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Ecological Status of the Everglades: Environmental and Human Factors that Control the Peatland Complex on the Landscape

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2.1 Introduction

The Everglades was an almost impenetrable wall of sawgrass "plains" and reptile-infested waters according to the early Spanish and American explorers (Ives 1856; Lodge 1994). Its name may have come from the term "Never Glades" as first used by Vignoles (1823). Originally called Pa-hay-okee ("grassy lake") by the resident Native Americans, the Everglades was later popularized and put forward as a threatened environment that needed federal protection by Marjory Stoneman Douglas's seminal 1947 book *The Everglades: River of Grass*. Her wonderful "river of grass" metaphor has unfortunately led to a simplistic view of the complexities of the Everglades ecosystem, how it functions on the landscape, and how its diversity of communities should be managed to sustain this subtropical wetland (McCally 1999). It is often referred to as the "Everglades marsh or swamp" by local residents, biologists, and engineers; however, it is correctly identified as a fen (Richardson 2000; Keddy 2000; Rydin and Jeglum 2006; Grunwald 2006). In more generic terms, the entire wetland would be referred to as a peatland by wetland ecologists in North America or as a mire by those in Europe. A mire is a wet terrain dominated by living peat-forming plants and is often used in botanical and ecological investigations of vegetation types. "Peatland" is a more universal term used to define a terrain covered by peat, usually to a minimum depth of 30–40 cm. Even if the site is drained, it is still a peatland, but if it loses its original peat-forming plants it is no longer considered a mire (Sjors 1948; Rydin and Jeglum 2006). The terms peatland and mire are therefore not used interchangeably by peatland ecologists in Europe and North America. Here we use "peatland" to generally represent the complex diversity of community types found on peat soils in the Everglades and "fen" in a more strict sense to represent alkaline peat or calcium mineral-based ecosystems found over vast portions of the Everglades landscape. Future detailed research on groundwater flows and geochemistry will be needed to distinguish which specific locations within the Everglades function as true groundwater-influenced fens vs. peatland types with surface- or rainfall-dominated inputs (discussed later in Sect. 2.5). Thus, the Everglades should not be classified as a swamp because it is not a forest-dominated wetland, and it is not technically a marsh because marshes are characterized by standing or slow-moving water with submerged, floating-leaved, or emergent plant cover rooted primarily in mineral soil with nutrient-rich overlying waters (Rydin and Jeglum 2006). These are important distinctions when one considers how different marshes and swamps are from peatlands in terms of their hydrologic controls, biogeochemistry, rates of peat accretion, plant and animal communities, and successional development. To maintain some continuity of terms in this volume with historic usage we do use the word marsh to refer to specific community types like cattail marsh. Nevertheless, the terms "Everglades peatland" or "fen" by themselves do not reveal the vital and multifaceted hydrologic connections and nutrient sources that historically existed between the Everglades and surface water runoff coming from Lake Okeechobee via the Kissimmee River, the close connections of groundwater and surface waters in the region due to the karst limestone underlying the wetlands, and most importantly the seasonal influence of the key water source – rainfall (Parker et al. 1955; see Chap. 7).

The Everglades peatland complex was created by blocked drainage due to development of limestone substrata of various porosities overlain on a flat basement rock and confined by sandy ridges that developed from sea level rise and fall about 5,000 years before present (YBP) (Gleason and Stone 1994). The thin layer layers of porous rock that formed during earlier glacial periods absorbed, stored, and transmitted water at different rates, a characteristic crucial to the formation of a myriad of different plant communities found in the Everglades mire even today (McCally 1999). For example, the landscape, while dominated by sawgrass, is interlaced with periphyton-rush sloughs, wet prairies, and ponds, and it is dotted with tree islands and willow heads. The proportion of each community type varies greatly along a north-to-south hydrologic gradient (see Chaps. 4, 7–9, and 12). Most of these plant community associations evolved under low phosphorus (P) concentrations because the main source of water was rainfall with extremely low P concentrations (Redfield and Urban 1997). The exception to communities evolving under low P concentrations are tree islands and the vegetation around alligator holes (Davis 1943; Loveless 1959; Steward and Ornes 1975b; Craft and Richardson 1993a; Sklar and van der Valk 2002) as well as plant communities adjacent to Lake Okeechobee with its high historical TP concentrations >30 $\mu g\ l^{-1}$ (Walker 2000). Another factor maintaining P limitations in the Everglades, unlike northern peatlands or fens, is the nitrogen-fixing blue-green algae community, or periphyton, found in open-water sloughs. Because of the periphyton community's high rates of nitrogen fixation, Everglades soils are exceptionally high in nitrogen (2–4% by weight); thus, very high N:P ratios (>100) exist, further driving the system to severe P limitations (Richardson et al. 1999).

To fully understand the Everglades ecosystem, it is necessary to understand how human interventions over the last one hundred years have dramatically altered the natural Everglades development processes that started more than 5,000 years ago. Thus, the objectives of this chapter are to provide the reader with a basic foundation for understanding how the Everglades ecosystem complex has developed and to provide an analysis of factors controlling ecosystem functions in the Everglades today. It is not our intent to review in detail the geological formations and processes that have led to the development of the Everglades, as so many great articles and volumes have been written on this topic (e.g., Brooks 1968; Gleason 1974a; Perkins 1977; Gleason and Stone 1994). To accomplish our goal and help interpret the specific research and restoration lessons presented in the chapters that follow we (1) present a brief review of Everglades peatland formation and characterize the wetland processes that led to development of this vast peatland complex, (2) provide a proper classification of the Everglades system that might help in the development of a more appropriate restoration management framework, (3) review the factors controlling ecosystem structure and succession of communities found within the peatland complex today as compared to historical conditions, and (4) compile and synthesize historical and current data on some key elements of precipitation trends, hydrologic shifts, and nutrient inputs on a landscape scale.

2.2 Formation of the Everglades: The Historical Everglades Prior to Major Anthropogenic Impacts

One of the key benefits of examining the long-term history of the Everglades is that it is possible to learn about natural variations in the system prior to the industrial era as well as determine what environmental factors controlled the formation and development of the Everglades. Knowledge of the rates and magnitudes of change, as well as of recovery rates from disturbances, is critical to future restoration plans. Restoration plans that incorporate natural variation and known responses to disturbance are also likely to be more ecologically and economically feasible.

The Everglades mineral substrate formed a large basin or trough during the Pleistocene, and shallow marine sediments were deposited, primarily during the Sangamon interglacial stage 125,000 YBP (Davis 1943; Parker and Cooke 1944; Gleason 1984). The retreat of the northern U.S. glaciers 18,000–16,000 YBP, blockage of drainage from the Everglades due to rising sea level, a change to a subtropical climate, and the concurrent increase in rainfall allowed for the development of the Everglades as we know it. Three limestone formations underlie the Everglades. The Miami Formation is found in the southern Everglades National Park (ENP) region; the Anastasia Formation, comprised of sandy calcareous sandstone, is found in the northeast area; and the Fort Thompson Formation, which underlies the northern half to a depth of 50 m, is mostly marine and freshwater marls, limestone, and sandstone (Enos and Perkins 1977).

A geological study of the bedrock that underlies the Everglades shows a differentiation in permeability from north to south. Low-permeability limestone underlies the northern portion of the Everglades basin around Lake Okeechobee and extends into the northern half of WCA-3 and into the western portions of WCA-2. In the southern section of WCA-3 and the southeastern section of WCA-2B, there is an abrupt shift to highly permeable limestone (Gleason et al. 1974; Perkins 1977). This has important ramifications for the movement and storage of water, peat development, and the establishment of plant communities. Moreover, construction of any water storage areas in the lower eastern part of the Everglades would be subject to severe water loss unless extensive and expensive efforts were made to line the reservoirs due to the high permeability of the underlying bedrock. According to Gleason et al. (1974), bedrock configuration established the drainage directions prior to peat deposition in the Everglades. For example, Lake Okeechobee flowed through a channel eastward to the area now known as WCA-1, and a deep depression bisected the lower Everglades and created a southwest flow gradient toward Florida Bay (Fig. 2.1). These patterns appear to have changed little over the course of time. For example, Gleason et al. (1974) note that tree island orientation is correlated with drainage directions expected from bedrock topography. The only detailed vegetation map of the Everglades came from early survey work of Davis (1943) and was based on his extensive field observations in the late 1930s (Fig. 2.2). Although, the mapping was done prior to any massive increase in farming in the Everglades Agriculture Area (EAA), many of the large canals had been dug and peat subsidence had started according to his field notes. His map provides distributions

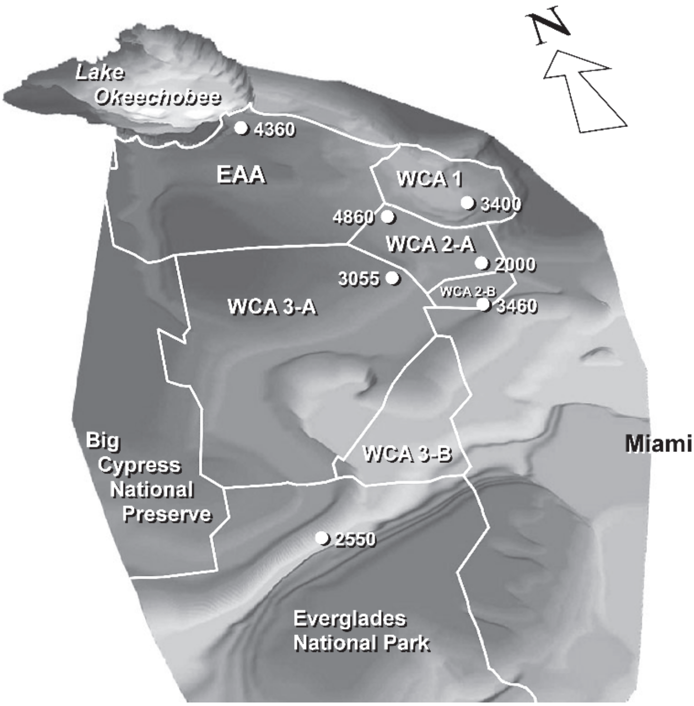

Fig. 2.1 Bedrock map of the Everglades prior to peat development based on kriging of USGS depth measurements and isopleths maps (Parker et al. 1955; Parker and Cooke 1944). *Darker shades* represent higher regions (bedrock plateau south of Lake Okeechobee, etc.) and *lighter shades* represent depressions or troughs in the bedrock (e.g., in WCA-1 and in lower WCA-3A, WCA-3B and in the northern portion of the ENP where Taylor slough is now found). Also shown are basal dates of peat from ^{14}C measurements (McDowell et al. 1969; Gleason et al. 1974; Craft and Richardson 1998)

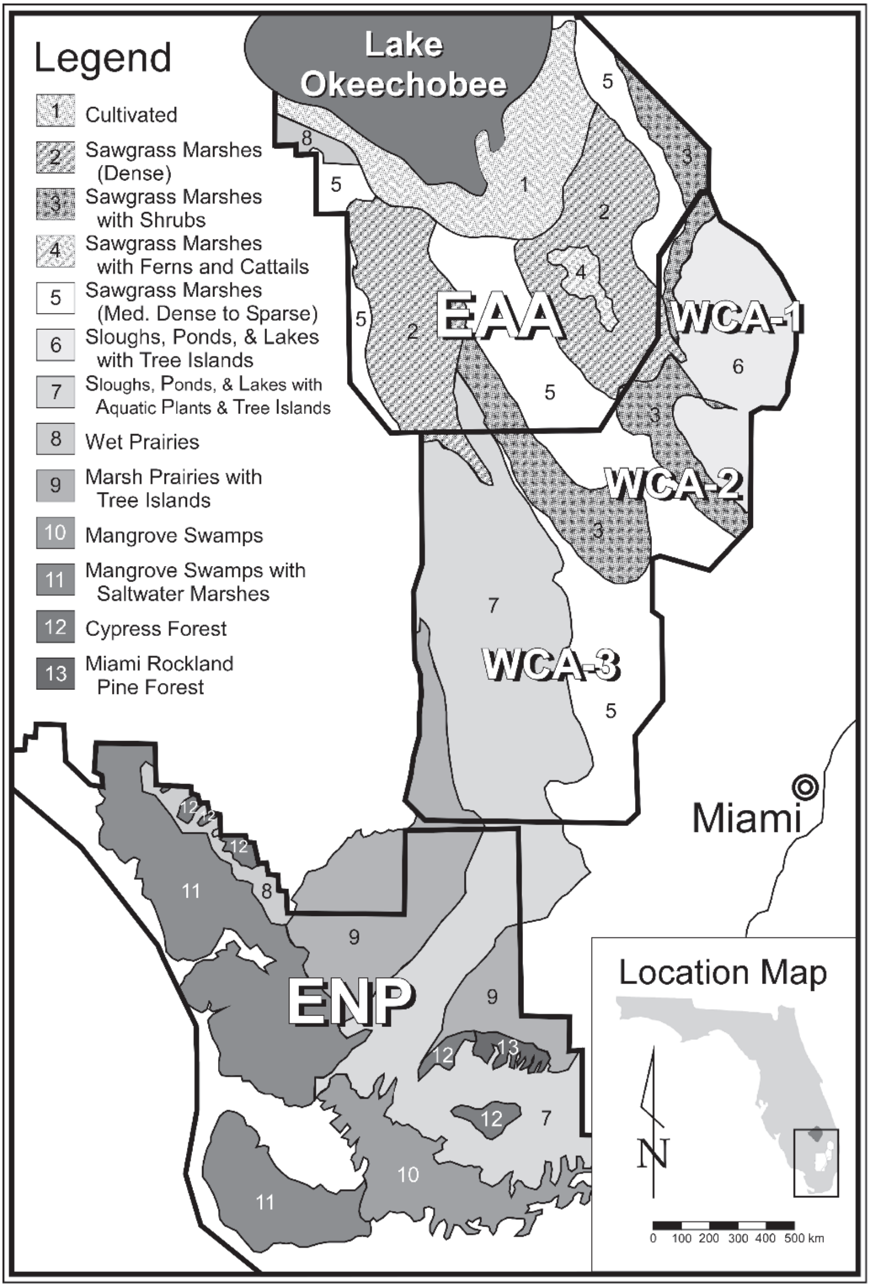

Fig. 2.2 Historic map of the vegetation communities in the Everglades based on the map of Davis (1943). The map has been redrawn and simplified from the original map, and the boundaries of the current Water Conservation Areas (WCA-1, WCA-2A, and WCA-3A), the Everglades National Park (ENP), and the Everglades Agricultural Areas have been added

of all the major plant communities and indicates that higher densities of sawgrass stands existed in the northern Everglades. He clearly mapped wet prairies and ombrotrophic (rainfall-driven) areas in WCA-1. The map also demarks slough and ponds areas, mangrove swamps, and vast areas of tree islands as well as pointing out a large stand of ferns and cattails (>8,000 ha or 20,000 acres) in what is now the EAA. The reason for the prevalence of cattails, which are commonly found in P-enriched areas, is unknown because farming had not started to any extent; however, Davis (1943) suggested that it may have been related to fires occurring shortly before the mapping, which would have released P from the burned peat.

2.2.1 Peat Formation

Paleoecological studies based on the examination of peat or soil cores have been used to provide information about the Everglades system since its formation (Gleason et al. 1974; Willard et al. 2001; see Chap. 12). Radiometric dating analyses such as ^{14}C and ^{210}Pb allow researchers to date the soil strata in the cores. Peat type and pollen remains provide a picture of what the vegetation of the area was like at each time period. Diatom remains provide information concerning water quality in the past, particularly past pH and nutrient levels. A view of the general location of the numerous cores that have been analyzed in the Everglades by various researchers and summarized in this section is given in Fig. 2.3.

