetland hydrologic conditions. Drought facilitated greater dominance by species that required dryer conditions to establish.

Mulhouse et al. (2005) sampled four herbaceous depression meadow Carolina bay wetlands after a multi-year drought. They found that all bays lost previously abundant perennial aquatic species during the drought, and grasses, especially a rhizomatous perennial, *Panicum hemitomon*, expanded. The results of the study were consistent with a cyclic model of herbaceous Carolina bay vegetation dynamics in which aquatic and grass species alternately dominate as climate oscillates between wet and dry periods. Further, it appeared that in herbaceous Carolina bays, a handful of dominant aquatic and grass species in the vegetation may influence composition more strongly than widespread recruitment from the seed bank as hydrologic condition fluctuates. Stroh et al. (2008) stated that drought cycles may promote directional succession to forest in bays that are smaller, shallower, or colonized by flood tolerant hardwoods.

Finally, Kirkman and Sharitz (1994) manipulated the fire regime and soil disturbance in four grass-dominated Carolina bay wetlands during a prolonged drought period and examined vegetation community position and cover within dominant vegetation types prior to and after treatments. They found that species appear to persist with recurring and multiple disturbances because of seed banks, rhizomes, morphological plasticity during inundation, sexual reproduction following inundation, and perenniality coupled with early sexual maturity. They recommended that conservation management of intermittently inundated wetlands incorporate techniques to foster maintenance of endemic species richness and the recruitment of rare species. During periods of drought, it may be desirable to disturb the aggressive perennial vegetation to allow recruitment of less common species.

B. Flatwoods and Savannas

Sutter and Kral (1994) discussed several non-alluvial wetland types such as pond cypress savannas, longleaf pine savannas, and coastal plain small depression ponds. They estimated that more than one-third of the rare plant species in the southeast occur in such wetland communities. Sutter and Kral (1994) also noted several other recognized non-alluvial wetlands on flat coastal terraces including pine flatwoods and pond pine woodland. These communities were also recognized by Nelson (1986). Flatwoods and savannas have high fire frequencies and generally occur on sandy soils. They are distinguished primarily on structural grounds, flatwoods having denser canopies. The hydrology of most of these systems depends primarily on rainfall and hydrologic fluctuations and provides an unpredictable environment for a mix of species, all of which must endure periods of flooding and drought.

Although there is no literature on the effects of drought on these flatwood and savanna systems, certain findings may be extrapolated. For instance, Sutter and Kral (1994) reported that the hydrologic fluctuations experienced by these communities are important in eliminating water-intolerant upland species, that, if present, could dominate the community and eventually reduce species richness. Drought also plays a role in facilitating conditions suitable for these fire-adapted communities. Sutter and Kral (1994) speculated that pond cypress savannas are usually located in fire-adapted natural communities previously dominated by longleaf pine. Fire probably entered these communities from the uplands during periods of drought. Therefore, drought may play a further role in shifting community types.

Flatwoods depression wetland, Brunswick County, North Carolina (Dan Tufford).

C. Other Non-Alluvial Wetlands

Burkett et al. (2005), in their paper on ecosystem response to climate change case studies, presented some general information on response of wetlands to changes in hydrology. Although the case studies do not occur in our area of interest, the general responses that follow are informative. Wetland plant communities are very responsive to changes in surface-water levels. Water levels are, in turn, often responsive to changes in climate. In freshwater wetlands, water levels typically increase during cool periods and decrease during warm periods. The expected linear response of plant communities in the short-term is a reduction in emergent species in years when water levels are high and a resurgence of emergent species from the seedbank when water levels are lower and sediment is exposed. Over longer time periods ranging from decades to centuries to millennia, the expected linear response is deeper open water with few emergent species during cool and wet climate phases, and shallow marsh with shoreline and mudflat species during warm, dry phases. Prolonged warm, dry phases with greater primary production by emergent species may accelerate basin infilling in protected areas not subject to wave attack. Successional processes may accompany reduced depths of the basins, leading to further vegetation changes that include the invasion of shrubs and trees, although return of high water can reset the successional stage again.

Conner et al. (2011) conducted a ten-year study of three sites within a longleaf pine/swamp blackgum forest system on the southern end of the Waccamaw Neck area of Georgetown County, South Carolina. Each site represented a variation of typical hydrological condition ranging from “Dry” to “Intermediate” to “Wet.”[7] Aboveground net primary production (ANPP) of trees, diameter growth, and litterfall was monitored from 2000 to 2009.

The study site was under severe drought conditions from July 2001 through late summer 2002 and again in 2007. Diameter growth was negatively affected in all three sites, but with different patterns. Diameter growth in the Wet site was more sensitive to drought conditions in 2001–02 and 2007 than either Dry or Intermediate sites. While droughts did not seem to have a significant impact on litterfall in the Wet site, litterfall in the Intermediate site was more sensitive to the drought than either Dry or Wet sites. ANPP was significantly lower in both Intermediate and Wet sites in 2001. Highest ANPP occurred in the Intermediate and Wet sites in 2003 following a return to more normal water levels at the end of the drought. Maximum tree production occurred on the Wet site in 2003 and it exceeded total ANPP of any site in 2001. In the Dry site, ANPP remained relatively consistent throughout the study when compared to Wet and Intermediate sites. While litterfall estimates are well defined with 3–5 years of data, data collection is continuing in order to assess drought impacts on stem growth across the gradient. These impacts are still not clear with 10 years of data. While the authors cited literature describing mortality from severe drought of upland oaks (*Quercus sp.*) and hickories (*Carya sp.*) in inland states, they reported very little mortality observed in this study.

[7] “Dry” is indicative of sand ridges with a longleaf pine (*Pinus palustris*) association, “Intermediate” includes wet flats with laurel oak (*Quercus laurifolia*) and loblolly pine (*Pinus taeda*.), and “Wet” consists of swales/poorly defined creeks with a gum-cypress association (*Nyssa* spp.–*Taxodium* spp).

D. Non-Alluvial Wetland Fauna

Several literature citations exist on the impact of drought on wetland fauna including reptiles, and wading birds. Note that wading bird impacts presented here may apply to other freshwater wetlands.

Seigel et al. (1995) monitored aquatic snakes at a South Carolina wetland site during the extensive drought during the summers of 1985-1987, before, during and after the drought. Long-term changes in abundance and species composition were marked; the prey base was altered radically, overall abundance of snakes was lowered, and the Florida green water snake (*Nerodia floridana*) was not seen at the site for five years after the drought.

Willson et al. (2006) studied movement patterns and demography of seven species of semi-aquatic snakes at an isolated 10-hectare freshwater coastal plain Carolina bay wetland (Ellenton Bay, South Carolina) that experienced complete drying during the 2000 to 2003 drought. Since wetlands of this type experience drought often, inhabitants have developed strategies to escape or survive unfavorable conditions. For example, evolutionary adaptation and pressure allows aestivation during dry periods for black swamp snake (*Seminatrix pygaea*) (Winne et al. 2010). However, drought may render these wetlands temporarily unsuitable for many species, sometimes for years at a time. In the case of the reptiles in this study, different species adopted different drought-survival strategies and consequently were affected differentially by the extended drought. Cottonmouths (*Agkistrodon piscivorus*) migrated to and from the wetland annually, fared well, and reproduced during the drought. Banded watersnakes (*Nerodia fasciata*) suffered a dramatic population decline and apparently did not reproduce, while Florida green water snakes were locally extirpated. Black swamp snakes aestivated within the wetland and were less affected by the drought than green water snakes.

Gibbons et al. (1983) studied the effects of a major drought on aquatic turtle populations in a Carolina bay that had been under study for fifteen years. These long-term studies are critical to differentiate the effects of significant drought from natural variability. As was the case with snakes (above), five species responded differentially to the drying of a major aquatic habitat. The redear slider (*Pseudemys scripta*) and Florida cooter (*P. floridana*) emigrated in greater numbers, and fewer females laid eggs than in any previous year. The common musk turtle (*Sternotherus odoratus*) and chicken turtle (*Deirochelys reticularia*) did not reproduce at the level of previous years but did not abandon the aquatic habitat. Reproduction and emigration of the eastern mud turtle (*Kinosternon subrubrum*) were not appreciably different from other years. Again, these aquatic organisms seem to have adapted different strategies to cope with the stress of drought.

While addressing northeastern vernal pools that are ephemeral and hydrologically similar to some of our coastal southeastern forest pools, Brooks (2009) related that if average pool hydroperiods are shortened (as with drought conditions), fewer pools will provide breeding habitat to support local amphibian populations. The losses of potential breeding pools will increase inter-pool distances and affect larger amphibian metapopulations by negatively impacting inter-pool movements, recolonization, and genetic exchange (Gamble et al. 2007).

Certain wading birds are particularly subject to drought impacts. For example, white ibises (*Eudocimus albus*) breeding in salt marshes along the South Carolina coast appear to depend heavily upon crayfish from freshwater wetlands as a source of food for their nestlings. Bildstein et al. (1990) reported a significant decrease in numbers of nesting ibis and nesting success in drought year 1985. The two largest wading bird colonies along the South Carolina coast (at the time of the study) – Drum Island in the Cooper River and Pumpkinseed Island in Winyah Bay – declined from over 12,000 pairs at each site in a very wet year (1984) to less than 2000 pairs at each site in 1985, one of the driest springs on record. Similar dramatic declines in nesting success were also noted. Other more piscivorus wading birds breeding on Drum Island did not exhibit similar declines in numbers or reproductive success in 1985. Bildstein et al. (1990) suggested that declines in both the numbers of nesting ibises and in reproductive success at the two sites resulted from the relative unavailability of crayfishes during dry periods. Furthermore, as wading bird colonies are a key source of nutrient transfer from freshwater systems into estuaries, Bildstein et al. (1992) reported that the white ibis contribution to nutrient import to the North Inlet Estuary in South Carolina was eleven times greater during a wet year with normal reproductive success (1984) than during the subsequent dry year (1985) when there was a large decline in reproductive success.

Similar freshwater feeding bird problems have been reported in the literature. Mooij et al. (2002) reported that system-wide droughts in the Everglades can threaten the endangered snail kite (*Rostrhamus sociabilis*) population since they feed almost exclusively on the freshwater apple snail. Gaines et al. (2000) reported that breeding success for wood storks (*Mycteria Americana*) in coastal Georgia in drought year 1977 was less than half of what is was in wetter years. Differences in hydrologic condition and the resulting prey availability in coastal zone freshwater wetlands greatly affected foraging habitat use and breeding success of the three stork colonies studied. In 1977, foraging storks used estuarine foraging habitats much more intensively than they did in wetter years and their breeding success was less than half of wetter years. The authors concluded that freshwater wetlands appear to be very important to storks breeding along the Georgia coast. In drought years, estuarine wetlands by themselves may not be able to support the breeding population of storks in the region.

IX. Coastal Upland Forests

Hanson and Weltzin (2000) speculated on the sensitivity of forests to future drought disturbance from climate change. They stated that the primary response will be a reduction in net primary production and stand water use and mortality of small stature plants (i.e. seedlings and saplings) as a likely consequence of severe drought. Severe or prolonged drought may render even mature trees more susceptible to insects or disease. Drought-induced reductions in decomposition rates may cause a buildup of organic material on the forest floor, with ramifications for fire regimes and nutrient cycling. However, increased CO_2 may counter some drought effects with plant responses of deep rooting and substantial reserves of carbohydrates and nutrients that make mature trees less susceptible to water limitations caused by severe or prolonged drought.

They further provided information on the effects of water stress that has been cited as the ultimate constraint on the distribution and abundance of plants. Chronic and episodic droughts that affect soil water availability constrain seedling establishment, mediate plant-soil and plant-plant interactions, and limit forest productivity.

Noormets et al. (2010) studied carbon fluxes in response to drought in a coastal plain loblolly pine plantation located on the lower coastal plain of North Carolina. The site is classified as Mid-Atlantic flatwoods. The study was conducted over three years including a severe drought in 2007. They found that interannual differences in net ecosystem CO_2 exchange were caused by declines in canopy and whole tree hydraulic conductance that declined with growing precipitation deficit and decreasing soil volumetric water content. However, they also found no marked effect on gross ecosystem productivity and ecosystem respiration, both seemingly insensitive to drought. Net ecosystem carbon exchange was reduced with drought.

Longleaf pine (*Pinus palustris*) trees are found across the breadth of the Coastal Plain from Virginia to East Texas in a variety of environmental settings (Henderson and Grissino-Mayer 2009). These trees are typically long-lived (400–500 years maximum age). Henderson and Grissino-Mayer (2009) compared tree-ring chronologies to monthly temperature, precipitation, Palmer Drought Severity Index (PDSI), and Palmer Hydrological Drought Index (PHDI) data from the pertinent climate divisions at three sites. They found that PDSI and PHDI had the highest correlation with longleaf pine growth, and the strongest relationships between longleaf pine growth and these variables occured between July and November. Precipitation in the spring and summer was also positively related to growth at all sites. They expressed confidence in the Palmer Index correlations as they integrate the available water content of the soil, temperature, and precipitation, and therefore the composite nature of the variables more closely reflects the conditions required for tree growth than precipitation or temperature alone.

In a Louisiana study of fine root production and carbohydrate concentrations of mature longleaf pine, Sayer and Haywood (2006) found that a 1998 drought was associated with a three-month delay in peak root growth. During a prolonged drought in 1999-2000, there was reduction in root starch storage. They further reported that similar responses of tree root growth to water availability during drought have been observed but are variable in magnitude. For example, Jones et al. (2003) found that during drought, less longleaf pine fine root production occurred on drier compared to wetter microsites in a Georgia coastal plain forest. Joslin et al. (2000), however, reported only small and non-significant decreases in fine root production in response to water deficit in an upland *Quercus* forest. It is apparent that in addition to soil water availability, other microsite variables as well as their spatial and temporal variability can influence fine root production during drought (Jones et al. 2003; Joslin et al. 2000; Sword and Tiarks 2002).

In explaining the reduction of both the accumulation and maximum concentration of coarse root starch following the prolonged 1999-2000 drought, Sayer and Haywood (2006) cited several papers finding drought reduction of photosynthetic rate. Jose et al. (2003) found that the photosynthetic rate of longleaf pine seedlings is sensitive to water deficit, and other studies have shown that the photosynthetic rate of plantation loblolly pine is reduced by drought (Ellsworth 2000; Tang et al. 2004). Citing Kosola and Eissenstat (1994) and Caldwell et al. (1998), they further stated that tree roots can survive and grow in dry surface soil if deep soil water is available. However, by the end of the prolonged (three-year) drought, deep soil water may have approached depletion leading to deep fine root mortality.

While we failed to find much discussion in the literature pertaining to upland forest wildlife and drought, several papers from other areas discussed the adverse effects of drought (particularly summer droughts) on mast production (Sork et al. 1993; Johnson 1994). Acorns are a staple product of oak forests and are consumed by many species of birds and mammals including deer, bear, squirrels, mice, rabbits, foxes, raccoons, grackles, turkey, grouse, quail, blue jays, woodpeckers, and waterfowl.

X. Discussion

A. Key Findings

Short-term droughts are part of natural climate patterns and coastal systems have evolved mechanisms to tolerate such conditions. Indeed, some systems (e.g., fire-dependent, cypress dominated forested wetlands) rely on drought or extended dry periods for their sustainability. However, while numerous coastal systems are deficient in long-term investigations concerning drought impacts, the work reviewed for this report indicates that drought can have effects on most coastal systems. The severity of these effects depends upon the longevity and recurrence interval of drought event(s) and is compounded by other anthropogenic stressors on the system (see below for discussion of compounding stressors). Many of our aquatic and wetland coastal systems are particularly vulnerable to long-term and recurrent droughts as discussed further below. However, even upland forests show productivity and root system damage from droughts. Some potentially beneficial effects of drought found through literature review include temporary changes in nutrient input into aquatic systems that may reduce potential for algal blooms and increase water clarity thereby enhancing macrophyte productivity and reducing hypoxia. However, these effects may only be short-term and their beneficial nature is strongly site dependent.