Peat formation in the Everglades began around 5,000 YBP in the northern Everglades and around 2,000–3,000 YBP further south (Gleason and Stone 1994). These dates are the result of ^{14}C dating of basal peats in numerous peat cores. Thus, the Everglades is geologically a relatively young ecosystem. The rate of peat accumulation from north to south has always been of interest, and peat cores collected from four sites in the Everglades – one each from Loxahatchee, WCA-2, WCA-3A, and the ENP – indicate an interesting trend. Accelerated mass spectrometry ^{14}C dates for samples from depths between about 40 and 50 cm indicate that there has been more peat accumulation in the north. For instance, the date estimated for the 36–38 cm depth from a core in ENP was about 2,400 YBP. However, for the core collected from WCA-2A the date estimated for the interval 33–48 cm was only about 800 YBP (C.J. Richardson, unpublished data). This suggests different patterns of peat initiation and much higher rates of accretion in the northern Everglades compared to the southern. Another indication of this trend was an analysis of peat depths in WCA-2A vs. WCA-3A. Ninety depth probes to bedrock taken during our grid soil surveys in 2001 (Chap. 6) throughout WCA-2A averaged 144.5 ± 37.3 cm in depth, with a median value of 144 cm and a range from 62 to 252 cm (Fig. 2.4). Several deep peat areas were found in the northwest and southeast portions of WCA-2A. Peat was generally deeper in the northern than the southern part of WCA-2A. Fifty-nine depth probes taken in 2000–2003 throughout WCA-3A averaged 80.1 ± 36.3 cm in depth, with a median value of 76.5 cm and a range from 16 to 180 cm. Peat depths were shallowest in the northern part of WCA-3A

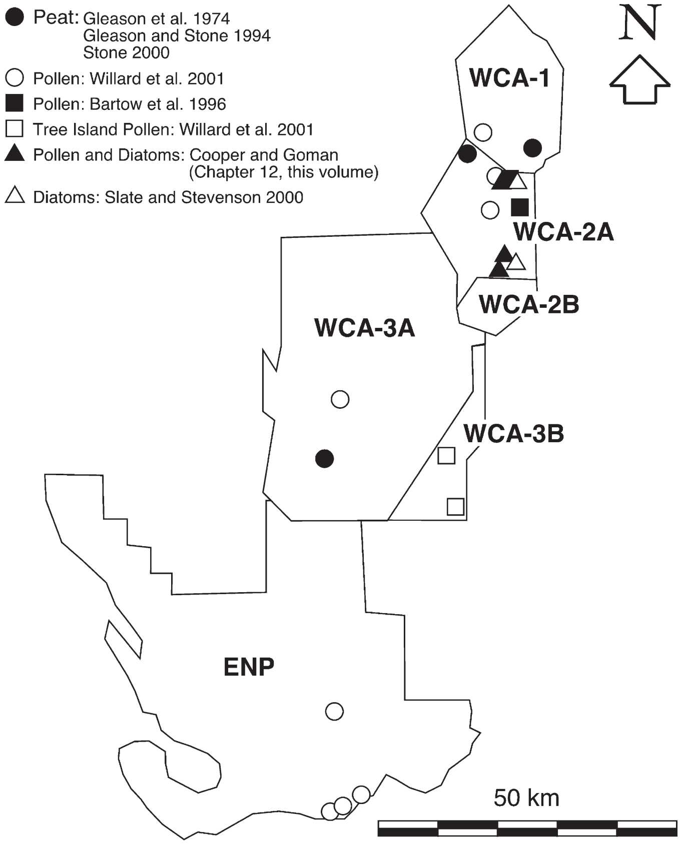

Fig. 2.3 Location and sources of peat core sites in Water Conservation Areas and the Everglades National Park used to assess historic communities

compared to the southeast corner of WCA-3A (Fig. 2.5). Severe fires in the north along with dry, peat-oxidizing conditions may have contributed to the shallower peat depths in the north. Thus, peat depths in the more northern WCA-2A were 1.8 times those found in the southern Everglades in WCA-3A and clearly demonstrated a north-to-south trend in peat accumulation over the past 5,000 years (Chap. 3), corroborating the evidence from the radiometric dating of peat cores.

The soils of the Everglades are recent Holocene Histosols and Inceptisols (Gunderson and Loftus 1993). The soils are primarily peats and mucks that had accumulated to a depth of nearly 4 m in the north but are less than 20-cm deep in portions of the ENP (Stephens and Johnson 1951; see Chap. 3). Thus, historic northern peat depths are nearly twice what is currently found. The deepest peats in the southern Everglades are found in depressions and major water flows, such as the Shark River slough. Gleason (1984) dated the basal peats and found that peat deposition began as early as 5,490 ± 90 YBP, but most peats date from 2,000 to

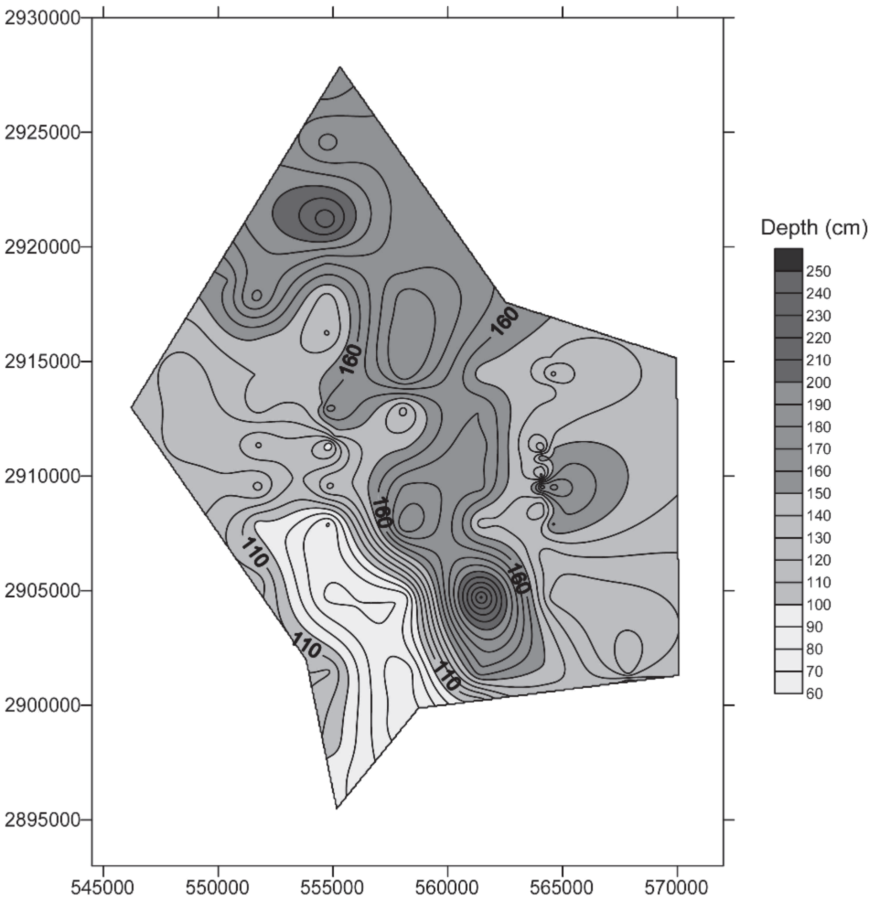

Fig. 2.4 Isopleths of peat depth to bedrock in WCA-2A taken during 2001. The map is based on 90 sample depth measurements

4,500 YBP. However, the tree islands are more recent formations and date only from 1,300 YBP (Gleason and Stone 1994; Craft and Richardson 1998). The other dominant and oldest soil type is a calcitic mud, an Inceptisol, formed by cyanobacteria (blue-green algae) that reprecipitate calcium carbonate or marl ($CaCO_3$) originally derived from the limestone substrate (Browder et al. 1994). It is found underlying most of the peatlands and has been dated at 6,470 YBP (Gleason and Stone 1994). It is also sometimes found in layers within the peat, indicating periods of short seasonal hydroperiod as compared to the longer period of flooding required for peat formation by macrophytes.

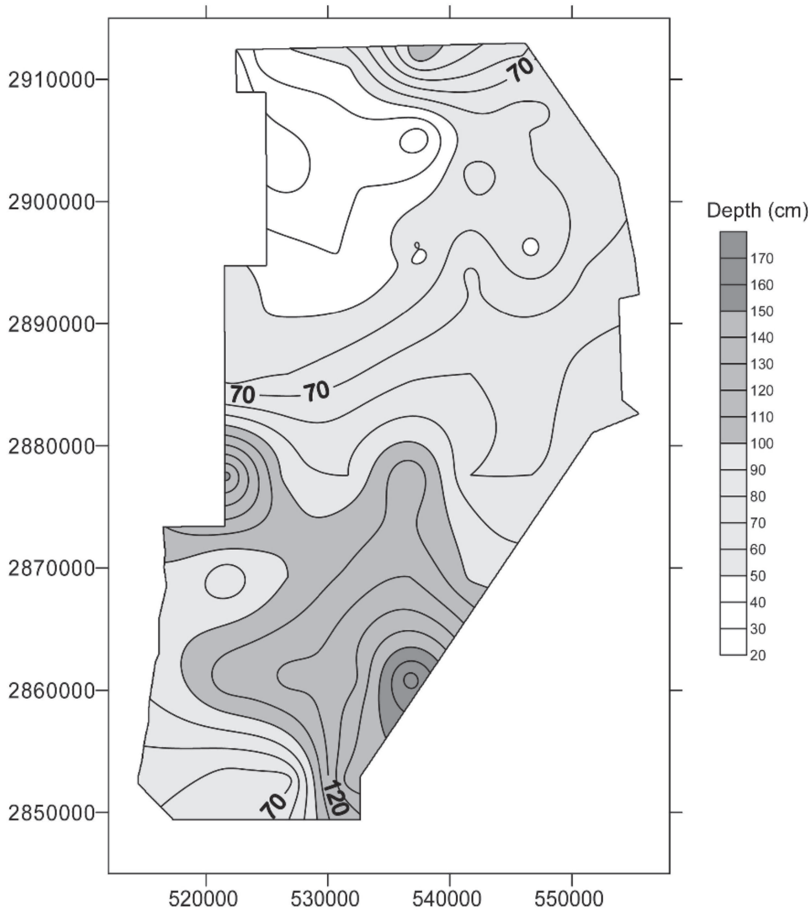

Fig. 2.5 Isopleths of peat depth to bedrock in WCA-3A taken during 2001. The map is based on 59 sample depth measurements

2.2.2 Peat Analysis as a Tool to Determine Environmental Changes

Data obtained from the examination of a number of peat core stratigraphies indicate that between 2,000 and 3,000 YBP there was probably a period of reduced seasonal flooding in the central to southern Everglades (Fig. 2.6; Gleason et al. 1974). Data from sites in the north were less conclusive, indicating only that an environmental change had occurred. Several peat cores (Gleason et al. 1974; Gleason and Stone 1994; Stone 2000) indicate that vegetation shifts at a given coring site were common over the last 4,000–5,000 years in the Everglades (Fig. 2.6). Unfortunately, many of these cores have only a basal date, making it difficult to determine exactly when

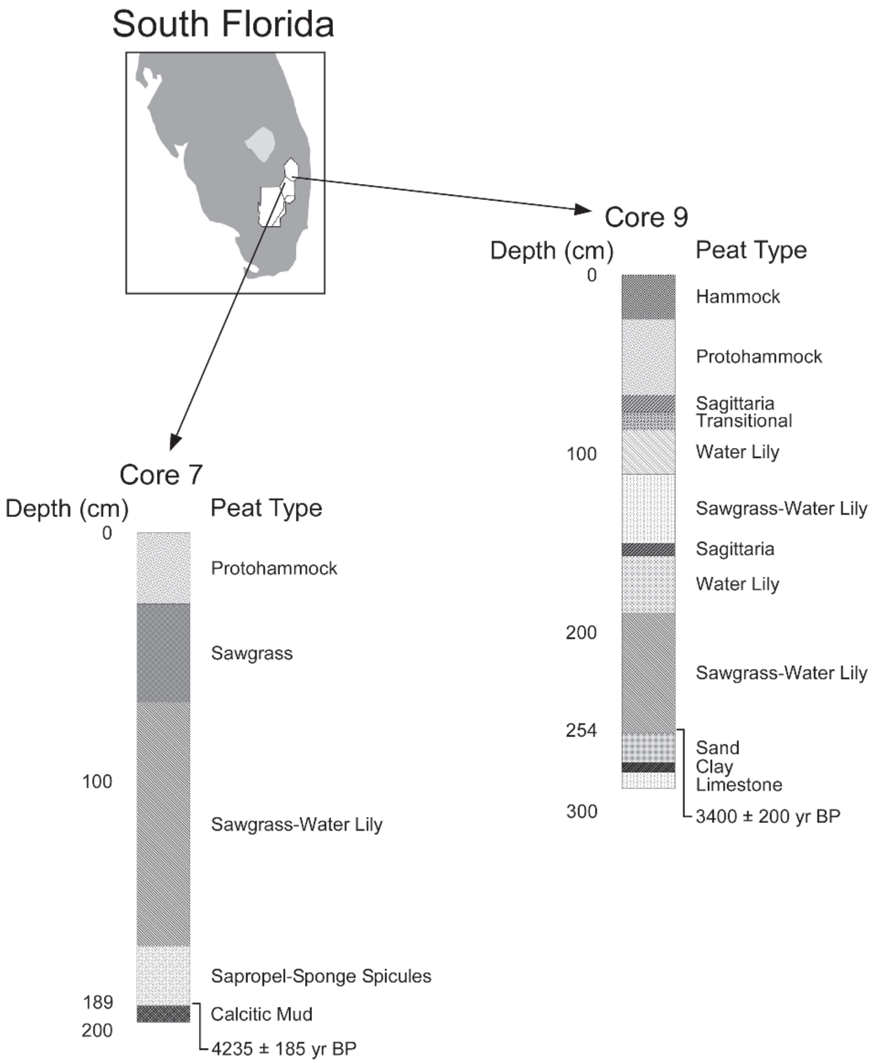

Fig. 2.6 Peat stratigraphy of tree islands from WCA-1 and WCA-2, modified from Gleason et al. (1974)

these vegetation shifts occurred. In a peat core collected from a tree island in the northern region of WCA-2A (core 7), there was a bottom ^{14}C date of 4,235 YBP in a sapropel layer. Below this dated horizon was a calcite mud layer. Calcite mud layers in peat cores are thought to represent drier periods that favored the precipitation of calcium carbonate. Above the sapropel layer is a thick layer of sawgrass–water lily peat that probably spans a period of about 2,000 years (between about 3,500 and 1,500 YBP). Above this layer is a thinner sawgrass layer, probably spanning the period between about 500 and 1,500 YBP, followed by a proto-hammock layer

(Fig. 2.6). Although there was no cattail peat found in this core, cattail pollen has been found in the fen (sawgrass/water lily) peat toward the bottom of the core, suggesting that cattail was abundant during the initial stages of peat formation (Stone 2000). Peat cores typically have alternating layers of sawgrass and water lily community indicators over much of their lengths, as shown in core 9 from WCA-1 (Gleason et al. 1974; Gleason and Stone 1994; Stone 2000). It is not known if these vegetation shifts are the result of climatic or hydrologic changes or whether they simply reflect lateral shifts in the vegetation mosaic (Stone 2000). However, of critical importance for restoration and management plans, the data suggest that these Everglades community types were capable of rapid reestablishment when conditions were favorable.

2.3 Indicators of Changed Hydroperiods and Drainage Effects

A study of pollen stratigraphies by Willard et al. (2001) found that between 1,200 and 2,000 YBP, there was an interval of longer hydroperiods and relatively deep water in the northern Everglades as evidenced by the pollen profiles from western WCA-1 and northern WCA-2A. There were shallow water conditions and probably droughts between about 1,200 and 800 YBP. After 800 YBP, there was a return to wetter conditions in the northern Everglades. Pollen indicative of slough vegetation was common until about AD 1930, when there was a shift to taxa indicative of shallow water (Willard et al. 2001). During the last century, there have been rapid changes in the plant communities in the Everglades. Both pollen and peat type show that vegetation patterns over the last 500 years were relatively stable in comparison to the major changes that took place in the twentieth century (Gleason and Stone 1994; Willard et al. 2001). In general, the landscape was less fragmented and less compartmentalized prior to the twentieth century. More recently, there has been a shift in vegetation and loss of sloughs and tree islands (Chaps. 8 and 9).

The pollen data suggest that anthropogenic changes to the hydrology of the Everglades had an impact on the vegetation as early as AD 1930, as evidenced by increases in weedy annuals (Willard et al. 2001). Although anthropogenic impacts are often thought to have occurred mainly in the northern sections of the Water Conservation Areas (WCAs), pollen profiles from the southern areas also reveal the effects of drainage (Bartow et al. 1996; see Chap. 12). These changes in the pollen stratigraphy likely reflect changes in the drainage patterns. For instance, by AD 1917, four major canals had been dug, and by AD 1928, the Tamiami Trail construction was complete (Light and Dineen 1994). The next period of development did not occur until the early 1950s, when the eastern perimeter levee was constructed, as well as water control structures that allowed discharge from what was to become WCA-2 into WCA-3. In the 1960s, the WCAs were established, and water control structures that regulated flow from WCA-1 to WCA-2 were constructed (Light and Dineen 1994).

Concomitant changes in vegetation occurred between AD 1950 and AD 1960 when there were increases in sawgrass, indicating a drier hydrologic regime. Willard et al.

(2001) assert that the drying was induced by anthropogenic changes in the area (e.g., building of canals and levees and the establishment of the WCAs) because rainfall patterns during this period show a higher than average precipitation. Therefore, the drier conditions were not simply the result of climatic changes in precipitation. Bartow et al. (1996) also found that there was a shift to pollen indicative of drier communities after about AD 1950 in cores from the northern and southern portions of WCA-2A. Cooper and Goman (see Chap. 12) saw a similar shift toward drier conditions after about AD 1960 based on pollen from cores in WCA-2A.

Another important impact of anthropogenic changes to the natural hydrology of the system is the loss of tree islands. Prior to ca. AD 1950, tree islands were more common in WCA-2A (Chap. 8). The decline has been attributed to water impoundment during the 1960s that resulted in the drowning of tree islands. A study of soil cores collected from existing tree islands indicates that tree islands had formed by about AD 1200. Tree islands were also found to have substantially higher soil TP (concentrations often exceeding 2,000 $\mu g\ g^{-1}$ of TP in the soil profile) and higher elevation than the surrounding fen areas (Willard et al. 2002). This increased P in the tree island soils is due in large part to the roosting of birds on the islands. Thus, tree islands are P hotspots where the birds move P from low P habitats such as sloughs via fish and insect harvesting and deposit it on tree island soils as concentrated bird droppings or guano. These transfers have a significant effect on the increased growth patterns in vegetation surrounding tree islands, especially downstream. Moreover, when these islands burn, P is released downstream, further increasing vegetation growth and diversity (C.J. Richardson, unpublished data). Thus, the decline in tree islands has important implications for the Everglades ecosystem.