Response to drought varies among ecological systems depending on the system of concern and how that system is adapted to existing seasonality of flows or less frequent, multi-year droughts. Therefore, defining drought for ecological systems in general is difficult. For the most part, the ecological literature does not discuss drought in terms of the drought classification system presented in the Introduction section (I.C) – "meteorological," "agricultural," "hydrological," or "socioeconomic" drought. Although most articles do not specifically mention the hydrological drought classification, this review found that hydrological drought is of the most interest and concern for ecosystem research in the project area. Drought is discussed primarily in terms of the hydrology-related impacts that affect coastal ecosystems, such as river discharge, freshwater inflows, water level, and water table depth. However, and as indicated in the previous sections, several studies do use the PDSI – a drought indicator used by the agriculture sector – to assess drought impacts on marsh and forest health. Finally, many of the studies do not discuss drought explicitly, but rather focus on the "ecological effects" or "causal links" associated with drought. The focus of such studies is to understand other environmental processes and stresses that may be related to drought, for example, how changing salinity or nutrient concentrations affect "attributes" such as water quality, reduced populations of estuarine organisms and shifts in aquatic communities, or how soil moisture levels and drying affects coastal habitats and biota. Figure 6 provides examples of indices or tools used to study the role of drought (timing, duration, severity) in causing or contributing to ecological changes or impacts.

Most of the research included in this report discuss and examine the impacts of multi-year droughts rather than seasonal droughts. The articles found for this report exemplify the wide range of processes and factors that contribute to drought and a diverse set of methods used to study such processes and factors. Time frames vary widely across studies, ranging from controlled experiments, to projects that investigate the impacts of short-term, intense events, to assessment of ecosystem change occurring during longer-term, multi-year droughts.

Estuaries, including their tidal freshwater component, are at the heart of coastal aquatic ecosystems and may well represent the coastal system most critically affected by drought. It is also the coastal system that was found to have the most literature regarding drought impacts. Wetz et al. (2011) expressed concerns that through their potential to dramatically reduce freshwater delivery to estuaries, droughts represent an important stressor of estuarine ecosystem functioning. However, there is a lack of sufficiently long time-series studies that leaves the scientific community with only a limited understanding of the full effects of drought on estuaries and the coastal zone. The effort to develop this State of Knowledge report verifies that there are a few, but fairly limited number of long-term or time-series studies from which to draw conclusive impact analysis.

Nevertheless, available literature to date suggests that droughts can exert both ephemeral and prolonged multiyear influences on estuarine ecosystem processes. Effects may include changes in nutrient loading, phytoplankton dynamics and primary productivity, resultant changes in secondary production, food web dynamics, water quality, and fish habitat. Studies document changes in the species composition, distribution, abundance, and health of fish and invertebrates attributable to changes in freshwater flow. Changes in river flow are also linked to changes in migration patterns, spawning habitat, and fish recruitment. Prolonged drought can have long-term modification effects on estuarine trophic system dynamics and the dependent fish community. As demonstrated by Wetz et al. (2011), the extent to which droughts influence nutrient loading, plankton productivity and biomass, water quality, and fishery habitat seems to depend on their timing and post-drought climate patterns. The eutrophic state of the estuary and other watershed characteristics play a large role in determining the nature and severity of impact. Other research draws a parallel between duration and intensity of the drought and the longevity of the impacts.

Reference	Indices or Tools Used
	Drought indices
Alber et al. (2008)	Utilized the monthly values of the PDSI for the climate division where each dieback event occurred in evaluating drought as a factor in salt marsh dieback.
Childress (2010)	Found a strong correlation between the PDSI, raised salinity, and decrease in the blue crab population.
Griswold et al. (2008)	Used the SPI to identify the frequency and duration of droughts in southwest Georgia in a study of macroinvertebrate characteristics and successional pattern.
Henderson and Grissino-Mayer (2009)	Compared tree-ring chronologies to monthly temperature, precipitation, PDSI, and Palmer Hydrological Drought Index (PHDI) data from the pertinent climate divisions at three sites to examine the relation between climatic conditions and longleaf pine growth.
	Hydrological data and models
Burkholder et al. (2006)	Used a network of long-term USGS monitoring stations, cross-estuary transect flow measurements, and mass water volume transport data to develop a model with which to study trends in nutrient concentrations and loadings in the Neuse Estuary (NC). The ten-year study period included a three-year severe drought.
Cahoon et al. (2011)	Measured marsh porewater levels to examine local groundwater conditions and perturbations for a study of Texas and Louisiana marshes during and after extreme drought.
Conrads et al. (2010)	Developed a hydrological model for the Savannah River that is based on 20 years of streamflow data and predicts salinity intrusion in the Savannah River with a 1-2 foot sea level rise and low flow.
Dai et al. (2010)	Used MIKE SHE to evaluate the effects of altered temperature and precipitation regimes on the streamflow and water table in a headwater forested watershed.
Lu et al. (2006)	Used MIKE SHE to study the hydrologic processes of a low relief coastal forested watershed and to test its sensitivity to potential climate variability and change.
Wetz et al. (2011)	Used monthly mean river flow (from USGS gauging station near Kinston, NC) to investigate the effects of drought on nutrients, water quality, plankton and biogeochemical measurements within the eutrophic Neuse River estuary (NC).
	Ecological indicators
Conner et al. (2007b)	Created a flood and salinity sensitivity tolerance index for species typically found in tidal freshwater forests. This system could facilitate predictive analysis of plant community composition shifts in light of changing drought conditions.
Maul et al. (1999)	Developed a forested wetland soil perturbation index as a reference standard primarily for indicating disturbance from timber harvest. It is a biogeochemical assessment using soil organic matter (SOM), total organic carbon (TOC), total Kjeldahl nitrogen (TKN), and total phosphorus (TP) as possible indicators of resilience.

Figure 6. *Examples of indices and tools referenced in this report.*

Although the literature is not conclusive about drought as the single cause of saltmarsh dieback in all localities, it has been strongly implicated in concert with other stressors. There is a clear relationship between decreased marsh productivity and high pore water salinity to which drought can contribute. Also, coupled with and compounded by rising sea level and anthropogenic water control, severe and repeated drought will likely result in upstream estuarine isohaline movement and concomitant estuarine community shifts. There are many documented cases in our study area where this has occurred (see Sections V.C.2 and V.F.2 of the report). Oligohaline and especially tidal freshwater communities are probably most at risk.

Tidal freshwater marshes and tidal freshwater forested wetlands are particularly vulnerable to the effects of droughts. Few ecosystems exist in a more vulnerable location for shifts in salinity and flood regime (Doyle et al. 2007b). Hydrology is a dominant environmental variable controlling interannual variation in plant species composition of tidal freshwater marshes and the combination of inflow and salinity is responsible for shifts from forest to marsh and from tidal freshwater to

oligohaline systems. Studies show that small increases in frequency and duration of inundation, which might occur due to watershed land-use changes, sea-level rise, or land subsidence, can reduce the plant diversity in tidal freshwater marshes. Numerous authors reported on studies showing the adverse effects of salinity intrusion into tidal freshwater forests. There appears to be a consensus that any extended periods of chronic or elevated exposure to saltwater intrusion without freshwater recharge will ultimately lead to tree death and forest dieback. In all cases, the presence of saltwater has been shown to compromise tree health and growth potential such that exposed tidal sites and forests near the estuary interface are expected to grow more slowly than are tidal forests without salinity pulsing. Tree growth suppression during drought years and above normal tides may be an indicator of an eventual threat to tree health and potential for forest dieback from salinity intrusion especially in light of rising sea levels.

Non-alluvial wetlands are highly threatened systems from various land use changes and drainage efforts. They contain and support many rare and endangered species. Lack of connection to surface water systems makes them critical to amphibian population growth and survival. Hydrology in these systems is largely driven by rainwater making them particularly vulnerable to extended droughts that are likely to result in vegetation community shifts and invasion by upland species. According to Poff et al. (1996), climate change impacts including altered precipitation patterns are projected to be greater on systems with rainfall-dominated hydrology, rather than those with groundwater dominated hydrology. Peat based systems may be highly at risk from deep fires exacerbated by prolonged drought.