2.4 Evidence of the Recent Effects of Anthropogenic Disturbance

2.4.1 Paleoecological Evidence

Although many of the major impacts in the Everglades occurred after AD 1950, anthropogenic effects are evident earlier in the century as demonstrated by both pollen and diatoms in soil cores profiles. Paleoecological studies of the Everglades that span millennia as opposed to recent decades show that vegetative communities have shifted in the past prior to anthropogenic impact; however, these communities typically persisted over long time periods (centuries).

Willard et al. (2001) also found that weedy annuals increased around AD 1930, probably due to drainage and even nutrient inputs. Public outcries also suggest that the landscape experienced substantial anthropogenic perturbations prior to AD 1950. Small (1929) published a book entitled *From Eden to Sahara: Florida's Tragedy* in which he decried the destruction of natural communities in south Florida.

Evidence of nutrient-rich agricultural runoff and runoff from Lake Okeechobee is documented in soil cores from WCA-2A. Cores collected from areas near the outflow of canals from the EAA show increases in *Typha* pollen after AD 1960 (Fig. 2.7;

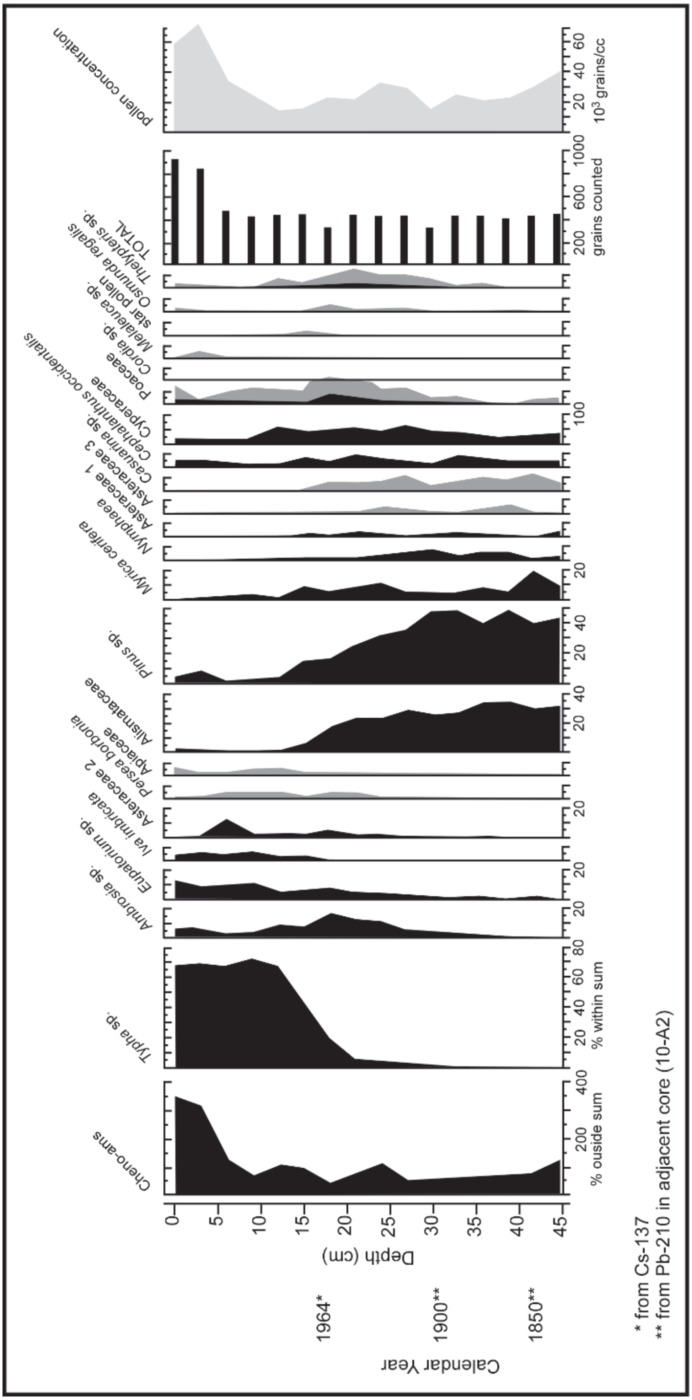

Fig. 2.7 Pollen profile showing changes in vegetation from early 1800s until 1995 in a peat core from an enriched part of the C transect (10C-1) in northern WCA-2A (from Bartow et al. 1996)

Chap. 12; Willard et al. 2001). Furthermore, analyses of diatoms from soil cores from WCA-2A show that diatom species indicative of eutrophic conditions start to increase after AD 1960 (Slate and Stevenson 2000; see Chap. 12). These data suggest that diatom communities may serve as early warning indicators of eutrophication.

Soil cores collected in various areas of the Everglades have provided details concerning how vegetation has changed over time. Cores collected from the nutrient-enriched sites (10-C1) contained two distinct zones, the recent zone 1 (0–17 cm) and the older zone 2 (17–45 cm) (Fig. 2.7). The 0–17 cm depth was predominantly comprised of *Chenopodia–Amaranthaceae* (Cheno-ams) and cattail (*Typha* spp.). Cheno-ams, such as pigweed, are indicators of terrestrial environments and frequently colonize disturbed areas such as fallow agricultural lands. Cattail often is found in nutrient-enriched areas or areas characterized by extended hydroperiod and/or higher water levels in the Everglades where it has displaced slough communities or weakened sawgrass areas. The great increase in cattails in the 1960s and 1970s closely follows the artificially higher water levels maintained by the south Florida Water Management District during this period (SFWMD 1992, Fig. 8.4). The increase of other terrestrial species, such as ragweed (*Ambrosia*), dog-fennel (*Eupatorium* type spp.), beach-elder (*Iva imbricata* Walt.), and an unclassified member of the *Asteraceae* family, suggest the area was drier during the 1980s than in the past. Combined, these trends indicate major shifts in hydroperiod patterns for the site.

Below 17 cm, the pollen is predominantly *Alismataceae* and pine (*Pinus* spp.). The *Alismataceae* includes emergent aquatic vegetation, such as duck-potato and arrowheads, which are common in slough communities where surface water is generally present most of the year (Fig. 2.7). Water lily pollen, another component of the slough, also is greater below 17 cm. The large amount of pine pollen below 17 cm is attributable to original pine forests that existed on the Atlantic Coastal Ridge, north of Big Cypress National Preserve, and on the sandy flatlands northeast of the WCAs (McPherson et al. 1976). Pine is a prolific pollen producer, and the pollen is easily dispersed by the wind; therefore the pine pollen in the soil cores may not be indicative of local vegetation (Brown and Cohen 1985). The pollen of several species such as buttonbush (*Cephalanthus occidentalis*), grasses, and marsh ferns (*Thelypteris*-type ferns) are present in relatively constant amounts throughout the core. The presence of these species throughout the profile suggests that the large proportion of Cheno-ams and cattail are not diluting the other pollen types in the core. Sawgrass (Cyperaceae) is also present throughout the core, but decreases sharply around 13 cm, suggesting its displacement by other species, especially cattail and Cheno-ams.

The distribution of pollen in a core from the unenriched site (Fig. 2.8) lacks some trends present in the core from the nutrient-enriched site. Care must be taken in comparing these two sites because the peat accretion rate at the unenriched site is much lower. Changes in this core correspond to a period of intensive efforts to regulate the hydrology of the remaining Everglades, including construction of canals and levees along the eastern perimeter of the Everglades and construction of the WCAs (Light and Dineen 1994). However, there also appears to be a shift in the distribution of pollen types 8 cm below the surface corresponding to the late 1940s

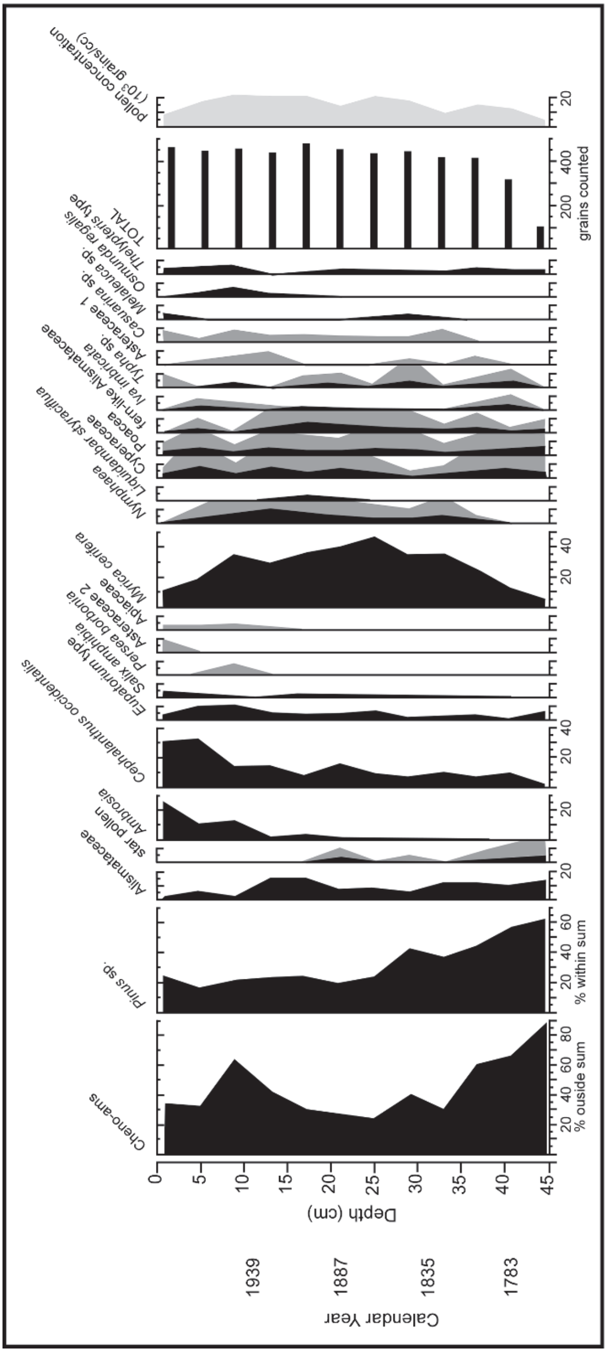

Fig. 2.8 Pollen profile showing changes in vegetation from early 1800s until 1995 in a peat core from an unenriched part of the C transect (10C-6) in southern WCA-2A (from Bartow et al. 1996)

and early 1950s. This 0–8 cm zone has a large amount of pollen from Cheno-ams, ragweed, and buttonbush, as well as a large but decreasing amount of wax myrtle (*Myrica cerifera*). Of note is the fact that these species really increased in the pollen count in the early 1950s. As mentioned previously, Cheno-ams and ragweed are indicators of terrestrial environments. Buttonbush is a plant species that exists under stable hydrologic conditions (BAPM 1988). The large proportion of buttonbush pollen in this zone in the past few decades may reflect a different hydroperiod caused by impoundment of WCA-2A in 1961. Jones (1948) observed that wax myrtle has been increasing in areas dominated by sawgrass during and after the construction of the canal system. It has been hypothesized that this increase in wax myrtle is a result of a lower water table and reduced fire frequency (Loveless 1959). In the recent years, this species appears to have decreased in importance, possibly due to shifting hydroperiod pattern, i.e., alternating periods of extremely high and low water levels controlled by the District (Fig. 8.4).

Below 8 cm (prior to about 1950), there is an increase in pollen representative of aquatic vegetation such as arrowheads and water lily. The increase in aquatic vegetation at depth in this core and in the core from the enriched site suggests that anthropogenic hydroperiod alterations have resulted in a reduction in the extent of aquatic (slough) communities during the past 50 years. This decrease in slough habitat suggests that the present-day Everglades are drier than in the past, presumably as a result of anthropogenic drainage activities. Of interest is the periodic but regular increase and decrease of sawgrass and *Typha* spp. during the past 200 years. This may reflect long-term weather patterns and resulting changes in hydrologic conditions. As in the enriched core, pine pollen is greatest near the bottom of the core and decreases toward the top. This decrease is most likely explained by logging operations that occurred on the eastern seaboard as the urban communities began to rapidly expand as well as in mid-Florida as the land began to be developed for agriculture (Watts 1975). Several additional arboreal species, Australian pine (*Casuarina* spp.), and Brazilian pepper (*Schinus terebinthifolius*) as well as the wetland species melaleuca (*Melaleuca quinquenervia*) are known as exotic weed species throughout southern Florida, but they are virtually absent in either core except at the upper 5 cm for *Melaleuca*. These species thrive on dry upland areas or in association with tree islands and are relatively sparse in WCA-2A, comprising less than 10% of the fen area (SFWMD 1992). *Melaleuca*, however, has become a major weedy pest species throughout the Everglades and now is found on thousands of hectares as a pure monoculture (Bodle et al. 1994; SFWMD 2006).

2.4.2 Changes in Water Quality as a Result of Canal Digging

The examination of diatom remains revealed another effect that the construction of canals had on water quality. Two studies found a higher incidence of the acid-loving diatom genus *Eunotia* prior to AD 1960 (Slate and Stevenson 2000; see Chap. 12). This indicates that conditions were more acidic prior to about AD 1960. The change is probably related to the deepening of drainage canals that took place in the late 1950s, which exposed the underlying limestone bedrock, thus reducing acidity and likely increasing calcareous periphyton mat formation. This suggests that the extensive periphyton mats that we see today in the northern Everglades may not have been present prior to the digging of the large canals.

2.4.3 Summary of the Environmental History of the Everglades

A summary timeline of general environmental changes in the northern, central, and southern Everglades based on the information presented earlier is presented in Fig. 2.9. The paleoecological analysis clearly shows that the Everglades is a dynamic ecosystem that has shown major shifts in plant communities over the last 5,000 years. Peat initiation started earliest in the northern Everglades and last in the south according to basal peat dating. There are also inherent differences between the northern and the southern Everglades as evidenced by bedrock geology, peat type, hydrology, vegetation, and diatoms. The northern part of the Everglades has historically shown higher peat accretion rates compared to the southern areas. The most dramatic changes have taken place during the twentieth century and are associated with development, particularly agricultural practices and the construction of drainage canals. These changes have included a loss of slough areas and tree islands, and an increase in pH. Changes in the northern Everglades affected by agricultural runoff resulted in higher incidences of eutrophic diatoms. Later changes included the replacement of sawgrass and slough communities by *Typha* (Fig. 2.7). A loss of acidophilic diatoms occurred after about AD 1960, probably related to canal digging that exposed limestone bedrock (Chap. 12). These shifts in flora and an understanding of their causes have important implications for restoration. For example, the plans that are focused on maintaining a calcareous periphyton mat community in areas that were formerly acidic, and therefore probably had few periphyton mats, may not aim for the appropriate community target if historic community restoration is the goal. Furthermore, as the bedrock becomes less exposed over time by soil/peat accumulation, there may be less calcium and the pH of the water may decline, resulting in a loss of calcareous periphyton cover.