Researchers repeatedly discuss the cumulative effects of drought in relation to other system stressors. The biggest "natural" stressor associated with drought in estuaries is salinity. Effects of drought, most often through the associated salinity stressor, can be compounded by anthropogenic alterations. These include construction of dams and associated water management activities, dredging and maintaining deeper (and deeper) navigation channels, development resulting in conversion of natural communities and proliferation of non-porous surfaces that increase the flashiness of urban runoff, ditching for agricultural and timber production that increases runoff of nutrients and eroded sediments, and conversion of coastal interior wetlands that decreases retention time of nutrients and other pollutants and reduces groundwater recharge opportunities. Of particular note, is that dams control the majority of our river systems above estuaries. During a drought, management of large reservoirs for water supply has the potential to compound instream flow and critical estuarine freshwater inflow problems. As per Dame et al. (2000), increasing human population densities in the coastal zone are placing ever-greater demands on the coastal aquifers, resulting in larger interaction between saline and freshwater aquifers. Additionally, dams have cut off access to and modified historic anadromous fish riverine spawning habitats (currently drowned by reservoirs). Such loss of spawning habitat (or access to it) is significant and has resulted in historic declines in anadromous species. Other anadromous species are floodplain spawners (e.g. blueback herring) that depend on seasonal connections between riverine systems and their floodplain wetlands. Springtime droughts would compound the problem for anadromous species by further reducing access to, and the amount of, suitable spawning habitat.

B. Related Research Themes

1. Drought and Climate Change

Another factor that has been alluded to, but not covered in detail in this report, is climate change. This phenomenon presents both direct and indirect associated impacts to estuarine and coastal systems.

In the Southeast, changes in annual and seasonal precipitation patterns are expected to affect the timing and amount of water availability and contribute to increased risks of flooding and drought (Karl et al. 2009; Scavia et al. 2002; Committee on Environment and Natural Resources 2008). Although estimates of the magnitude and timing of changes in precipitation patterns are variable, reflecting the high level of uncertainty associated with these predictions, changes in precipitation patterns, in conjunction with warmer water temperatures and sea level rise, could play a major role in the intensity of drought effects. Precipitation patterns are projected to be more episodic, with less frequent but more intense storms and longer inter-event droughts (Bates et al. 2008). Predicted warming air temperatures could exacerbate the effects of drought by increasing the rate of evapotranspiration. Warmer water temperatures could exacerbate the effects of low dissolved oxygen. Salinity has been mentioned numerous times in this report as a major stressor with direct ties to drought. Salinity changes are also perpetuated by relative sea level rise. The low elevations along the South Atlantic Coast make it particularly vulnerable. Drought may play a critical role by reducing freshwater inputs to estuaries that deliver large quantities of suspended sediments critical to the accretion of wetland soils. Such accretion is critical to offsetting rising sea level impacts on coastal wetlands.

Any potential precipitation changes accompanying climate change, such as decreased precipitation and hydrologic impacts, are pertinent to drought effects. Kennedy et al. (2002) stated that decreased precipitation and delivery of fresh water alters food webs in estuaries and affects the amount of time required to flush nutrients and contaminants from the system. Although reduced river flow would decrease nutrient input in estuaries with relatively uncontaminated watersheds, there could be different effects in polluted watersheds that contain point and non-point sources of nutrients and contaminants that are not a function of river flow. The combined effects of human development and reduced river flow would degrade water quality conditions, negatively affecting fisheries and human health through such changes as increased presence of harmful algal blooms and accumulation of contaminants in animals and plants

Conner et al. (2007c) demonstrated how tidal freshwater forested wetlands are affected by the extent and distribution of saltwater. Freshwater flows, sea level, and tidal ranges all affect salinity levels. These factors are expected to be affected by climate change and may increase forested wetlands exposure to saltwater in the future. Doyle et al. (2007a) suggested that it is not drought on its own that causes impacts, but interactions with other factors such as low riverflow and high tides which may increase saltwater intrusion/salinity.

The response of coastal and estuarine ecosystems to climate change impacts (including sea level rise, increased temperature, changes in rainfall distribution and freshwater inputs, and changes in the frequency and intensity of storm events) will be a product of the complex interactions between physical, chemical, biological, and human responses to climate-related changes and factors, occurring at different spatial and temporal scales (Day et al. 2008; Burkett et al. 2005). Day et al. (2008) identified four ecogeomorphological responses of coastal and estuarine wetlands to these climate-related changes: changes in elevation, boundary or edge distribution, areal extent, and composition of soil or sediment.

Ultimately, “Changes in freshwater, sediment, and nutrient delivery are likely to be highly regional and also are difficult to predict because they will be affected by changes in regional precipitation, changes in circulation and sediment trapping processes, shoreline erosion rates, and local and regional human activities” (Day et al. 2008, 478). Future inflows will affect the ability of wetlands/coastal ecosystems to respond to sea level rise and storm events (Scavia et al. 2002; also discussed in Nicholls et al. 2007; Fischlin et al. 2007). Other research suggested that a decrease in freshwater input to coastal ecosystems will adversely affect the ability of coastal wetlands to survive accelerated sea-level rise, while increased freshwater input is expected to increase the ability of wetlands to survive (DeLaune et al. 2003). In related work, Cahoon et al. (2006) investigated the possible relationships between sediment delivery and wetland elevation and the possible impacts of climate change (sea level rise, water fluxes) on coastal wetlands. Poff et al. (2002, 28) also noted “… decreased freshwater inputs are likely to lead to less accretion, lowered productivity, and saltwater intrusion.”

Burkett et al. (2005) discussed how ecosystem response may include linear as well as non-linear responses to environmental/climate changes. Coastal wetlands, for example, will be particularly threatened by sea-level rise (which is expected to increase flooding of coastal areas, alter tidal ranges, and increase average and peak salinities in estuaries) and by changes in the hydrologic cycle that are expected to affect the strength, timing, and volume of freshwater inflows, as well as chemical and sediment loads, to coastal ecosystems (Scavia et al. 2002; Burkett et al. 2005). Mendelssohn and Morris (2000) showed that coastal wetlands are more vulnerable to multiple stressors, rather than an individual stressor

acting alone. This is a key concern for coastal ecosystems that are already exposed to, or are projected to be exposed to, multiple stresses, not only climate change but population growth and development which increases demand for and stresses upon coastal and marine resources (Scavia et al. 2002; Fischlin et al. 2007).

Michener et al. (1997) examined interactions across multiple factors (rising sea level, altered storm patterns) and found that an increased frequency, intensity, and duration of hurricanes and tropical storms may (in terms of water balance) counteract water shortages created by drought. However, they also noted that it may be climate extremes and extreme events that will have the most significant impacts on ecosystems and species, rather than gradual changes in means or averages. Williams et al. (2003) examined the effects of multiple and interacting factors on a coastal forest, including sea level rise, storm activity, and drought. They found that the vulnerability of their study site to extreme events (storm events, drought) was increased due to damage caused by previous stresses (sea level rise interacting with drought which caused saltwater intrusion).

Climate change is also expected to contribute to changes in and impacts to water quality conditions. Anthony et al. (2009, 10) noted: "The expected increase in the variability and intensity of precipitation events is therefore expected to produce increased variability, both spatially and temporally, in salinity and dissolved oxygen concentrations in coastal lagoons." Increased surface water inputs may increase delivery of sediment and nutrients, with increased nutrient inputs thereby increasing the risk of eutrophication. Paerl et al. (2011) discussed how changes in regional precipitation patterns, and more specifically the intensity, frequency, and duration of extreme events such as flooding and droughts, may promote the prevalence of Cyanobacteria Harmful Algal Blooms (CyanoHABs). Increased salinity and changes in vertical stratification are important factors that may benefit some CyanoHABs; such conditions have been found to support the expansion of CyanoHABs in nutrient-enriched brackish waters (see Paerl and Fulton 2006). With more variable and intense precipitation events (floods and droughts) expected to affect freshwater discharges, nutrient loads, and salinity and stratification regimes, residence times will also be more variable. And, longer residence times are expected to aggravate CyanoHAB problems. In summary, increases in hydrologic variability and extreme conditions such as protracted droughts benefit CyanoHABs (Paerl et al. 2011), and variable weather patterns, such as storm events and droughts that may be part of climate change, are expected to alter water column conditions in a manner that stresses aquatic organisms (Deaton et al. 2010).

2. Drought Adaptation

Climate adaptation refers to actions or measures taken with the intent to improve the capacity of a human or natural system to cope with or respond to climate events, stressors, and related impacts. This report demonstrates how existing climate variability and drought negatively impacts coastal Carolina ecosystems. In addition, changes in annual and seasonal precipitation patterns are expected to affect the timing and amount of water availability and contribute to increased risks of flooding and drought in the Southeast United States. Given these considerations, study authors found documents that indicate an increasing interest in the identification and implementation of adaptation measures intended to reduce climate risks and impacts on coastal ecosystems. We found three types of documents that addressed this topic:

- peer-reviewed literature,
- general synthesis and guidance documents developed by federal agencies and conservation non-profit organizations, and
- region-specific assessments and action plans.

While the focus and scope of these documents varies, some key themes should be noted. First, the literature we found generally relates to topics such as rivers, ecosystem restoration, and climate change (and climate change adaptation). Drought is frequently discussed as a climate stressor but typically is not the primary or only focus of adaptation discussions.

Second, many of the adaptation measures identified are general in nature and are intended to promote ecosystem resilience and adaptability, although several authors do note that site-specific adaptations will depend on local and regional threats (Palmer et al. 2009; Watts et al. 2011; Peterson et al. 2008). Region-specific assessments and action plans relevant to this report include state plans (e.g. North Carolina Coastal Habitat Protection Plan, Deaton et al. 2010; Initial Recommended Regional Water Plan, Coastal Georgia Council 2011), estuary and watershed management plans (e.g. Albemarle Pamlico National Estuary Comprehensive Conservation and Management Plan, Draft 2011; Altamaha River Watershed Conservation Action Plan, TNC-GA 2005; Lower Roanoke River Project Conservation Action Plan, TNC-NC 2005), and multi-state or regional plans (Southeast Aquatic Habitat Plan, SARP 2008).

Third, it is recognized that decision makers and resource managers should consider both climate and non-climate stressors – and the interactions across stressors – when developing adaptation plans and measures for coastal ecosystems. The interaction of the two main stressors for drought impacts (altered precipitation patterns and water management) can act synergistically to create cumulative impacts or act "commensally" whereby water management can to some degree ameliorate the adverse effects of climate related droughts. The Southeast, for example, has in the last few decades begun to feel the competition for water use long experienced in the western United States. As climate driven droughts become more frequent and severe, there will be insufficient water to meet all demands.

Fourth, documents recommend a wide range of adaptations. Suggestions include improving the monitoring of hydrological and ecosystem processes, supporting research and improving understanding of climate impacts and natural system responses, implementing best practices for ecosystem and water management, and identifying where laws and policies can be modified (or introduced) to conserve and protect coastal ecosystems. Improving monitoring and data collection is viewed as critical for developing, implementing, and monitoring the progress of adaptation measures related to coastal ecosystems. Appropriate and adequate monitoring systems can help to establish a record of ecosystem conditions and trends and thereby a baseline from which to assess the effectiveness of management and adaptation measures (Southeast Aquatic Resources Partnership 2008; APNEP 2011; Peterson et al. 2008).

Adaptation recommendations with relevance to drought can be characterized into two key categories: 1) water planning and management and 2) land planning and management. Again, while drought is not the primary focus, understanding drought impacts on ecosystems and ecological processes is an important component of these strategies.

Water Planning and Management

Measures to protect water quantity and quality include establishing and maintaining flows that resemble natural flows as closely as possible and implementing and enforcing practices that maintain appropriate water quality.

Knowledge of adequate flows to maintain river and stream system ecological functions with minimal degradation is an essential step to fostering drought adaptation for ecosystems. Recent and ongoing studies in the Savannah River basin (see research needs above), provide an excellent example of analyzing instream flow needs for natural systems including maintenance of: water quality; fish habitat; migratory fish movements; floodplain wetlands hydrology; tidal freshwater communities; and estuarine salinity gradients and freshwater inflow needs.

On a regional scale, the Southeast Aquatic Resource Partnership[8] and the Southern Instream Flow Network (SIFN)[9] are working to accomplish these or closely related objectives. Among the objectives of the SIFN is integration and dissemination of flow-related work by agencies, universities, and nonprofit organizations (e.g. American Rivers, The Nature Conservancy).

[8] http://www.southeastaquatics.net/

[9] http://www.southeastaquatics.net/programs/sifn

Determination of seasonal instream flow ranges mimicking historic natural patterns that provide a scale of resultant natural resource condition will enable informed decision-making for basin-wide water allocation planning purposes. Such knowledge would aid Federal and other dam management during droughts. Section 7 (a) (2) of The Endangered Species Act requires all Federal agencies to protect listed species as part of any federal action including river management decisions during times of drought.

Basin-wide water resource planning is a key element in preparing for adaptation to greater drought conditions. Such planning should inventory and project water needs in the basin incorporating water needs to maintain natural ecosystem function and determining how a balance of water allocation can be met during drought periods. Other basin-wide plan drought adaptation measures would benefit ecosystem exposure to drought through reduction of other compounding stressors (e.g., excess nutrient loading) and/or conservation of water that would be more available for ecological instream flow needs. These include:

- Reducing nutrient loading to rivers, lakes and estuaries. Protecting healthy basin wetlands and restoring degraded wetlands to enhance nutrient intake will reduce nutrient loading (from Poff et al. 2002). Furthermore, basin scale reduction strategies for nitrogen and phosphorus should be planned and take into account hydrologic variability including droughts (as well as hurricanes and floods) (from Paerl et al. 2004). Protecting water quality is a primary objective of several regional plans (Southeast Aquatic Habitat Plan, SARP 2008; APNEP 2011; Altamaha River Watershed Conservation Action Plan, TNC-GA 2005; Lower Roanoke River Project Conservation Action Plan, TNC-NC 2005) and is an essential general strategy included in general guidance documents (e.g. U.S. EPA 2009; Peterson et al. 2008).
- Modifying dam operations to more closely resemble the natural hydrologic regime (TNC-GA 2005; TNC-NC 2005; Peterson et al. 2008; Palmer et al. 2009; Watts et al. 2011).
- Locating any new reservoirs only off-channel so as not to disrupt the natural downstream flow of water and sediments critical to riverine ecosystems. Passage of sediment to estuaries is also critical for the accretion of soil in coastal wetlands, which are threatened with inundation by rising sea levels (from Poff et al. 2002).
- Minimizing groundwater pumping for irrigation, human consumption, etc., that removes water from aquatic and wetland ecosystems (from Poff et al. 2002). More recognition of, and accounting for, the direct linkage of ground- and surface-water is needed when making water management decisions. Groundwater is often viewed as an alternative to surface water in meeting water demands when, in fact, they are both part of the same reservoir of fresh water. As land management activities alter the exchange of surface- and groundwater (either recharge or discharge), there are likely to be compounding medium- and long-term impacts on ecosystems and ecological processes at and near the coast (see also: Southeast Aquatic Habitat Plan, SARP 2008; TNC-GA 2005). Palmer et al. (2009) and Watts et al. (2011) discuss the need to integrate groundwater and surface water management and use.
- Developing standards for efficiency in water use and implementation strategies to ensure effective water use savings during droughts.