2.5 Classification

As mentioned earlier, the Everglades would be classified as a fen or peatland by most of the world's peatland ecologists. It is classified as a Palustrine SYSTEM, CLASS Emergent Wetland, SUBCLASS Persistent, WATER REGIME, Semipermanently Flooded, WATER CHEMISTRY Fresh-Circumneutral, SOIL, Organic according to the USFWS wetland classification (Cowardin et al. 1979). However, a more comprehensive functional peatland classification is needed to

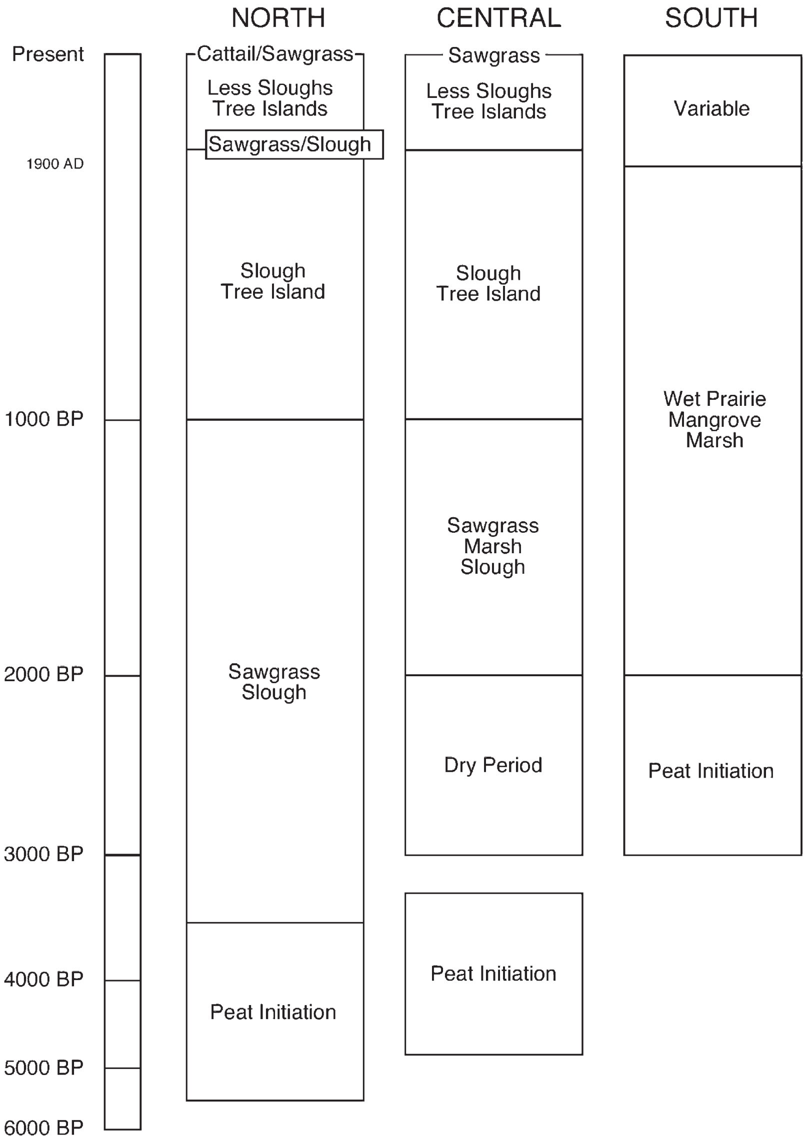

Fig. 2.9 A summary timeline diagram depicting the overall trend in the vegetation history of the northern, central, and southern Everglades from peat initiation around 5,000 years ago up to the present. The diagram is based on composite information from cores taken throughout the Everglades (see Fig. 2.3)

incorporate the sources of water and provide some indication of potential differences in its nutrient status across the landscape.

Several complex environmental gradients caused by the geological substrate, hydrologic flows, and nutrient inputs were responsible for the distinct formation of the Everglades peat types and the north-to-south decrease in peat depths. The degree of wetness and aeration, along with gradients of calcium, salinity, pH, and nutrients, controlled the development of the plant communities (Chap. 12). In addition it has long been recognized that the origin of mire water is a major factor controlling peatland development (Du Rietz 1954). These formation features help define its classification. For example, the Everglades is often described as a rainfall-driven ecosystem, which would normally classify it as an ombrogenous peatland (nourished only by precipitation). However, due to the size and complexity of the Everglades it cannot be easily placed into a single classification. To emphasize the chemical source on productivity, wetland ecologists today refer to ombrogenous sites as ombrotrophic. In the past thousand years before drainage ditches and peat oxidation took place, ombrotrophy may have been true for many portions of the Everglades, but is relevant today only for the raised center portion of the Loxahatchee (WCA-1A). Today, massive subsidence of peat has taken place due to drainage from deeply cut canals and ditches. Large portions of the Everglades are now nourished by waters that have passed over or through calcareous mineral parent soils and are then released through canal gate structures (Stephens and Johnson 1951; see Chap. 7). Chemistry profiles and gradients found within the Everglades (Chap. 6) make it clear that portions of the Everglades are currently nourished by mineral groundwater and would be referred to as minerogenous or in modern terms are minerotrophic systems (Richardson 1995; Vaithiyanathan and Richardson 1997a). Thus, the current Everglades ecosystem is not simply rainfall driven.

The Everglades peatland complex was historically a mixture of defined hydrologic system types. For example, to further define the peatland water flow regime, the term minerogenous is further divided into major hydrologic systems know as topogenous, soligenous, or limnogenous peatlands (von Post and Granlund 1926; Sjors 1948). Topogenous peatlands have flat water tables located in basins with no outlet or a single outlet and inlet. Soligenous peatlands have a slope with directional water flow through the peat or surface. Limnogenous peatlands are located along lakes and streams and are flooded by these waters (Rydin and Jeglum 2006). To simplify terminology, many peatland ecologists would follow the convention of using the term fen for minerogenous and bog for the ombrogenous types. Thus, historically the Everglades peatland complex would have had several types of dominant hydrologic systems. For example, historically a limnogenous peatland that was historically located along the southern edge of Lake Okeechobee no longer exists due to the building of the Hoover dike in the 1930s (Fig. 2.10a). The main hydrologic system for the Everglades would have been classified as soligenous with a minor slope and water flowing generally south (Parker et al. 1955). The center portions of WCA-1 would have originally been topogenous (Fig. 2.1), eventually evolving into the ombrotrophic system it is today (Fig. 2.10b). It is important to realize that the dominant water sources for various parts of the Everglades naturally evolved but have also been greatly altered by the vast system of canals and dikes that have been installed since the early 1900s. Today most of the Everglades would be loosely classified as soligenous, but in reality it is totally a managed system and should be reclassified as managenous (managed water flow) (Fig. 2.10b).

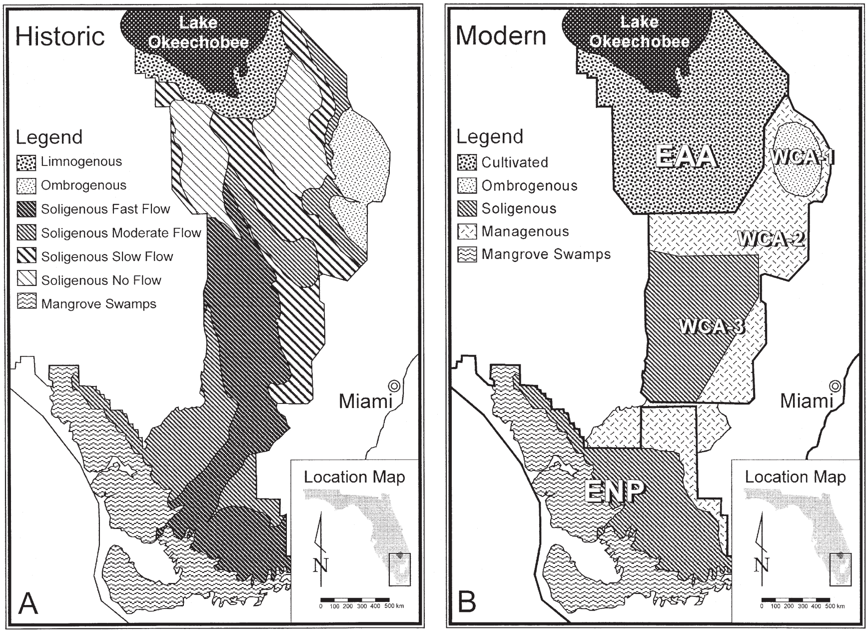

Fig. 2.10 A comparison of hydrologic peatland classifications in the historic Everglades with the present-day Everglades. Shown are the general categories of ombrotrophy and hydrologic flow, systems that control types of peatland formation

The Everglades may also be classified based on nutrient gradients. The terms oligotrophic, mesotrophic, and eutrophic have been adapted from limnology and used to further explain gradients of increasing productivity due to increasing nutrient availability, especially N and P. The oligotrophic class is somewhat broader than ombrotrophic and includes minerotrophic sites with low pH and underlain by nutrient-poor sandy soils as found in pocosins in North Carolina (Bridgham and Richardson 1993). However, there are sites like the Everglades that would be classified as oligotrophic (low productivity) in minerotrophic conditions with very high pH and Ca content, because P has become limiting due to adsorption to Ca and high N-to-P ratios (Richardson and Vaithiyanathan 1995; Richardson et al. 1999; see Chap. 6).

These differences in water source and nutrient conditions for the Everglades peatland (Fig. 2.10a,b) show that an understanding of the hydrologic equivalence concept proposed by Bedford (1996) will be essential to any restoration effort. The hydrologic equivalence concept proposes that hydrological conditions similar to those of the original ecosystem type must be restored on the landscape to restore equivalent ecosystem functions. Landscape hydrologic equivalence will have to be considered in the restoration of the modern Everglades if we ever hope to maintain the diversity of Everglades habitats and communities. For example, care must be taken to maintain the ombrogenous portions of the Everglades like WCA-1A, reestablish limnogenous peatlands south of Lake Okeechobee, recreate conditions for soligenous peatlands where topographically possible, and reduce managenous water flow conditions (water pumping and release across narrow outlets) within portion of the Glades. Unfortunately, hydrologic equivalence concepts for peatlands have not been considered in the current restoration plans (CERP 1999).

2.6 Factors Controlling Succession in Everglades Communities

2.6.1 Ranking of Factors

Historically, the primary factor controlling the long-term development of Everglades plant communities is climate. The amount and seasonal distribution of water from year to year controlled the hydrologic dynamics of the fen system. The hydrologic conditions in turn controlled the fire patterns. The native seed bank was responsible for the regeneration of endemic plant communities once they were disturbed or altered (van der Valk and Rosburg 1997). Massive landscape development in the past 100 years has resulted in regulated hydroperiods (i.e., the number of days that the Everglades ecosystem has standing water at or near the surface) and altered hydropatterns (the distribution of water within the wetland as noted earlier), which in turn have changed fire frequency patterns and fire intensity. Increased P and N loadings from agriculture and urban runoff and introduced exotic species in the early 1900s have all significantly affected plant and animal communities of the Everglades and the WCAs of today (Craft and Richardson 1993b; Davis and Ogden 1994b; DeBusk et al. 1994; Qualls and Richardson 1995; Vaithiyanathan and Richardson 1997a). Importantly, it has been demonstrated by numerous studies that P is the limiting plant nutrient in the Everglades (Steward and Ornes 1983; Koch and Reddy 1992; Richardson and Vaithiyanathan 1995; Craft and Richardson 1997; Richardson et al. 1999; Noe et al. 2001). Thus, the increase in P concentrations as a result of agricultural runoff may have a dramatic effect on local plant communities in addition to the hydrologic changes associated with water management efforts.

The main difficulty for ecologists is in separating the influence of primary climate-driven factors like rainfall, hydroperiod, and fire from the secondary human factors of drainage and flooding, nutrient additions, site disturbance, and exotic species invasions. Moreover, the influence of anthropogenic inputs of nutrients and water varies greatly in each portion of the Everglades, depending on proximity to canal input structures, mode of delivery (i.e., point or nonpoint source) and whether water delivery is seasonally pulsed or continuously released. In other regions like WCA-3A, vast stands of exotic species, such as *M. quinquenervia* and *S. terebinthifolius* (Brazilian pepper) provide a seed source for the ever-increasing spread of these species, although intensive and expensive control measures are underway by state and federal agencies.

To better understand the current status of the main factors controlling the Everglades plant communities, we present a scaled model of impacts (Fig. 2.11). The general scaling is only given as a basis to understand what is now affecting the

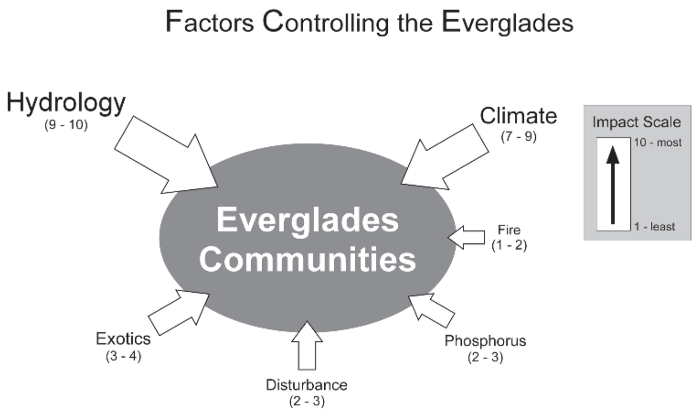

Fig. 2.11 Primary factors controlling Everglades wetland plant community succession. The size of the *arrows* depicts the magnitude of impact. The percentage of the Everglades area that is impacted by each factor is based on area surveys in 2000 and was used to scale the factors

overall Everglades since specific areas of the Glades may be influenced to a larger or smaller degree by any particular factor like exotic species or nutrients. The scaling for each factor is based on the total area that is currently affected by hydrologic management, exotic species, nutrient inputs, etc. (Chaps. 6, 9, and 24). The current status or impact of each factor is scaled from 0 to 10 and indicates that the most important factor controlling the vegetation communities at the present time is the altered hydrologic regimes. As Bedford (1996) has noted, hydrologic shifts are the dominant factor responsible for wetland changes.

As a result of the altered drainage patterns that resulted from water management in the Everglades, climate is no longer the dominant factor controlling plant community succession since altered drainage patterns are so dominant throughout the Everglades. Other factors in order of importance are the invasion of exotic species, phosphorus additions, disturbance, and fire. Here disturbance refers to the planned new water structure changes, new water inputs, and Stormwater Treatment Area (STA) releases. One could argue that the scaling of some of these factors should be different, as in the case of fire influences in some years or at some locations. However, our scaling system weights longer overall relative impact factors and shows that current hydrologic management controls everything, including the amount and intensity of fires much more than in the historic past. In addition, fire's effect is now reduced, especially since fire is an event managed in many areas of the Everglades by the State of Florida Fish and Game Commission (Chap. 9). It is true that massive fires still occur in the Everglades, but they are now often caused by the excessive drying out of areas due to the managed shifting of water from one area of the Everglades to another. The constant manipulation and general lowering of Lake Okeechobee water levels and flows as compared to historic conditions also keep parts of the Everglades far drier than in the past while other areas are maintained with waters deeper than historical levels. This indicates that location and the timing of fires in the Everglades have been affected by water management practices.

Some may argue that P is the main factor controlling most areas of the Everglades, but when one examines the amount of area impacted by P enrichment, it is not the case. While P enrichment can have a great impact on plant community structure, the actual area of the Everglades that is affected by P enrichment is not extensive. In fact, less area is impacted by P than that by invading exotic species. Qian and Richardson (Chap. 24) found, for example, that WCA-1, WCA-3, and the ENP have 81, 91, and 94% of their area with soil P concentrations less than 500 mg kg^{-1}, a value above this indicating enrichment beyond historic levels (Bruland et al. 2006). Further supporting this view of the limited role of P was the recent finding of Bruland et al. (2006) who reported only 263 ha (0.11%) of WCA-3A displayed soils above 500 mg kg^{-1}. Thus, our working hypothesis regarding controlling factors is that the restoration of the Everglades community complex is dependent primarily on the creation of natural hydroperiods and hydropatterns that must include periods of drought and fire. These conditions must be based on the ecological requirements of the dominant species of each community on the landscape. Of course, the reduction of P input to historic levels is critical to the restoration effort, as is the removal of the exotic species.

2.6.2 Succession

Succession in the Everglades has been summarized by Gunderson and Loftus (1993) (Fig. 2.12). They demonstrate that succession of Everglades communities is influenced mostly by disturbance to the hydrology and, in turn, fire frequency and

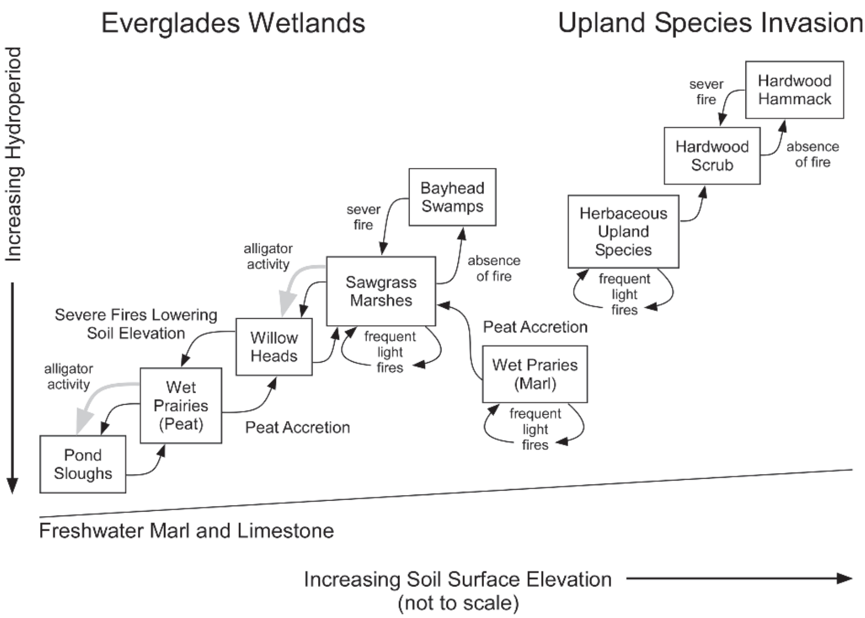

Fig. 2.12 Succession patterns in the Everglades (modified from Richardson 2000)

intensity. Plant communities (Chap. 4) are found along an elevational gradient that translates directly into a hydrologic gradient, which controls fire intensity and frequency. The gradual build-up of marl soil or peat via accretion (1–2 mm $year^{-1}$; Craft and Richardson 1993a) results in the gradual increase in elevation, which changes the hydroperiod for the species. Ponds are the wettest sites, and soil accretion eventually allows them to develop into wet prairie communities, then willow heads and even sawgrass if not severely burned. Frequent light fires have little effect on this successional sequence. Severe fires burn the peat soil and lower the sites, which results in a reversal of this sequence and moves the communities back to wetter habitats. The lack of fire during drought or drainage allows for the invasion of upland macrophytes, scrub, and hardwood species. Alligator activity also acts to change the hydrology and nutrient status of areas and can result in pond development and maintenance (Kushlan 1974, 1987). More recent studies have demonstrated the importance of tree islands in the Everglades and revealed that they are phosphorus "hot spots" on the landscape, i.e., they act as a reservoir of P on the landscape due to the transfer of P from low-concentration surrounding areas by roosting birds and predators (Sklar and van der Valk 2002). The storage and release of high P concentrations from the tree islands have important implications for the ecological successional patterns of the Everglades that are not well understood. We do know that the southern tail ends of tree islands are often areas of higher productivity due to the release of P and that burning of tree islands also releases large amounts of P to downstream areas (C.J. Richardson, unpublished data). The successional dynamics of the Everglades is thus mainly controlled by the interaction of climatic patterns (droughts and rainfall) and human alterations on hydroperiod, which in turn influences fire frequency and the degree of fire intensity as well as the transfer and release of P on the landscape. An understanding of the long-term climatic patterns is thus important to understanding changes within the Everglades.