Land Planning and Management

Many documents with a land planning and management focus discuss ecosystem conservation and restoration measures in the context of climate adaptation. Such measures are intended to mitigate the impacts of climate variability and change (including impacts from drought and low inflows) and the adverse impacts that often result from interactions between climate and non-climate factors. Recommended actions include:

- Identifying and protecting climate refugia can help species cope with climate impacts (TNC-GA 2005; Gregg et al. 2011; Griffith et al. 2009; Mawdsley et al. 2009). Similar measures would expand the amount of habitat acquired and preserved, in order to help restore ecosystem functions, protect movement corridors, and enhance the resilience and adaptability of coastal ecosystems to future changes (Mawdsley et al. 2009; Scavia et al. 2002).
- Implementing land use practices (including the developing and maintenance of riparian buffers, implementing low impact development with more permeable surfaces) that can help to enhance water quality and reduce riverine hydrograph changes (Palmer et al. 2009; APNEP 2011; U.S. EPA 2009).
- Allowing for upslope migration of coastal marshes. This strategy is discussed in relation to sea level rise and climate change. However, significant decrease in freshwater inflows due to persistent or repetitive drought may exacerbate the physical impacts of sea level rise resulting in some of the same effects (e.g. Scavia et al. 2002; TNC-NOAA 2011).

XI. Literature Gaps and Research Needs

The most critical research needs include examining drought impacts in ecosystems not studied by existing research, implementing long-term studies to identify and examine causal linkages, and developing new tools and models – including ecological indicators – with which to monitor ecological change and impacts during drought.

A. Specific Ecosystems and Species

- There seems to be relatively little research on the following systems relative to drought impacts: coastal upland forests (particularly from an ecological impact perspective), freshwater stream tributaries of tidal rivers, pocosins and other non-riverine wetlands in the study area, savannas, maritime (barrier island) systems, nearshore ocean, intact coastal impoundments, and estuarine high marshes.
- There appears to be considerable study of tidal freshwater forested wetlands which may be the most sensitive coastal system to the effects of drought and increased salinity exposure. The same is not necessarily true for tidal freshwater marshes in our study area. While there is much literature found for this system in New England (although primarily lab studies) for example, local research is relatively minimal. In the inevitable advent of sea level rise coupled with drought scenarios, it would be beneficial to invest in research to determine the ability of these important and diverse systems to move upstream and therefore remain a component of our coastal ecosystem.
- Additional studies are needed to discern the importance of droughts during different seasons. While numerous studies mention the importance of seasonal flows to support aquatic organism needs in estuaries, more work is needed, particularly in tidal freshwater systems. Seasonal droughts could affect the ability of anadromous species to cue on migratory movements, as well as preclude use of critical floodplain spawning habitat for species like blueback herring (*Alosa aestivalis*).
- The results of ongoing or new studies on coastal impoundment efforts at controlling salinity in tidal freshwater communities (if these exist) would be very informative. Such studies should focus on the long-term ability to provide a controlled salinity regime while minimizing the impacts of reduced connections and exchange with the estuary. Many impoundments, particularly those in National Wildlife Refuges, are managed for particular biotic communities with specific salinity tolerances. Maps that provide Refuge managers with horizontal and vertical detail about projected changes in salinity are needed to help them plan relocation or other adaptive measures. This also necessitates a habitat modeling capability as an essential planning tool.
- Additional work is needed on the effects of drought on seagrass beds. These areas are critical parts of estuarine habitat in parts of North Carolina, yet little work on drought impacts to this habitat was found in the literature. The key species of concern is *Zostera marina* (at the southern limit of its range in North Carolina), which is reported to be euryhaline, tolerating salinities from 5 to full seawater. Therefore, increased salinity should not be a problem. If droughts, as reported in the literature, result in increased light penetration in estuaries, it is possible they may have a positive effect on *Zostera* productivity. Research is also needed to better understand how drought conditions may adversely affect submerged aquatic vegetation (SAV) habitat by creating conditions that benefit nuisance species such as macroalgae or bryozoans that have been known to overwhelm SAV in high salinity waters.
- Studies are needed to address how drought stress may impact organism disease vectors. For example, Power et al. (2006) studied *Perkinsus* infection of oysters in coastal Georgia following a prolonged drought. Unfortunately, all sampling was conducted during a post-drought period and was unable to offer direct cause and effect evidence. However, the work could provide a baseline facilitating future study regarding *Perkinsus* in southeastern estuaries during a drought. Some *Vibrio* species are also of concern in this context. These human pathogens appear to have a preference for higher salinities. So they may migrate further inland in a low freshwater inflow and higher sea level environment (Wetz et al. 2008).
- Work is also needed on the effects of drought on invasive floral and faunal species in all coastal ecosystems of the study area.

B. Causal linkages and long-term studies

- The causal linkages between drivers, stressors, ecological effects, and attributes in the conceptual ecological model presented earlier in this report are ideal subjects for further investigation. While many of these have been covered in this report, long-term studies controlling for outside stressors to scientifically define cause-effect relationships in these linkages would be valuable in impact analysis and remediation. A few causal links remain poorly addressed in the literature we uncovered, including the relationships between 1) "altered upland and non-alluvial and wetland community structure and functions" and "reduced wildlife populations" and 2) "reduced floodplain connectivity" and "reduced floodplain dependent fish and wildlife" (see Fig. 2, Conceptual Ecological Model for Drought Impacts). Understanding of linkages could be strengthened through studies that compare, for example, year class strength of floodplain spawners [e.g., anadromous blueback herring (*Alosa aestevalis*)] after critical drought and non-drought Spring spawning seasons.
- More research and information is needed on the groundwater resource in the study area and resulting drought impacts on groundwater dependent ecosystems and baseflow in streams. Also, further research is needed to understand the direct linkage of ground-and surface-water as it relates to water management and ecosystem requirements.
- Baldwin (2007, 149) suggested that "…it is possible that short-term salinity pulses during drought may be more important in structuring swamps than the average, low salinity level they experience over the long term." Research is needed to better *understand potential responses* to salinity pulsing, given the expected changes in SLR, hydrologic changes, and more frequent and severe saltwater intrusion events due to climate change.
- Long-term or time series studies capturing multi-year and repeated drought events are needed to understand drought impacts, especially for ecosystem processes. The significant supra seasonal and long-term events are often unpredictable and variable, thus more difficult for organisms to cope with through evolved adaptations. At the levels of populations and communities, short-term studies many enable some generalizations about the biotic responses to the disturbance of drought, but more long-term studies about ecosystem processes need to be conducted (Lake 2003). It would be beneficial to take advantage of available sites with already established long-term data sets for different coastal ecosystem types. Long Term Ecological Research sites, National Estuarine Research Reserves, National Estuary Program Sites, State and Federal Refuges, etc. would be ideal places for such research. An intensive and continuous monitoring program that builds on these resources would require an organized and coordinated, multi-state, multi-institutional approach. Such a program has existed for Chesapeake Bay and its tributaries for decades; however, nothing of this scale exists for either the Albermarle-Pamlico Sound or Winyah Bay systems, the second and third largest estuaries on the U.S. Atlantic Coast. The Savannah River estuary was studied for several years in connection with a proposed harbor deepening project. This work, with possible expansion, could be a model for monitoring and research in other estuaries in the Carolinas.
- Greater information on the longevity of droughts in relation to long-term impacts and/or length of recovery is warranted. This will enable better decision making for water management during drought periods, allowing more informed impact analysis to balance instream flow with other water needs.
- In investigating the causes of marsh dieback, Cahoon et al. (2011) commented on the importance of long-term, high-resolution elevation data sets to understand the prolonged effects of water deficits on marsh elevation change.
- Many authors, including Griswold et al. (2008), expressed the need for long-term data to allow for better interpretation of the effects of infrequent disturbances on aquatic ecosystems. Additionally, long-term precipitation indices (i.e., 48-month standardized precipitation index) can indicate the likelihood of a return to drought, allowing for the collection of pre-disturbance data.

C. Tools, Models, and Ecological Indicators for Drought Impacts

The Research and Monitoring Group of the CISA workshop discussed in the introduction of this report identified several areas requiring further research. One of the first priorities is to identify which variables are most important to monitor and assess drought-impacted ecosystems (e.g., chemical, physical, or biological) and to determine the appropriate temporal and spatial scales to study this topic. Such a process will entail assessing the quality and appropriateness of currently available data and establish priorities for collecting additional data if necessary. Researchers highlighted the need for better long-term, fine resolution biological and ecological data; information related to climate variability; and socio-economic data. The variability associated with the El Niño Southern Oscillation has received some attention, but other types of climate information are sometimes difficult to obtain. With coastal areas experiencing increasing development and water demands, understanding and integrating social data (e.g., land use change, water allocation rules, and other socioeconomic processes) into water and drought management decisions is also critical. Participants discussed a need for better models to analyze the ecological impacts of drought. Such models would have the ability to detect ecological change, incorporate human influences on ecological impacts, or indicate ecological flow needs. This group also indicated a desire to create a central location for data. This tool would be manifest in the form of a drought portal that would provide access to primary data (water quality, hydrology, meteorology, ecology, biology, etc.) as well as a repository for additional and relevant modeling tools and drought studies.

References cited in this report support the suggestions made at CISA workshop. As noted in the Discussion Section (X.A), the literature revealed use of existing drought indexes (e.g. PDSI, PHDI), some computer modeling and integration of data sets as useful tools for investigating the effects of drought. However, innovative tools for discerning conclusive cause and effect relationships were somewhat limited.

- There is a need to develop a set of indicators of ecological change that may be caused by drought. Most will be ecosystem dependent and many can probably be derived by intervention analysis[10] of existing monitoring data. A search for leading indicators will be most helpful to resource managers and those dependent on a particular ecosystem service that could be degraded or destroyed by drought, especially severe or serial drought.
- Indicator species sensitive to drought-induced changes should be selected for coastal ecosystem communities and studied long-term to develop baseline conditions for comparison with continued sampling during long-term or repeated drought situations. For example, Georgia has initiated such studies with blue crabs (Meryl Alber, personal communication to authors, 2011). Many long-term data sets of great utility may already exist and can be used with new or continued sampling during droughts to better determine cause and affect relationships. Ongoing research can utilize the intervention analysis mechanism mentioned in an earlier bullet.
- Maul et al. (1999) developed a forested wetland soil perturbation index as a reference standard primarily for indicating disturbance from timber harvest. It is a biogeochemical assessment using soil organic matter (SOM), total organic carbon (TOC), total Kjeldahl nitrogen (TKN), and total phosphorus (TP) as possible indicators of resilience. They purport that such a biogeochemical assessment would provide a systems approach to wetland functional integrity for wetlands of similar type within the same geographic region and can be one useful component of an index of biotic integrity for wetland ecosystems. This potentially could be used as a tool for studies to determine changes in wetlands function from drought impacts, although a sensitivity analysis of the method for these purposes would be needed.
- Paerl et al. (2003), Paerl et al. (2007), and Paerl et al. (2010) promoted the idea of phytoplankton as a good indicator of ecological change. Community composition changes substantially between high and low inflow scenarios. With low flow, there is a dominance of dinoflagellates and cyanobacteria. These changes impact eutrophication, food web, biogeochemical (hypoxia), and habitat conditions.
- Humphries and Baldwin (2003), reporting on an Australian freshwater stream conference, stated that innovative approaches, such as microsatellite DNA analyses, can reveal that the effects of drought may be profound and long-lasting, resulting in population bottlenecks and altering the course of the evolution of species.

[10] Intervention analysis or event study is used to assess the impact of a special event on the time series of interest.

- Several authors recommended the use of hydrological models to better understand the biological, chemical, physical processes within a watershed. Dai et al. (2010) ran a physically based distributed hydrological model, MIKE SHE, to evaluate the effects of altered temperature and precipitation regimes on the streamflow and water table in a headwater forested watershed of Huger Creek which drains into the East Branch of the Cooper River near Charleston, SC. Dai et al. (2010) promoted MIKE SHE as a potentially valuable tool for such predictions but encourage further work to improve intermittent streamflow simulation. Similar findings were made in a MIKE SHE modeling study in a mixed pine-hardwood wetland in the Santee Experimental Forest (Lu et al. 2006). Of particular value would be a verified MIKE SHE or similar hydrological model that enables modeling of tidally influenced freshwater streams and wetland systems.
- Hydrological models similar to that described by Conrads et al. (2010) for the Savannah River may be of utility to determine conditions under which isohaline lines may shift upstream based on hydrological alterations including streamflow and sea level rise.
- Because of the strong correlation between hydrological conditions and nutrient loading, Paerl et al. (2004) recommended that basin scale reduction strategies for nitrogen and phosphorus should include hydrologic variability (hurricanes, floods, droughts).
- Resh et al. (1988) identified hydrologic regime, quantified as the long-term patterns of monthly maximum and minimum stream flow, as one of two significant features for assessing the effects of disturbance on stream ecology. Many other instream flow habitat related models and techniques exist and could be of use in modeling the impacts of droughts on stream aquatic habitats. These include the Nature Conservancy's Indicators of Hydrological Alteration (IHA) and the USGS Instream Flow Incremental Methodology (IFIM) and associated hydrological and physical habitat simulation modeling (PHABSIM).
- Michener et al. (1997) recommended utilizing the moisture continuum model for assessing the effects of shifts in moisture regimes on wetland ecosystems. The moisture-gradient model is based on the assumption that a warming and drying climate in a particular region will produce conditions that already exist for wetlands in the present climate, but at different geographic locations; thus assuming that space may act as a surrogate for time.
- Comparing wetlands that currently exist along a moisture continuum offers considerable predictive power (Grimm 1993). This approach assumes that most, if not all abiotic variables (i.e., geomorphologic changes, seasonality of temperature and precipitation, etc.) are characteristic features of intrinsic system states along the moisture continuum. By associating salient abiotic patterns and processes with discrete states that exist along the moisture continuum, one can begin to isolate factors that control dispersal and successful colonization of plant and animal species.
- Development of coupled biological/physical models for the region was included as a "priority need" in the South Atlantic Regional Research Plan with the following rationale: "Coupled biological/physical models are useful for synthesizing information about current conditions, evaluating hypotheses about the relative importance of different forcing mechanisms, and forecasting. Coupled models for the coastal ocean would provide information about circulation patterns, coastal upwelling, and the movement of nutrients and pollutants in the area. They could also be used to evaluate alternative scenarios and project climate and weather impacts such as drought, changes in the quality and amount of freshwater inflow, and flooding" (Laporte et al. 2010, 15-16).

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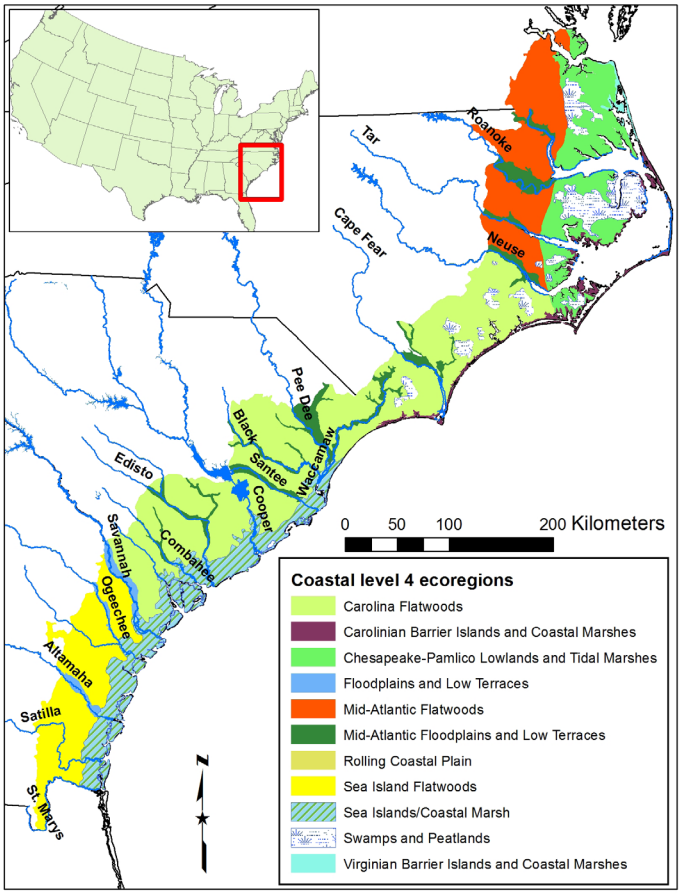
Appendix. Summary of the Level III and IV Ecoregions in the Study Area

The table below, in three parts, is reduced and combined from two tables that were downloaded from the US Environmental Protection Agency Ecoregion website[11]. There are three Level III ecoregions included in the study area: 63 - Middle Atlantic Coastal Plain, 65 – Southeastern Plains, and 75 – Southern Coastal Plain. The table includes a subset of the features that characterize the Level IV ecoregions.