2.7 Climate and Hydrology

2.7.1 Climate

The subtropical climate of south Florida has hot humid summers, mild winters, and a distinct wet season with 80% of the rainfall falling from mid-May to October (MacVicar and Lin 1984). The Everglades has more in common with tropical climates in that a wet/dry season is probably more important to vegetation composition than winter/summer differences in temperature. Daily temperatures average above 27°C from April to October in the northern part of the Everglades and from March to November in the south. Average daily temperatures are above 10°C even in winter, but freezing temperatures do occasionally occur. The key component of climate controlling vegetation patterns and succession is the amount of precipitation. A 110-year weighted average analysis of annual rainfall over south Florida (1895–2005) shows distinct drought and heavy rainfall periods when compared to the

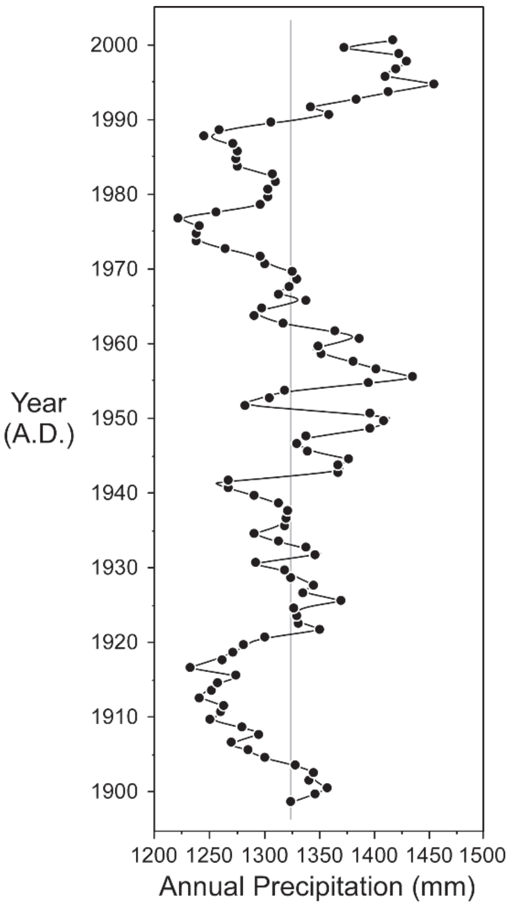

Fig. 2.13 Annual precipitation (nine-point smoothing) in the Florida Everglades based on data from 1895 to 2005. The average rainfall from 1895 to 2005 is shown with a *horizontal line* (data source from NOAA at http://www.ncdc.noaa.gov/onlineprod/drought/xmgr.html)

long-term average annual rainfall of 1,320 mm year^{-1} (Fig. 2.13). The Everglades underwent distinct periods of drought beginning in the early 1900s lasting through the mid-1920s. There was a similar drought period during the 1970s and 1980s up until about 1990. A long-term wet period began in the 1940s and lasted through the 1960s, broken only briefly in the 1950s. The highest period of rainfall, totaling 1,450 mm, was seen in 1995. Importantly, the EAA, which drains partially into WCA-2A, and the ENP, the southern most remnant of the original Glades, have received annual rainfall consistently below the historical rainfall for southern Florida. During the period 1970–1985, the EAA and ENP received less rainfall 80 and 67% of the time, respectively (SFWMD 1992). From the period 1970 to 1989, the EAA received less rainfall than average 14 out of 19 years. Extensive droughts with rainfall 250 mm below normal per year existed for six of those years, and in

1988–1989 they reached 510 mm below normal (SFWMD 1992). However, since the 1990s the Everglades have experienced dramatic increases in rainfall, with highest levels occurring in the mid-1990s (Fig. 2.13). These data indicate that the Everglades has experienced reduced rainfall for extended periods followed by significant rainfall periods that significantly altered the plant-growing environment. These rainfall patterns, when combined with effects of dikes and canal drainage, have resulted in severe drying and flooding of portions of the Everglades with a resultant shift in plant communities as noted in Sect. 2.3. Annual rainfall is the main driver of hydrology, but hurricanes (sustained winds of 120 km h^{-1}) are an important reoccurring event (every 3 years) in south Florida. Hurricanes can produce great wind damage and significant increases in annual rainfall and storm surges (Gunderson and Loftus 1993). Thus, extreme hydrologic events like hurricanes and droughts have had significant effects on the water budgets for south Florida and the Everglades. A severe drought occurs on average every 10 years. El Niño weather patterns result in greater than average rainfall in central and south Florida, while La Niña patterns have the opposite effect (Abtew et al. 2006). These shifts in rainfall patterns have also influenced the yearly nutrient loadings and concentrations (especially P) entering the WCAs (SFWMD 2004, 2005, 2006).

Evapotranspiration (ET) is also an extremely important component of the Everglades. It has been estimated that 70–100% of rainfall exits the Everglades this way (Dohrenwend 1977; Fennema et al. 1994). Evapotranspiration varies across south Florida where lakes, impoundments, and flooded wetlands evaporation losses equal potential ET (Abtew et al. 2006). Higher ET occurs in the southern part of the Everglades compared to areas north of Lake Okeechobee (Fig. 2.14a). However,

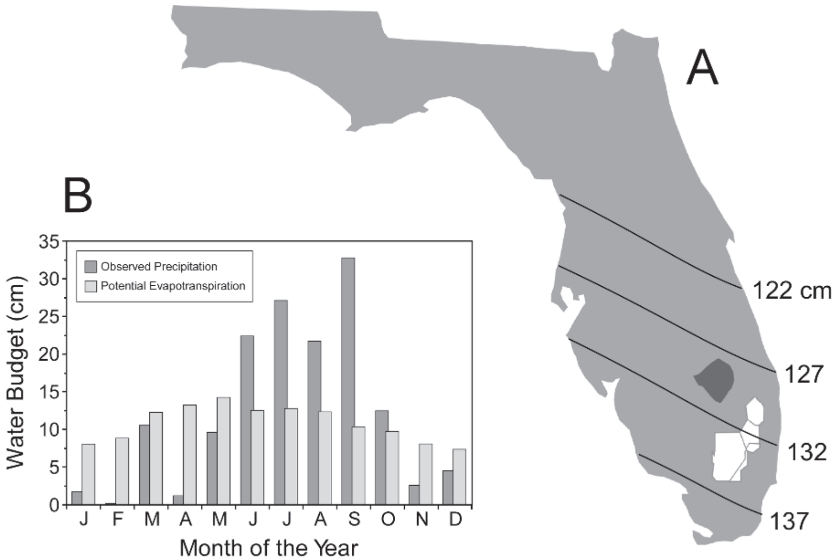

Fig. 2.14 (**a**) Spatial pattern of potential evapotranspiration (PET) isohyetal lines for south Florida-based work of Abtew et al. (2003). (**b**) Monthly average seasonal PET for 2001 in the SFWMD (data from Abtew et al. 2003)

the temporal variation in ET varies little in south Florida compared to annual rainfall variations (Figs. 2.13 and 2.14b) (Abtew et al. 2003). The combination of severe variation in rainfall along with water conservation measures has had a great impact on the Everglades landscape.

2.7.2 Hydrologic Shifts on the Landscape

In 1898, Willoughby reported in his book *Across the Everglades: A Canoe Journey of Exploration* that there were vast amounts of standing water, high dense sawgrass, and numerous upwelling of water from shallow pools in the bedrock. He stated, "All this moving water cannot be accounted for by the rain alone, and the water is too hard for rainwater, so that in all probability more comes from below than above" (Willoughby 1898). This, coupled with the documentation of waterfalls pouring out of the Everglades, upwellings from numerous springs at the edge of the Everglades, and freshwater bubbling up in Biscayne Bay in the early 1900s, clearly indicates an Everglades that maintained a large hydrologic freshwater head on the landscape and originally relied heavily on base flow, a much different hydrology than the one we see today. The role Lake Okeechobee played in supplying water to the Everglades was also not well understood. Historically, lake levels in excess of 20 ft. (6 m) were measured in the lake in the 1850s and as late as the early 1900s, and it was reported that when lake levels exceed 22 ft. (approximately 20.6 ft. NGVD) water would spill over the soil bank on the southern part of the lake into the Everglades (Steinman et al. 2002). Before major alterations and the building of Hoover Dike around the southern part of the lake, Ives (1856) reported in a military survey that at least eight rivers ran directly into the Everglades for 2 or 3 miles (3.2–4.8 km) and disappeared (McCally 1999). Thus, some believed that the periodic overflow of Lake Okeechobee was not the source of water that maintained sheet flow, but rather the rivers that according to early surveys continually supplied the northern Everglades. These discrepancies clearly indicate how poorly we understood the hydrologic relationship of the Kissimmee–Lake Okeechobee–Everglades complex. Moreover, the importance of surface and groundwater interactions in the Everglades was not really appreciated until the USGS report by Parker et al. (1955), who detailed studies on surface and groundwater flows and storage. Parker clearly showed for the first time the complexities of the hydrologic system that controlled the Everglades and that the extensive canal and dike system installed since the early 1900s (Chap. 7) had significantly altered water storage, surface and groundwater interactions, flow of water, and water depths throughout the Everglades.

It is not possible in this chapter to cover in detail the water plans and schedules that have been implemented or proposed over the past 50 years or give a full account of the planned system changes. However, it is possible to present some concept of how highly managed the Everglades hydrology has become, how limited our control over hydrologic flow and storage becomes under extremes of low and high rainfall events (e.g., hurricanes), and how restricted our movement (i.e., newly approved P release criteria) and storage of nutrient-enriched waters have grown to be.

For those wishing more extensive reviews of hydrology and water delivery plans and schedules they are available in Davis and Ogden (1994a), Porter and Porter (2002), the CERP (1999), and updates on the modified delivery schedules at the USACE Website (http://www.saj.usace.army.mil/dp/mwdenp-c111/index.htm).

Water flow in the Everglades was historically extremely slow due to the fact that, with virtually no elevation gradient (30 cm in 8 km) and high vegetational resistances to water flow, water moved sluggishly as surface water flow (0–1 cm s^{-1}) and even slower as groundwater (Rosendahl and Rose 1981; see Chap. 7). Of ecological significance is the impact of major human-made controls over water flow, direction of flows, and hydroperiods, which shifted the system from a soligenous peatland hydrology to a highly managenous (managed) peatland hydrology as noted earlier. For example, during the past 50 years Lake Okeechobee, an integral part of the northern historic Everglades water flow system released nearly 50% of its water into the Atlantic Ocean and Gulf of Mexico due to construction of the St. Lucie and Caloosahatchee canals, with only a fraction of historic flow now going to the ENP and Florida Bay (Fennema et al. 1994). Thus, the entire annual water budget for the Everglades has been greatly altered and the management tightly controlled by the Corps of Engineers mainly for flood control, water supply, freshwater conservation, and wildlife starting with the Corps' comprehensive plan in 1948. This plan was enacted when Congress passed a law (PL 80-858) for a massive central and southern Florida Project for flood control and other purposes (Secretary of the Army 1949).

The implementation of the Corps' 1948 plan and more recent SFWMD plans (1990, 1992, 2006) and Federal plans (CERP 1999) has resulted in a complex and often conflicting array of seasonally and annually revised (often revised during the year due to hurricane or drought conditions, etc.) water schedules for Lake Okeechobee, the WCAs and the ENP. When viewed from solely an engineering perspective, the plans are often considered to be one of the most successful landscape water management schemes in the world because it has kept millions of people dry during the wet season, supplies drinking and agriculture water even during droughts, reduces flood and hurricane impacts, and provides wetlands with water. By 2000, the central and southern Florida (C&SF) Project had over 1,000 miles (1,609 km) of canals, 720 miles (1,158 km) of levees, and approximately 200 water control structure that cover 16 counties and an area of over 18,000 square miles (6,948 km^2) from Orlando to the Florida Reefs (Fig. 8.3). But this maize of structures has had severe negative effects on Everglades ecology (see Plate 1). In simple terms, the water structures and delivery plans did not protect the structural complexity or functions needed to sustain the minerogenous Everglades peatland complex. This is no surprise since they were not designed with this knowledge or purpose. However, CERP restudy plans were designed on the principle of restoring the natural Everglades ecosystem while fulfilling the human water needs (e.g., flood control and a steady supply of water) on the landscape. These multiple goals are not easy to balance considering the year-to-year weather variations and ever increasing human water demands and water quality regulations.

The difficulty of managing this wetland/lake complex starts with Lake Okeechobee, originally a primary source of Everglades water. Prior to 1930, Lake

Okeechobee expanded and contracted depending on rainfall and inflows. After construction on the Hoover Dike was begun on the lake's southern borders in 1938 and completed all around the lake by 1960, water was confined and lake levels and outflows were totally regulated under a series of guidelines and a prescribed regulation schedule. The large littoral and marsh areas that extended north, south, and west of the lake were cut off from water by the dike, thus removing a large nutrient sink for the lake's excess nutrients. The main human-induced threats to the ecological health of Lake Okeechobee are now deemed to be excess nutrient loadings, especially P, altered hydroperiod, and invasion of exotic species (Steinman et al. 2002). Today outflow is highly regulated, as is the lake level under the current WS/E (water supply/environmental) schedule where, for example, on May 31 (start of the wet season) lake water level above 13.5 ft. (4.05 m) would require water release until levels drop below this trigger level. The trigger for releases increases slowly from 13.5 ft. (4.05 m) on May 31 to 15.5 ft. (4.65 m) on September 30 (Steinman et al. 2002). However, recent hurricane events and droughts have greatly altered these schedules, and as recently as 2005 and 2006 there was great controversy over the increased pulsed releases of high P-laden water into the estuaries. Thus, maintaining water levels and standard release schedules are very difficult for Lake Okeechobee water managers, further complicating downstream Everglades water regimes. Further information on the WS/E schedule and regulations, the multitude of SFWMD temporary deviation release schedules, and guidelines for pulsed releases for each zone (ABCD) in the lake can be found at the SFWMD Website (http://www.sfwmd.gov/org/pld/hsm/reg_app/lok_reg/index.html).

The effects of dramatic shifts in the water flow at the landscape scale can be easily appreciated by comparing flows under natural conditions and current water management plans using a Minard-type diagram of the historic surface and groundwater flows based on predictions from the Natural System Model (NSM; SFWMD 1998). In the diagram, the width of the lines shows the amount of water flowing along key points; the direction of flow is shown as well (Tufte 1983). The historical NSM model was run with no canals and dikes in place and then compared with flows measured in the mid-1990s with dikes and flow pumps and gates in operation (Fig. 2.15; Larson 1994). Under historic conditions, a balanced and similar annual volume (~1,481 × 10^6 m^3 or 1,200 k acre-ft. $year^{-1}$) of water was found leaving the Kissimmee Basin flowing into the ENP via the Tamiami Trail. Historically on average only 503 × 10^6 m^3 (408 k acre-ft. 503 × 10^6 m^3) of water left Lake Okeechobee annually because of high ET rates in the lake coupled with restricted flow south due to a natural soil berm, dense sawgrass, and no direct outlets to the Caloosahatchee or St. Lucie Rivers. The central glades had approximately 1.49 × 10^6 m^3 (1,200 k acre-ft.) of water, of which approximately half or 814 × 10^6 m^3 (660 k acre-ft.) exited the Everglades to the Lower East Coast (LEC) yearly (Fig. 2.15). The historical total discharge for the LEC to the Atlantic was estimated by the NSM to be 1,987 × 10^6 m^3 $year^{-1}$ (1,600 k acre-ft.).

By 1994, the annual Everglades water budget was highly regulated, and LEC flows dramatically increased from 1,987 × 10^6 m^3 to 4,579 × 10^6 m^3 (1,611–3,712 k acre-ft.) as freshwater water was being transported to the Atlantic Ocean via a

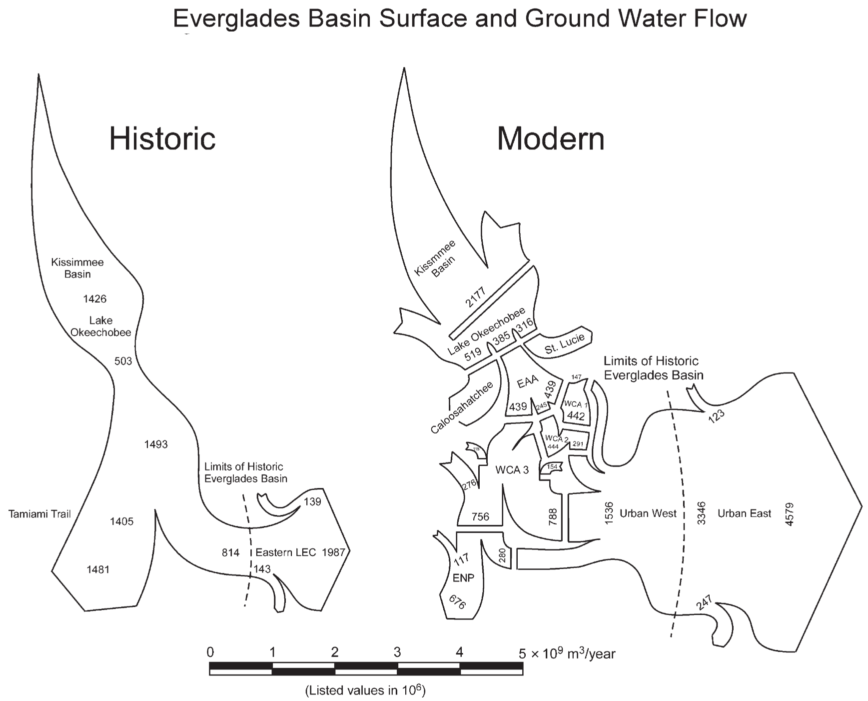

Fig. 2.15 Minard-type graphic of the historic (before 1880) and modern average annual water flows (based on 1993 SFWMD LEC report data and Larson 1994). The line widths are proportional to the volume of the water flows. Values are given as 10^6 m^3 year^{-1}

complex series of canals and pumping stations at the expense of flows into the ENP (Fig. 2.15). Importantly, water inputs into the ENP were less than half of historic inputs, and flows shifted east to the LEC had more than doubled. The transportation of freshwater to the Atlantic Ocean was orchestrated through directives from the Corps and SFWMD to keep the urban LEC from flooding. The Kissimmee basin due to dredging, canals, and extensive agricultural and cattle land development also released 52% more water (1,233 × 10^6 m^3 or 1,765 acre-ft.) into Lake Okeechobee in 1994 than in past centuries; in addition, this greater volume of water was at times extremely high in P concentrations (>500 µg l^{-1}) (Walker 2000). The high concentrations of P in lake water have been a problem during the past 35 years and currently prevent the release of excess water directly to the Everglades due to the current 10 µg l^{-1} TP water criterion approved by the US EPA in 2005 (SFWMD 2006). Lake releases to both the St. Lucie and Caloosahatchee Rivers under the modern flow regime are ~40% of the lake inflow and, when combined with flows to the EAA, reach 56% of the lake of inputs (Fig. 2.15). Inflows in 1994 into WCA-1A, WCA-2A, and WCA-3A were 439 × 10^6 m^3 (356 k acre-ft.), 687 × 10^6 m^3 (557 k acre-ft.), and 1,432 × 10^6 m^3 (1,161 k acre-ft.), respectively. Similar shifts in water budgets in and out of ENP and the WCAs were estimated from 1965 to 1995 under both natural conditions (NSM version 4.5) and under the 1995 base managed conditions (SFWMD version 3.5) by Sklar et al. (2002). They also noted striking differences between natural and managed surface and groundwater flows. Ten years later in 2004, flows into the WCAs were surprisingly comparable to those reported a decade earlier since outflows from Lake Okeechobee were 3,229 × 10^6 m^3 (2,618 k acre-ft.) or nearly three times higher than those reported in 1994, in part due to the effects of numerous hurricanes in late 2004 (SFWMD 2005). Importantly, outflows into the St. Lucie and Caloosahatchee Rivers increased 2.2 times and 3.2 times compared to 1994 while flows to the ENP were similar.

The influence of dramatic increases of water from three major hurricanes (Charley, Frances, and Jeanne) and a remnant of hurricane Ivan that all hit south Florida in the fall of 2004 was still seen in massive alterations to 2005 water management allocations. Input and outputs to Lake Okeechobee were increased 1.5 times above recent values (4,319 × 10^6 m^3 or 3,502 k acre-ft.), and releases were two times (3,556 × 10^6 m^3 or 2,883 k acre-ft.) modern records (SFWMD 2006). As a result, the SFWMD had to release exceptionally high volumes of water, 2,469 × 10^6 m^3 (2,002 k acre-ft.), to the St. Caloosahatchee estuary and higher than normal volumes, 872 × 10^6 m^3 (707 k acre-ft.), to the Lucie estuary. As mentioned earlier, these nutrient-laden freshwater waters continue to plague the estuaries on both coasts, and recent water volume increases have exacerbated the problem. These increased inputs to the Caloosahatchee River have changed estuarine salinity, flows, and nutrient inputs, all of which can affect estuarine fishes, manatees, benthic communities, oysters, and clams as well as submerged aquatic vegetation and the remaining mangrove forests (Barnes 2005). Water inflows into WCA-1A in 2005 were 588 × 10^6 m^3 (477 k acre-ft.), values below recent average levels. However, inflows into WCA-2A were 1,209 × 10^6 m^3 (980 k acre-ft.), nearly double 1994 values and much higher than recent records, while flows into WCA-3A of 1,686 × 10^6 m^3 (1,367 k acre-ft.) were similar to recent values (SFWMD 2006). These reallocations resulted in variations in water conditions in the northern vs. the southern Everglades, but surprisingly the ENP received 65% lower inputs even during this wet period (SFWMD 2006). These differences in water allocations over space and time emphasize the difficulty and challenges in maintaining a balanced flow pattern that will sustain and restore the Everglades and meet human needs.

2.7.3 Future Hydrologic Plans

To help overcome some of the major water management problems of past plans, the central and southern Florida Project Comprehensive Review Study (The Restudy) was undertaken by the Corps of Engineers under the Water Resource Development Acts of 1992 and 1996. The Corps was tasked with developing a comprehensive plan to restore and preserve the south Florida natural ecosystem while enhancing water supplies and maintaining flood protection. Restoration of the Everglades ecosystem was the key purpose of the plan, but as required by law the plan also provided for necessary water-related needs of the region, including urban and agricultural water supply and flood protection. The Restudy Plan emphasized four key problems (1) the reduction in total area of the Everglades ecosystem by 50%; (2) the reduction of water flows to the Everglades by 70%; (3) the deterioration of water quality; and (4) the reduction and damage of natural habitats such that 68 Everglades species were endangered. The plan was to be based on scientific research to correctly time the return of the right quantity and quality of water to each habitat and develop a flexible and adaptive approach that was multiagency/multidisciplinary in nature. Key to the plan was the removal of 400 km of dikes and levees, the construction of new filter wetlands, and the use of hundreds of underground aquifer storage and recovery (ASR) wells over a 20-year period. The benefits were that 80% of the "new or retrieved" water was to be sent to the ecosystem and 94% of predrainage flows returned to the ENP while maintaining flood control and water supply for a sustainable south Florida. The implementation plan was to achieve ecosystem restoration as soon as possible and by 2010 have more than 50% of the hydrologic restoration completed. The overall plan was to take over 30 years at an estimated cost of $7.8 billion, with costs to be shared by the State of Florida and the Federal Government. According to news releases from the Corps in 1999 and the CERP Website (http://www.evergladesplan.org/indes.aspx), sound science and peer review were integral parts of the plan's adaptive management approach of continuously monitoring and making changes when necessary to achieve maximum benefits.

The CERP plan was designed to restore more natural flow to the Everglades complex, and increase water volume to the ENP without drowning tree islands in the northern and central WCAs (Kloor 2000). Highlights of the plan, when implemented, proposed flows and allocations that would result in a 20% reduction per year of LEC losses to the Atlantic Ocean, from $3,641 \times 10^6$ m^3 to $4,578 \times 10^6$ m^3 (3,172–2,953 k acre-ft.) and 442×10^6 m^3 (358 k acre-ft.) of new environmental water allocated to the ENP. Flows of $2,025 \times 10^6$ m^3 year^{-1} (1,642 k acre-ft.) into Lake Okeechobee were projected to be near 1994 levels, but outflows to the Caloosahatchee were doubled from 519×10^6 m^3 to $1,029 \times 10^6$ m^3 year^{-1} (421–834 k acre-ft.). EAA makeup water from the lake in the amount of 203×10^6 m^3 year^{-1} (165 k acre-ft.) was also planned for additions to the WCAs.

Almost immediately, the plan was under attack from environmentalists and scientists who were concerned that too little water was being allocated to the ENP, although under the plan more water is allocated than in the past (Kloor 2000). Another key concern was that moving extra water to the park would come at the expense of the central Everglades ecology. These areas would have to bear the increased flow, which in all likelihood would damage the tree island habitats (Chap. 8) and lead to a loss of key species. Leading the objections were the Miccosukee Tribe, who have over 100,000 ha of holdings in the central Everglades and view the tree islands as key to their hunting and ceremonies. The Miccosukee also worried that the extra water would be laden with excess nutrients (Kloor 2000). A case in point is the EAA makeup water, which is currently too high in nutrients to meet the current standards. Finally, the concept of utilizing several hundred ASR wells storage was highly criticized as an untested method of water storage and retrieval under the geologic and hydrologic conditions of south Florida.

The first step in the CERP process was the Water Resources Development Act of 2000, which was signed into law by President Clinton on 11 December 2000. Ten identified restoration projects with a combined budget of $1,400,000,000 were approved. They ranged from construction of 20,200 ha (50,000 acres) of agricultural area water storage reservoirs to levee seepage management projects in WCA-3A and WCA-3B. However, of key importance was the focus on STA projects around C-9, C-11 impoundment, and Taylor Creek/Nubbin Slough; the raising of the bridge on the eastern portion of the Tamiami Trail; and the filling of the Miami Canal. The C-111 N spreader canal project was to be implemented to improve water distribution and connectivity and sheet flow in the southern Everglades. One hundred million dollars was approved for adaptive assessment and monitoring. Most of these projects were slated to start construction between 2004 and 2005. With an emphasis on delivering more water to the ENP to more closely mimic historic conditions, the USACE devised a modified water delivery (MWD) plan for the Everglades National Park and south Dade Canals (C-111). The plans are outlined in detail on the USACE Website (http://www.saj.usace.army.mil/dp/mwdenp-c111/index.htm) along with development schedules for new projects to the existing central and southern Florida (C&SF) Project. These projects were required to enable water deliveries for the restoration of more natural hydrologic conditions in ENP. These improvements are to enable the reestablishment of the historic Shark River Slough flow-way from WCA-3A through WCA-3B to ENP. However, of major concern in the delivery of water to the ENP is the loss of endangered species habitat for the Cape Sable seaside sparrow (*Ammodramus maritimus mirabilis*), Everglades snail kite (*Rostrhamus sociabilis plumbeus*) and wood stork (*Mycteria americana*). In December 2006, the USACE released its Final Supplemental Environmental Impact Statement (FSEIS) for the Interim Operation Plan (IOP) for the protection of the Cape Sable seaside sparrow (CSSS) and other species, with a recommendation that alternative plan 7R was the best water release schedule according to model predictions of lower impacts for the CSSS, snail kite, wood stork, and their critical habitats (USACE 2006). This alternative also allows for emergency operations under high rainfall conditions. Despite all the management guidelines and the recent alterations to the system, questions still remain on whether water delivery schedules will be adequate to maintain ecosystem integrity of the ENP and maintain populations of the endangered species.

However, an assessment of the water allocated through the 12 structures and S333 to ENP provides us with insight into how water deliveries to the ENP have changed over time, how closely mandated regulation schedules have been followed, the effects of extreme climatic events on delivery schedules, and whether water deliveries can meet restoration needs on an annual basis (Fig. 2.16). Clearly, water management deliveries to northern parts of the ENP have changed dramatically over the past 30 years. In 1970, the U.S. Congress passed a law (PL 91–282) that mandated flow to the park was to be 388×10^6 m^3 year^{-1} (315,000 acre-ft.), with 320×10^6 m^3 (260,000 acre-ft.) allocated for Shark River Slough (Light and Dineen

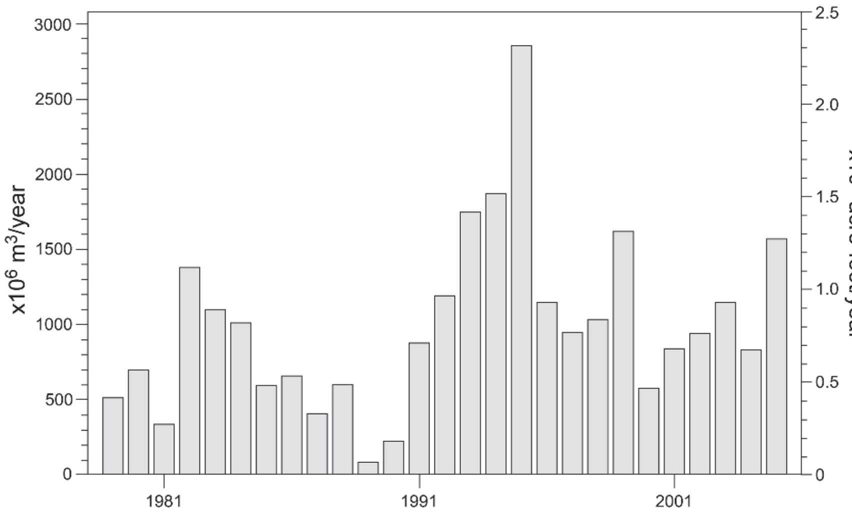

Fig. 2.16 Annual flows of water into the ENP into the northern ENP south of Tamiami Trail and east and west of the L-67 extension via S333 (net to ENP), and the sum of the S12ABCD structures (data from the SFWMD 2006; T.K. MacVicar, personal communication)

1994). As noted earlier, the ENP originally received as much water as the LEC, more than $1,400 \times 10^6$ m^3 $year^{-1}$ (1.1 million acre-ft.) according to the NSM (Fig. 2.15). These volumes are supported by recent simulation runs (NSM46) by the USACE indicating that $1,693 \times 10^6$ m^3 (1,373k acre-ft.) of water flowed into the northern ENP south of Tamiami Trail and east and west of the L-67 extension (USACE, unpublished January 2006 model run using model SFWMM version 5.6).

The first striking feature of the 27-year record is that flows varied greatly from year to year; moreover, flows only exceeded 1 million acre-ft. $year^{-1}$ ($1,233 \times 10^6$ m^3) once from 1978 until 1992 (Fig. 2.16). Flows to the ENP were the lowest in 1989 (0.8×10^6 m^3 or 69k acre-ft.) due to the extensive drought that year. Ironically, this was my first year (senior author) of sampling in the Everglades, and virtually no water passed through the 12 structures. Surface water sampling was impossible, and extensive fires burned throughout the ENP. The highest flows were in 1995 and reached $2,837 \times 10^6$ m^3 (2.3 million acre-ft.) during a year of exceptionally high rainfall (Fig. 2.13). Thus, the ENP has been kept exceptionally dry during some periods due to the lack of water and then drowned in wet years even though a mandated water delivery schedule was in place. Since 2002, the ENP has been receiving water under the alternative 7R schedule of the MWD plan, under which deliveries should average closer to 979×10^6 m^3 $year^{-1}$ (794k acre-ft.) according to the SFWMM model (USACE, unpublished January 2006 modeling). Flows in 2003 were near 1 million acre-ft. (810×10^6 m^3) and in 2005 reached $1,566 \times 10^6$ m^3 (1.27 million acre-ft.). In 2006, an exceedingly dry rainfall year, the park received only 573×10^6 m^3 (465k acre-ft.). Thus, while the shifts in water delivery have been less dramatic under the MWD schedule now in place than earlier delivery schedules, year-to-year variations in rainfall still highly influence release volumes due to a lack of upstream water storage reservoirs. Currently, water continues to be pumped to the ocean and estuaries, and this pumping will continue until water reservoirs are constructed, a plan now under way with the State of Florida's Accelerator-8 funding. Unfortunately, it still appears that lower amounts of water will be delivered to the ENP in drought years due to human and agricultural allocations. These shifts in allocations are of major concern and more recently the USACE has developed under its adaptive management plan a new series of alternative model runs to more closely mimic historic flows. It has for example put forward alternative plan 5 with annual flows in the northern ENP of $1{,}202 \times 10^6\ m^3$ (975 k acre-ft.) (http://www.saj.usace.army.mil/dp/mwdenp-c111/index.htm). Thus, the plans are in place to modify and update the MWDs to sustain the ENP, but whether the water is available each year to meet these guidelines is still controlled to a large degree by climatic conditions and upstream human demands on water allocations.