The geomorphology of the southeast coast is generally composed of marshes, dunes, beaches, and barrier islands at the land margin. Inland from that are flatwoods that are periodically dissected by large rivers, many of which are alluvial. The proximity of the flatwoods to the land margin varies by location along the coast. Local distinctions include the relative abundance and extent of Carolina Bays, pocosins, riverine swamps, floodplains, and sounds.

There is significant overlap of the Level IV ecoregions between extreme southeastern Virginia and northeastern North Carolina and between the Georgia coast and the south coast of South Carolina (Fig. 1 is repeated here for visual context).



[11] http://www.epa.gov/wed/pages/ecoregions.htm

Level III or IV Ecoregion	Physiography	Climate			Potential Natural Vegetation	Land Use and Land Cover
		Precipitation Mean Annual (in)	Mean Temp. Jan. min/max	July min/max (F)		
63 Middle Atlantic Coastal Plain						
63b. Chesapeake-Pamlico Lowlands and Tidal Marshes	Low, flat plains and peninsulas; poorly drained with swamps, some low gradient streams with sandy and silty substrates, a few large lakes; estuaries and sounds.	48-55	32/52	69/88	Wet hardwood forest (bottomland oaks, tulip poplar, sweetgum, maple, swamp tupelo); mesic mixed hardwood forest (beech, tulip poplar, maple, oaks, sweetgum); some pond pine woodlands and longleaf pine; tidal and nonriverine cypress-gum swamps.	Cropland with wheat, corn, soybeans, potatoes, cotton, and peanuts; evergreen forest, mixed forest, forested wetlands, pine plantations, pasture, marsh.
63c. Nonriverine Swamps and Peatlands	Low, broad flats and interstream divides, poorly drained, a few lakes, low stream density; channelized drainage is common.	50-56	32/54	69/89	Pocosins (fetterbush, ti-ti, inkberry, pond pine); pond pine woodland; Atlantic white cedar forest; nonriverine swamp forest (bald cypress, pond cypress, swamp tupelo, loblolly pine, red maple).	Forested wetlands, evergreen forest, mixed forest, some cropland and pine plantations.
63d. Virginian Barrier Islands and Coastal Marshes	Barrier islands, dunes, beaches, lagoons, estuaries, tidal marshes.	48	33/52	69/87	Salt and brackish marshes (cordgrass, saltgrass, rushes); tidal freshwater marsh (cordgrass, sawgrass, cattail, wild rice); maritime shrub (wax myrtle, yaupon); maritime dry grassland (saltmeadow cordgrass); maritime evergreen forest (live oak, sand laurel oak, loblolly pine); dune grass (beach grass, sea oats).	Marsh, forested wetland, evergreen forest, urban, wildlife habitat, beaches, recreation, fish and shellfish production.
63e. Mid-Atlantic Flatwoods	Flat plains on lightly dissected marine terraces; swamps, low gradient streams with sandy and silty substrates.	46-50	30/52	68/89	Mesic pine flatwoods (longleaf pine, loblolly pine, oaks, hickories, bluestem); wet pine flatwoods (longleaf pine with loblolly or pond pine); pine savanna (longleaf pine, pond pine, bluestem); pond pine woodland; some oak-hickory and mixed forest.	Pine plantations, cropland with peanuts, cotton, corn, soybeans, tobacco, wheat, chickens, and hogs; pasture, mixed and deciduous forest.

Level III or IV Ecoregion	Physiography	Climate			Potential Natural Vegetation	Land Use and Land Cover
		Precipitation Mean Annual (in)	Mean Temp. Jan. min/max	July min/max (F)		
63g. Carolinian Barrier Islands and Coastal Marshes	Barrier islands, dunes, beaches, lagoons, estuaries, tidal marshes.	50-56	35/53	72/86	Salt and brackish marshes (cordgrass, saltgrass, rushes); maritime shrub (wax myrtle, yaupon); maritime dry grassland (saltmeadow cordgrass); maritime evergreen forest (live oak, sand laurel oak, loblolly pine); dune grass (sea oats, bitter panic grass, cordgrass, beach grass).	Marsh, forested wetland, evergreen forest, urban, wildlife habitat, beaches, tourism, recreation, fish and shellfish production.
63h. Carolina Flatwoods	Flat plains on lightly dissected marine terraces; swamps, low gradient streams with sandy and silty substrates; Carolina bays.	46-53	33/55	70/90	Longleaf pine-wiregrass; xeric sandhill scrub (longleaf pine-turkey oak-wiregrass); pond pine forest and woodland; some oak-hickory and mixed forest.	Pine plantations, mixed forest, forested wetlands, cropland of cotton, corn, soybeans, wheat, peanuts, tobacco, blueberries; production of hogs, broilers, and turkeys; some public land, wildlife habitat.
63n. Mid-Atlantic Floodplains and Low Terraces	Major river floodplains and associated low terraces; low gradient streams with sandy and silty substrates, oxbow lakes, ponds, swamps.	46-54	32/54	69/90	Southern floodplain forest. Includes cypress-gum swamp (water tupelo, swamp tupelo, bald cypress, pond cypress) and bottomland hardwood forest (bottomland oaks, red maple, sweetgum, green ash, bitternut hickory).	Forested wetlands, deciduous forest, some cropland on larger terraces.
65 Southeastern Plains						
65m. Rolling Coastal Plain	Dissected irregular plains and smooth plains; broad interstream divides with gentle to steep side slopes dissected by numerous small, low to moderate gradient sandy bottomed streams.	44-51	29/51	68/89	Mesic pine flatwoods (longleaf pine, loblolly pine, oaks, hickories, wiregrass); oak-hickory forest (southern red oak, post oak, hickories, pines); some mesic mixed hardwood forest (beech, tulip poplar, maple, white oak, red oak, sweetgum).	Cropland and pasture with cotton, soybeans, corn, wheat, sweetpotatoes, peanuts, tobacco, hogs, and chickens; some pine plantations, mixed forest, and forested wetlands.

Level III or IV Ecoregion	Physiography	Climate			Potential Natural Vegetation	Land Use and Land Cover
		Precipitation Mean Annual (in)	Mean Temp. Jan. min/max	July min/max (F)		
65p. Southeastern Floodplains and Low Terraces	Major river floodplains and associated low terraces; low gradient streams with sandy and silty substrates, oxbow lakes, ponds, swamps.	45-64	32/58	68/91	Southern floodplain forest.	Deciduous forest, forested wetlands, pine plantations on floodplains; cropland and pine plantations on terraces.
75 Southern Coastal Plain						
75f. Sea Island Flatwoods	Flat plains on lightly dissected marine terraces; swamps, low gradient streams with sandy and silty substrates.	48-53	38/62	70/92	Southern mixed forest.	Evergreen forest / pine plantations, forested wetland.
75i. Floodplains and Low Terraces	Major river floodplains and associated low terraces; low gradient streams with sandy and silty substrates, oxbow lakes, ponds, swamps.	48-50	35/58	69/91	Southern floodplain forest. Includes cypress-gum swamp (water tupelo, swamp tupelo, bald cypress, pond cypress) and bottomland hardwood forest (bottomland oaks, red maple, sweetgum, green ash, bitternut hickory).	Forested wetlands, deciduous forest.
75j. Sea Islands / Coastal Marsh	Barrier islands, dunes, beaches, lagoons, estuaries, tidal marshes.	48-53	37/58	72/89	Salt and brackish marshes (cordgrass, saltgrass, rushes); maritime swamp forest (tupelo, red maple, sweetgum, bald cypress); maritime evergreen forest (live oak, sand laurel oak, slash pine, loblolly pine); dune grass (sea oats, bitter panic grass, cordgrass, beach grass).	Marsh, forested wetlands, evergreen forest, urban, wildlife habitat, beaches, recreation, fish and shellfish production.

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