The correct timing and volumes of future water delivery schedules are not the only aspects of water delivery that need to be restored to maintain the original minerogenous portions of the Everglades peatland complex (limnogenous/soligenous/topogenous zones). Unfortunately, peatland hydrodynamics were not taken into account in the management plans; thus, the normal successional patterns and development of the Everglades fen will forever be altered. In the future, the Everglades will be maintained mostly as a managed or managenous peatland system. With only 50% of the original Everglades remaining and hundreds of control structures in place some say this is the only choice available. Peatland ecologists would argue that we have the opportunity with adaptive management to test alternative peatland restoration techniques and restore key components of the former Everglades. For example, restoration experiments on tree island habitats (SFWMD 2006) are showing some success, but larger scale work is needed on alternate flow regimes and delivery system effects on the peatlands themselves. Hopefully, the ombrogenous interior portion of WCA-1A will be maintained by not allowing surface water flows into the region so that the normal succession stage of marsh–fen–bog development can continue. Finally, a major continuing problem for water managers in the future will be trying to balance conditions to maintain soligenous peatlands conditions for the central Everglades habitats while being pressured continuously to alter hydrologic levels and flows for survival of endangered species at specific locations or for human water needs. By not maintaining the variety of specific hydrologic, nutrient, and fire conditions that shaped the diversity of Everglades habitats, endangered species arguments will continue to mount and ecosystem management will become more and more manipulated to the detriment of the natural community structure and diversity.

2.8 Nutrients: Rainfall and Anthropogenic Inputs

The effects of nutrients additions on plant and animal communities, the development of nutrient gradients, and nutrient storage in plants and soils are covered in detail in the following chapters so no attempt is made to cover these topics in this chapter. Here we provide more of a landscape picture of nutrient additions, primarily P because it is the limiting nutrient. We focus on the two main sources of nutrients, rainfall contributions, and runoff from the surrounding land uses. Unfortunately, little information had been collected on subsurface additions or upflux of nutrients, although we provide nutrient profiles and information on soil nutrient storage and releases in Chap. 6, and water column profiles of nutrients in the Everglades for some ions have been analyzed in the interior of WCA-2A (Vaithiyanathan and Richardson 1997a). Historically, the Everglades received nutrients primarily from rainfall, surface flow, and recycling within the system, especially after fire (Davis 1943; Swift and Nicholas 1987; see Chaps. 6 and 9). It is a P-limited system that has evolved plant species that can survive under TP water concentrations as low as $5\,\mu g\ l^{-1}$ (Koch and Reddy 1992; Richardson et al. 1995; Richardson and Vaithiyanathan 1995; Richardson et al. 1999).

2.8.1 Nutrients in Rainfall and Runoff

There are no reliable long-term historical records of the amount of nutrients in rainfall in south Florida. An analysis of rainfall nutrient loadings over the past few decades suggests that dry fall represents a major source of N and P into the Everglades, but numbers vary greatly in time and space (SFWMD 1992; Peters and Reese 1995). The flow-weighted mean for various stations in south Florida in the 1980s ranged from 20 to $220\,\mu g\ l^{-1}$ for total P with loads of 31–66 mg m^2 $year^{-1}$ (Hendry et al. 1981), but many consider these numbers too high due to collector contamination. There is concern when comparing data from different sources due to differences in sampling. For instance, you may be comparing data where a different type of sampler was used, data where either wet deposition (rainfall events) or dry fall (settling of particles) only was collected, or data where wet and dry fall were combined or collected in bulk samplers. These sampling variations can affect results and conclusions and make it difficult to pool data from different studies.

Cloud deposition, although considered important, is difficult to measure and is often not included. Another factor to recognize is the source of nutrients and whether the measuring systems are close to the ground or near objects that result in recycling of local materials. Because of these difficulties it has been suggested that a P deposition and nutrient monitoring program should focus only on wet deposition values since the chemistry is more certain than dry deposition or bulk samples (Redfield and Urban 1997). Given all these problems, it is not surprising that there is a lack of good data. However, virtually everyone agrees that a better estimate of rainfall P and nutrient input is needed to accurately estimate the nutrient budget for the Everglade. This is especially important for determining nutrient loading to undisturbed or ombrotrophic areas of the Everglades.

In 1997, the Director of the SFWMD, in a conference entitled “Atmospheric Deposition into South Florida,” declared that the atmospheric deposition of nutrients must be considered in all future actions to restore the Everglades (Redfield and Urban 1997). Concerns raised at the conference included problems with methods of collection, sample contamination, best estimates of mass loads of P to the Everglades, the need for establishment of monitoring programs, and how to separate local sources from long distance sources. Few of these problems have been addressed a decade later but some findings help place the importance of rainfall nutrients into perspective with respect to restoration. For example, simulations found significant effects of varying concentrations of rainfall P on water column P concentrations, especially in remote areas (Chen and Fontaine 1997). They reported that when 10 µg l^{-1} of P was in the rainfall the average water column P concentrations ranged from 2 µg l^{-1} in WCA-1A to 57 µg l^{-1} in WCA-2B. By comparison, when rainfall was estimated to be 100 µg l^{-1} WCA-1A and WCA-2B had 18 and 64 µg l^{-1} of P in the water column, respectively. Of concern was the ENP, where water column P increased from 6 to 26 µg l^{-1} under the two simulation scenarios. Raising further alarm was that fact that these concentrations, when used to calculate the loading rates, indicated that a rainfall concentration of 10 µg l^{-1} of P resulted in 883 metric tons of P in the EPA (the Everglades Protection Area includes all WCAs and the ENP and covers of 9,151 km^2), around half of the 1,682 metric tons of P from surface loading via structures. When rainfall P concentrations were assumed to be 30 µg l^{-1} the ratio of P load from rainfall to P load from structures increased by 1.6, meaning that rainfall contributed 60% more P into the EPA than structural inputs and runoff into the Everglades (Chen and Fontaine 1997). Although the concentrations used in the simulations cover too wide a range according to more recent data, they clearly show that rainfall contributes a significant part of the P budget of the Everglades (Redfield and Urban 1997).

To provide a better estimate of P rainfall contributions to the Everglades and overcome the concerns regarding elevated P concentrations in samplers due to contamination from bird droppings, insects, animal parts, and dust, a statistical screening technique was devised by the SFWMD whereby outliers were removed to obtain a better estimate of wet deposition from their samplers by using a Kalman filtering algorithm to provide a minimum error of variance estimation (Ahn 1997). Results from Aerochem wet/dry rainfall collectors at 15 sites run by the SFWMD since 1992 provided a mean P concentration of 10.9 µg l^{-1} with a standard deviation of 13.4 µg l^{-1}. Highest values ranged from 21.7 µg l^{-1} at S-127 to 5.6 µg l^{-1} at L-67A stations (Ahn 1997). When combined with the SFWMD District's average rainfall of 1,346 mm of rainfall the estimated annual P load was 14.8 mg m^2 $year^{-1}$. These values are close to the concentration and load values (15.4 µg l^{-1} and 14.8 mg m^2 $year^{-1}$) used by Walker (2000) in his modeling of rainfall P inputs into Lake Okeechobee. More recent models for determining P dynamics in the STAs have used P concentration and load values of 10 µg l^{-1} and 20 mg m^2 $year^{-1}$, respectively (Walker and Kadlec 2006). These P loading rates are close to average reported bulk sampler values of 23.8 mg m^2 $year^{-1}$ for 17 forested ecosystems in the U.S. and Europe (Likens and Bormann 1985) and 15 lake site values of 27.8 mg m^2 $year^{-1}$ in Michigan (Eisenreich et al. 1977) but much lower than wet/dry bucket values reported earlier for Florida by Hendry et al. (1981). A more recent detailed study in Japan, however, suggests that P inputs (wet and dry) dropped by nearly half from 25.3 to 13.1 mg m^2 $year^{-1}$ when local contributions were excluded (Tsukuda et al. 2006). In calculating the total P loadings for the Everglades EPA in 2004 the SFWMD used P loading values that ranged from 20 mg m^2 year^{-1} in the ENP to 35 mg m^2 year^{-1} values in WCA-1A and WCA-2A (SFWMD 2005). These range of values resulted in an annual atmospheric P loading of 193 MT, or 63.5% of the total loading to the EPA.

To obtain a more accurate picture of the major nutrients and ion rainfall inputs into the Everglades at our P dosing sites (Fig. 1.2 and Chaps. 14 and 15), we monitored nutrient input in Aerochem wet/dry rainfall collectors from September 1993 until September 1997 in two oligotrophic slough areas in WCA-2A (Richardson and Vaithiyanathan 1997a). Sample buckets were sampled and acid washed weekly, and wire bird guards and gauze were used to reduce contamination from sources besides atmospheric deposition (e.g., bird droppings, insects). Samples were carefully screened in the field for any signs of contamination, and any contaminated samples were discarded. In addition, a statistical screening for outliers (>2 standard deviations) was used to remove additional samples before analysis. The geometric mean total P concentration in rainfall from 1993 to 1997 (n = 149 sampling dates) was 10.5 µg l^{-1}, almost identical values to those reported by Ahn (1997) and used in recent modeling efforts. Orthophosphate values averaged 4.6 µg l^{-1} during 1993–1995 and 1997. Our results for P deposition showed strong seasonal and interannual variations, which reflect inputs from local fires, large-scale burning of sugarcane fields to the north, as well as seasonal and annual differences in rainfall patterns. For example, for some unknown reason (i.e., lab and field blanks were all within lab standards) P in rainfall was significantly higher in 1996 than all the other years we measured.

Our annual wet rainfall geometric mean P deposition from 1993 until 1997 was 23.6 mg m^2 year^{-1}, which was similar to values measured at various locations in south Florida and around the globe, which suggests that our value is a realistic estimate of Everglades P loadings (Table 2.1). Another way to determine if these P loading values are realistic was to compare measured atmospheric P deposition flux with rates of P accumulation in peat estimated from ^{137}Cs (~ last 30-year period of accumulation) and ^{210}Pb (~ last 125-year period of accumulation) dating of soil cores (Craft and Richardson 1993a; see Chap. 3) taken in ombrotrophic areas of the Everglades (Table 2.1). The center of the Loxahatchee refuge is a topographically high area, and it only receives nutrients from rainfall (ombrotrophic); thus, P accumulation rates should closely follow precipitation inputs. Our rainfall P loadings of 23.6 mg m^2 year^{-1} are close to the long-term P storage measured in the center of the ombrotrophic Loxahatchee Wildlife Refuge (WCA-1A) during the past 100 years (30 mg m^2 year^{-1}) but are double the 10 mg m^2 year^{-1} accumulation rates found in the past 30 years. However, the more recent estimate of P accretion rates with the ^{137}Cs technique is more susceptible to error due to the difficulty in measuring such low peat accretion rates accurately. In general, both estimates clearly show a low atmospheric loading rate for P, although dry deposition was not included.

The ENP, an area receiving the lowest levels of P runoff input due to P filtration by wetlands to the north and much reduced flow of water into the Everglades during the past few decades, displays three times the accumulation rate (30 mg m^2 year^{-1}) of P in the past 30 years than the Loxahatchee as a result of both input sources (runoff and rainfall). Interestingly, accumulation rates over the past 100-plus years are three times higher in the ENP than more recent estimates of inputs. This probably

Table 2.1 A comparison of annual atmospheric P flux density deposition rates for short-term and long-term P storage in Everglades soils based on ^{137}Cs and ^{210}Pb dating techniques, respectively (see Chap. 3)

Location	Reference	P deposition rates (mg m^2 $year^{-1}$)
WCA-2A	This study (1993–1997)	23.6
Entire Everglades	Ranges used in models for SFWMD (see citations in text)	14.8–35
U.S. forests	Likens and Borman (1985)	24
Czech Republic	Kopacek et al. (1997)	15–24[a]
Location	Reference	P accumulation rates (mg m^2 $year^{-1}$)
WCA-2A (unenriched)[b,c]	This study	60[b]–80[c]
WCA-3A (unenriched)[b,c]	This study	60[b]–80[c]
WCA-1 (ombrotrophic)[b,c]	This study	10[b]–30[c]
ENP[b,c]	This study	30[b]–90[c]
WCA-2A (enriched)	This study	460[b]

[a]Two locations for 16 years
[b]Cs-137 (Craft and Richardson 1998)
[c]Pb-210 (see Chap. 3)

reflects the much larger volume of water with low nutrient concentrations historically entering the ENP in the last century (Fig. 2.15) compared to current conditions; even given the low P concentrations, higher water volume would result in higher mass P loadings. The so-called unenriched areas of WCA-2A and WCA-3A show a doubling of P inputs compared to the ENP during the past 30 years due to increased P in runoff from agriculture (Table 2.1). Values for these areas 100 years ago are closer to the ENP accumulation rates, which suggest a more uniform rate of P input in the past across the Everglades and a lack of agriculture input to the north. By contrast the eutrophic areas in the WCAs are clearly dominated by runoff inflow where areas just south of input structures often have over 4,000 mg m^2 $year^{-1}$ of P coming through the gates as in WCA-2A in the north (Richardson and Qian 1999) and accumulate on average 460 mg m^2 $year^{-1}$ of P, thus potentially releasing over 3,500 mg m^2 $year^{-1}$ of P downstream (Table 2.1). Here rainfall contributes less than 1% of the P load to eutrophic areas. Thus, a massive reduction of nutrient mass loadings in runoff into the Everglades is needed to recreate historic conditions and reduce impacts to plant and animal communities.

2.8.2 *Nutrients in Runoff*

Agricultural runoff from the EAA and Lake Okeechobee both contribute water with significantly higher concentrations of N and P than is typically found in rainfall and in the Everglades (Craft and Richardson 1993b; Davis and Ogden 1994a; Walker

2000). A landscape analysis of the P gradient for south Florida showed that the dairy and cattle regions northeast of Lake Okeechobee had by far the highest total P input concentrations between 1973 and 1999, and the P load averaged 498 MT per year. The lake P concentration increased from ~40 μg l^{-1} P in 1973 to ~100 μg l^{-1} P by 1999 (Walker 2000). The average P concentration in water leaving the EAA farmland in the early 1990s was 150 μg l^{-1} P and was reduced to 115 μg l^{-1} P in the canals and edges of WCA-1 (SFWMD 1992). Water flowing out of WCA-2A into WCA-3A often contained 40 μg l^{-1} P. By the time surface waters reached the structures above the ENP, concentrations were 10 μg l^{-1} P.

A trend analysis of inflow, interior, and outflow P concentrations over the past 27 years reveals interesting patterns of higher P inputs into the northern WCAs and much lower P inputs into the ENP (Fig. 2.17). Over 70 μg l^{-1} P flowed into WCA-1

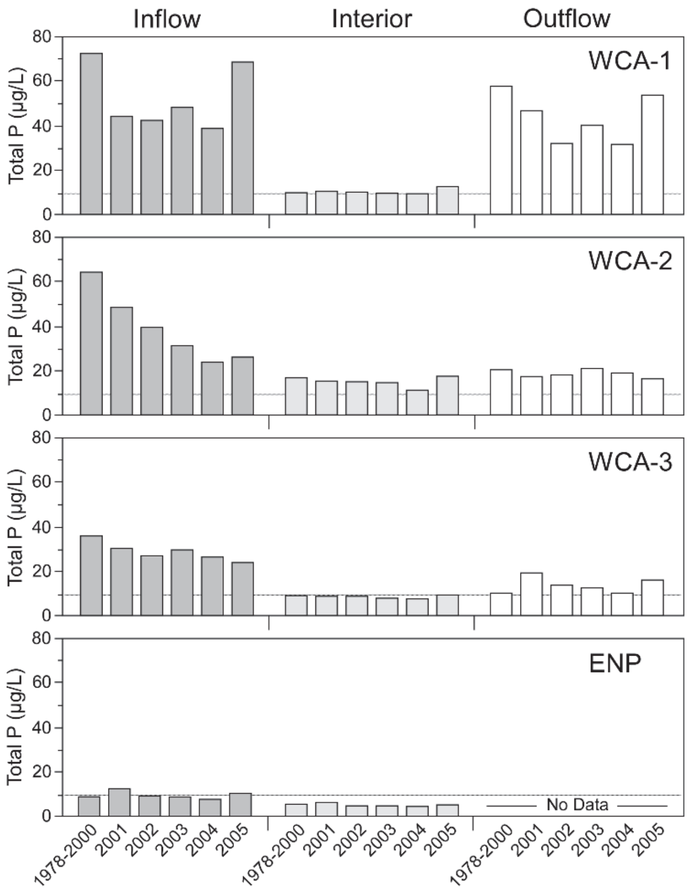

Fig. 2.17 A comparison of phosphorus input, interior, and outflow concentrations from 1978 to 2005 for WCA-1, WCA-2A, WCA-3A, and the ENP. All values are the annual geometric mean of total phosphorus (data from the SFWMD 2003, 2004, 2005, 2006)

on average from 1978 until 2000, with some inputs reaching maximum concentrations over 1,400 μg l^{-1} P (SFWMD 2006). From 2001 until 2004, inflow P concentrations decreased significantly in both WCA-1 and WCA-2A. In 2005, input concentrations increased in WCA-1 nearly back to values prior to any best management practice (BMP) implementations. The SFWMD attributes this rise to excess rainfall and runoff due to high hurricane activity in late 2004 and 2005 (SFWMD 2006). Inputs of P into WCA-3A from 1978 until 2000 averaged 36 μg l^{-1} P and had dropped to 24 μg l^{-1} P by 2005. Low-level mean P inputs (8.9 μg l^{-1} P) into the ENP were recorded over the period 1978–2000, and values rose very slightly in 2005 to 9.1 μg l^{-1} P. Interior P values were lowest in WCA-3A and ENP throughout the study period, averaging around 10 and 5 μg l^{-1} P, respectively. The SFWMD (2006) reported that interior P concentrations rose during low rainfall periods and droughts and were diluted during the wet season in all areas. WCA-1 had interior values that averaged 10 μg l^{-1} P from 1978 until 2000 and then rose in 2005 to 12.1 μg l^{-1} P. However, interior WCA-2A values were always the highest, averaging 17.1 μg l^{-1} P from 1978 to 2000 then rising slightly to 17.9 μg l^{-1} P in 2005. The rises in 2005 P interior values was attributed to periods of excessive rainfall and periods of soil P release during drought conditions (SFWMD 2006). However, the fact that WCA-2A interior site P concentrations have virtually remained the same even though inputs have been reduced by >50% may be due to the reflux of the large residual amount of resident P in soil at this site (Reddy and Rao 1983; see Chaps. 3 and 6).

According to the SFWMD (2006) report, approximately 85% of the P samples collected in 2005 in the entire EPA (all sites and areas) had values below 50 μg l^{-1} P, 51% were below 15 μg l^{-1} P, and 32% were at or below 10 μg l^{-1} P. These data indicate that the interior sites in the ENP and WCA-3A meet the 5-year geometric mean criterion of less than or equal to 10 μg l^{-1} P across all sites in three of five years, are annually less than or equal to 11 μg l^{-1} P across all stations, and are less than or equal to 15 μg l^{-1} P annually at all individual stations (SFWMD 2006). However, current inflow P concentrations into WCA-1, WCA-2A, and WCA-3A are still far in excess of the approved P criterion, and interior values of WCA-2A have not changed (Fig. 2.17). More troubling are the high concentrations of P that are still flowing out of the WCAs and toward the ENP. The northern WCAs release far higher P concentrations than WCA-3A, but all values are well above the USEPA-approved P criterion even though farm BMPs have been in place for over a decade and STAs are now in operation.

Of importance but often not addressed is the amount of SRP or orthophosphate (i.e., considered more readily available for uptake by organism) vs. TP in input, interior, and outflow areas. From 1978 to 2003, SRP comprised 34, 31, 26, and 29% of TP input concentrations into WCA-1, WCA-2A, WCA-3A, and ENP, respectively (SFWMD 2006). Inflow SRP concentrations compared to outflow concentrations showed considerable reductions in values for some of the WCAs (23–17 μg l^{-1} SRP in WCA-1, 18–5.2 μg l^{-1} SRP in WCA-2A, and 9–2.8 μg l^{-1} SRP in WCA-3A) but showed little change in the SRP to TP outflow ratios which remained at 31, 25, and 25%, for WCA-1, WCA-2A, and WCA-3A, respectively. Interior concentrations of SRP were quite low and averaged 1.6 μg l^{-1} SRP in WCA-1, 3.7 μg l^{-1} SRP in

WCA-2A, 1.7 μg l^{-1} SRP in WCA-3A, and 2.7 μg l^{-1} SRP in ENP over the 1978–2003 period.

Nitrogen, another important nutrient in the Everglades, often receives less interest since P is the limiting nutrient. Everglades soils often have more than 3% N by weight, and water values of NH_4–N are very high (Chaps. 3 and 6). However, a brief assessment of the total nitrogen (assessed as Kjeldahl N) shows a similar north-to-south gradient as P with higher input concentrations in the north and values decreasing in the south. Average annual long-term input concentrations (1978–2003) ranged from 3.4 mg l^{-1} going into WCA-1 to 1.2 mg l^{-1} flowing into the ENP (SFWMD 2006). Interior concentrations also followed a north-to-south gradient with long-term values averaging 1.6, 2.4, 1.5, and 1.3 mg l^{-1} in WCA-1, WCA-2A, WCA-3A, and ENP, respectively. Of more concern than the N concentrations alone are the shifts in N:P ratios in the various components of the Everglades because high ratios of this key index indicate more severe P limitations in the Everglades (Richardson et al. 1999). For example, inflow N:P ratios to WCA-1, WCA-2A, WCA-3A, and ENP averaged over the 1978–2003 period were 51, 52, 59, and 141, respectively, and were quite different from 2005 inflow ratios values of 32, 92, 70, and 108 for each respective area. These data suggest that shifts in the N:P ratio are taking place due to the upstream BMPs on farmland as well as the use of the STAs to remove P. Long-term, these shifts should improve and maintain P-limiting conditions in the interior of the fens. Interior N:P ratios for WCA-1, WCA-2A, WCA-3A, and ENP averaged over 1978–2003 were 119, 130, 160, and 257, respectively (SFWMD 2006). These general trends remained the same in 2005, indicating that P was extremely limiting in all interior locations, but especially in ENP.

Phosphorus loadings from the EAA have been implicated in the replacement of sawgrass by cattail in WCA-1 (Loxahatchee National Wildlife Refuge) and WCA-2A in the early 1980s (Toth 1987, 1988; Belanger et al. 1989; Urban et al. 1993). Belanger et al. (1989) asserted that additions of nutrient-enriched water to WCA-2A have contributed to the invasion of a monotypic cattail community. The high P levels in vegetation, soils, and surface waters of the cattail-dominated areas of WCA-2A suggest that P may be primarily responsible for the invasion of cattails in WCA-2A (Belanger et al. 1989; Richardson and Craft 1993; Vaithiyanathan et al. 1995, 1997; Craft and Richardson 1997; see Chap. 9). Thus, the control of cattail expansion and community shifts will require an understanding of the effectiveness of the Everglades BMP regulatory P reduction program since its implementation in 1996 (Fig. 2.18). Over 1,600 MT of P have been prevented from entering the EPA since 1996 and the predicted P load has been reduced from the baseline period (1978–1988, prior to any treatments or BMPs) average annual loading of 444 MT and P concentrations 173 μg l^{-1} P to 182 MT and 124 μg l^{-1} P by 2005 (SFWMD 2006). Of note is the sharp reduction in mass P loadings since full implementation of the BMPs in 1996, except for 2000 and 2005. These increases have been attributed to rainfall and weather events by the SFWMD but they were far below the predicted loads of over 400 MT in those years. Thus the BMP program has had a dramatic effect on reductions of P going into the EPA, but loads are still variable from year to year and will be greatly influenced by the P removal effectiveness of the STAs.

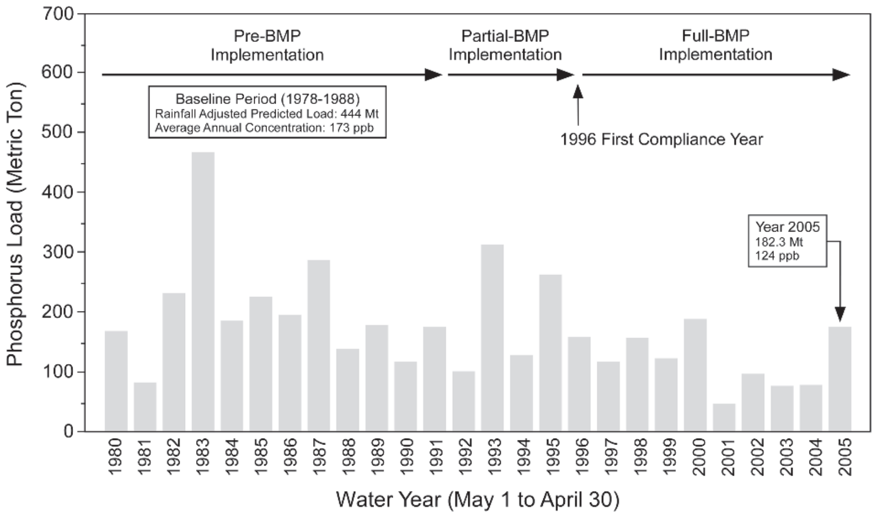

Fig. 2.18 Total phosphorus loads from the Everglades Agricultural Area (EAA) to the Everglades Protection Area (EPA) from 1980 until 2005. Best management practices (BMPs) are denoted as well as the first compliance year (data from SFWMD 2006)

The major hope for reducing P loads into the EPA in the future was the use of STAs to treat EAA, upstream and Lake Okeechobee waters prior to their release. The design criteria for these STAs were carefully planned and modeled to determine the correct sizing of the STAs by scientists at the SFWMD with assistance of engineering experts Robert Kadlec and William Walker (Walker 1995; Walker and Kadlec 2006). To date six STAs covering over 41,000 acres (16,564 ha) have been built, the earliest in operation since 1994–1995. They were designed initially around a 1.3–1.5 g m^2 $year^{-1}$ loading rate (Table 2.2). Since their inception they have been estimated to reduced P loadings by 617 MT into the EPA, and in 2005 all the STAs combined removed 189 MT of P, or 71% of the 268 MT of loadings (SFWMD 2006). The flow-weighted mean inflow ranged from 247 μg l^{-1} P in STA-1W to 78 μg l^{-1} P in STA-6 in 2005. Overall mean inflow P averaged 147 μg l^{-1} P and outflow concentration 41 μg l^{-1} P in 2005. Unfortunately, the actual loading rates to some of the STAs during operation have greatly exceeded their design loading criteria by more than a factor of 2, and outflow P concentrations have also greatly exceeded desired levels (Table 2.2). In fact, in 2005 and 2006 four of the STAs exceeded 50 μg l^{-1} P, and several had output values over 100 μg l^{-1} P. Importantly, only three of the STAs (STA-2, STA-34, and STA-6) were loaded with P near their design criteria, and only these sites came close to outflow concentration of 20 μg l^{-1} P over the entire period. Moreover, two STAs (STA-2 and STA-6) doubled their concentration output in 2005, which was again attributed to high rainfall and runoff inputs (Table 2.2).

In terms of P reductions, both the BMPs and the STAs have resulted in a significant decrease of P to the EPA. However, EAA outflow P concentrations continue to

Table 2.2 Summary of Stormwater Treatment Areas (STAs), inflow loads, and outflow P concentrations in 2006 (Walker and Kadlec 2006)

	Inflow P load (g m^2 year^{-1})				Outflow P (μg l^{-1})		Operating period	
STA	Design ECP	Design LTP	Operation period	Last 12 months	Operation period	Last 12 months	First	Last
STA-1W	1.4	1.0	2.7	1.7	66	125	July-01	July-06
STA-1E	1.4	1.1	0.6	0.8	174	122	May-04	July-06
STA-2	1.3	1.1	1.3	1.8	19	28	July-01	July-06
STA-34	1.3	1.1	1.2	1.5	19	24	Dec-03	May-06
STA-5	1.5	1.1	2.7	2.5	99	100	May-01	June-06
STA-6	1.3	1.1	1.4	1.6	21	26	Aug-00	June-06
All	1.3	1.1	1.5	1.6	43	48	Aug-00	June-06

Design loading criteria are shown for each the STAs between 1.0 and 1.5 g m^2 year^{-1}, but they have been exceeded in five of the six treatment areas in the last 12 months and on average for all sites

remain high, and STA reductions have not consistently reached the low concentrations 10–15 μg l^{-1} P that had been hoped for by many scientists. While P mass loadings are significantly reduced by more than 50% for the EAA, even in wet years like 2005, P concentrations remain too high for major improvements in the receiving waters as noted in Fig. 2.17. If the present trend continues and no additional STAs are built, the Everglades will continue to receive unacceptable concentrations and loads of P for the foreseeable future. This will have significant consequences for the native biota and ecosystem structure and function as explored in following chapters.

2.9 Conclusions and Lessons for Restoration

One of the gaps in our knowledge concerns how the diversity of communities was formed in the Everglades landscape complex. Paleoecological studies indicate a north-to-south trend in peat accumulation over the past 5,000 years, with considerable periods of longer and shorter hydroperiods long before drainage canals altered hydrologic flow and water levels. Numerous changes in vegetation have occurred over the past few thousand years, but recent alterations have resulted in an invasion of upland and exotic species as well as massive increase in cattail due to increased P mass loadings. The most dramatic changes have taken place in the twentieth century, which has resulted in loss of slough areas and tree islands, as well as extensive cattail invasions in the northern Everglades. A loss of acidophilic diatoms in recent times in WCA-2A indicate that this area was probably more acidic and had fewer calcareous periphyton mats than are present today.

The Everglades should be classified as a fen or, in more generic terms, a peatland – not a marsh or swamp. The hydrogeologic factors controlling the formation of fens are totally different from that of marshes and swamps, and this has important hydrologic consequences for restoration of the Everglades. Restoring landscape hydrologic equivalence will be essential to successful restoration of the modern Everglades if we ever hope to maintain the diversity of Everglades habitats and communities. For example, care must be taken to maintain the ombrogenous portions of the Everglades like WCA-1, reestablish limnogenous peatlands south of Lake Okeechobee, recreate conditions for soligenous peatlands where topographically possible, and reduce managenous water flow conditions (water pumping and release across narrow outlets) within portions of the Everglades.

Succession in the Everglades is mostly controlled by hydrologic conditions and, in turn, fire frequency and intensity. The impact of excessive nutrients, especially P, is critical in some areas of the Everglades as is the invasion of exotic species. Historic hydrologic flow patterns and volumes of water have been greatly altered as evidenced by the reductions in flow to the ENP and increased flow of freshwater to the oceans and Gulf of Mexico. We have only recently begun to understand the hydrologic relationship of the Kissimmee–Lake Okeechobee–Everglades complex. Moreover, the importance of surface and groundwater interactions in the Everglades is still not fully understood, and the influence of canals and pumping stations on community responses are for the most part unknown. The difficulty of managing the wetland/lake complex starts with the myriad of Lake Okeechobee regulation schedules and farm-use plans for water, which has a ripple effect on all downstream water conditions in the Everglades. The CERP plan was designed to restore more natural flow to the Everglades complex and increase water volume to the ENP without drowning tree islands in the northern and central WCAs, but difficulties abound in meeting regulated flow conditions when droughts or extreme wet seasons occur.

The shifts in water delivery have been less dramatic under the modified water delivery schedule now in place than with earlier delivery schedules but year-to-year variations in rainfall still highly influence release volumes due to a lack of upstream water storage reservoirs. Currently, water continues to be pumped to the ocean and estuaries and will continue until water reservoirs are constructed. The implementation of the CERP called for 50% of the hydrologic restorations to be completed by 2010; however, that deadline will not be met. Moreover, it still appears that lower amounts of water will be delivered to ENP in drought years due to human and agricultural allocations. Unfortunately, the correct timing and volumes of future water delivery schedules as well as the mode of delivery needed to restore the original minerogenous Everglades fen and peatland complex (limnogenous/soligenous/topogenous zones) have been ignored, and thus the normal successional patterns and development of the Everglades peatland will forever be altered. In the future the Everglades will be maintained mostly as a managed or managenous peatland system.

Phosphorus in rainfall is a major contributor to the overall P budget for the Everglades but plays a lesser role in areas where there is runoff from agriculture. However, the accuracy of P measurements in rainfall is not adequate at the present time, thus estimates of total P loadings to the Everglades are in question. Research to improve estimates is badly needed.

Data on nutrient concentrations and mass loadings for N and P are readily available and show good long-term trends of decreased inputs from agriculture and adjacent areas due to farmland BMP reductions and STA uptake and storage. Of importance but often not addressed is the amount of SRP or orthophosphate vs. total P in input, interior, and outflow areas. Another area that needs to be monitored carefully is the shifts in N:P ratios in the various components of the Everglades since high ratios of this key index indicate P limitation, a condition representative of the historic Everglades. Recent data suggest that shifts in the N:P ratio are taking place due to the upstream BMPs on farmland as well as the use of the STAs to remove P. Long-term, these shifts should improve and maintain P-limiting conditions in the interior of the fens.

According to the SFWMD (2006) report, the entire EPA in 2005 had approximately 85% of the P samples collected with values below 50 μg l^{-1} P, 51% below 15 μg l^{-1} P, and 32% at or below 10 μg l^{-1} P. These data indicate that the interior sites in the ENP and WCA-3A met the USEPA P criterion; however, current inflow P concentrations into WCA-1A, WCA-2A, and WCA-3A are still far in excess of the approved P criterion, and interior values of WCA-2A have not changed over the past 27 years. In terms of P reductions both the BMPs and the STAs have resulted in a significant reduction of P to the EPA. However, EAA outflow P concentrations continue to remain high, and STA reductions have not consistently reached the low concentrations 10–15 μg l^{-1} P that are needed to sustain the Everglades. While P mass loadings are currently significantly reduced by more than 50% compared to the 1970s and 1980s for the EAA, even in wet years P concentrations remain too high for major improvements in the receiving waters. If the present trend continues and no additional STAs are built to reduce current P overloading, then the Everglades will continue to receive unacceptable concentrations and loads of P for the foreseeable future. This will have significant consequences for the native biota and ecosystem structure and function as noted in a number of the following chapters.