Resilience of Mangroves

on the South Coast of Havana province, Cuba

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de L.B. para J. ☺

… hey Ricardo A. Herrera Peraza (1950-2006)… enough space over there for all my love and scientific admiration for you?... we keep seriously smiling…

ABSTRACT

Mangroves are important worldwide for a wide range of ecosystem services that contribute to human well-being (e.g., food and water consumption, recreation). However, 35% of documented mangrove vegetation disappeared in 1980-2005 mainly due to direct and indirect human impacts. Mangrove resilience typically manifests as regeneration of mangrove vegetation, either naturally or promoted by restoration.

The Gulf of Mexico and the Caribbean region are representative of the worldwide mangrove situation. The thesis addresses five study cases in Cuba, Mexico and USA. The cases are examples illustrating mangrove resilience through natural regeneration and restoration activities. Changes in vegetation, ground altitude and inundation as well as institutional aspects of mangrove restoration are addressed.

Mangroves and wetlands of the south coast of Havana province in Cuba were studied. Across these mangroves and wetlands, a road-like freshwater barrier was built during 1985-1991 to guarantee an adequate freshwater supply for agriculture and settlements, including Havana City. The barrier is 52 km long, and slows down the flow of freshwater into the sea by retaining water landward of the barrier. Besides achieving enhanced groundwater quality and quantity, the barrier caused mangrove dieback through flooding. The assessment of mangrove resilience took advantage of an empirically supported historical perspective. First, remote sensing using satellite images of 1985 and 2001 indicated major land-cover changes related to the construction of the barrier. Second, in plots representative of those changes, vegetation and abiotic factors (e.g., water level and soil redox potential) were surveyed in 2005. The land-cover changes were concentrated landward of the barrier, e.g., a decrease in wetland forests (from 4847 to 1206 ha; mainly plantations) and increase in flooded mangroves (from 11 to 1425 ha). The access provided by the road-like barrier promoted new seaward mangrove plantations (774 ha). As expected from the analyses of the satellite images, in both the dry and the rainy season in 2005, the mean water level was higher in dammed wetlands (16 and 43 cm) than in those located seaward of the barrier (-5 and 7 cm). Also, the landward wetlands had a more negative mean soil redox potential (seasonal extremes were -240 mV and -40 mV). In contrast, the major accumulations of water landward of the barrier (i.e., highest water levels) occurred in the two sectors with the highest number and density of spillways (10 and 14 spillways, and 0.6 and 0.8 spillways per km). Resilience of the mangrove cover to the barrier-induced flooding manifests as early recovery of mangrove vegetation (mangrove cover less than 60%, trees typically smaller than 4 m), and also as advanced recovery leading to a closed forest canopy (trees 4-11 m). Recovered vegetation can progressively enhance the change from permanent to seasonal flooding. The study shows that decreasing the water level towards non-permanent flooding can prevent the establishment of vegetation. Maintaining the spillways of the barrier, however, can enhance the recovery of mangroves. Management interventions are a promising way of supporting the restoration of mangrove covers.

The study proposes a methodological approach, based on qualitative mathematical modelling (loop analysis), for improving the assessment and management of resilience of environmental systems. The approach is presented through empirical data obtained in Cuban mangroves.

Resilienz der Mangroven an der Südküste der Havanna Provinz, Kuba

KURZFASSUNG

Mangroven sind weltweit äußerst wichtig für eine Reihe von Ökosystem-Dienstleistungen, die zum menschlichen Wohlergehen beitragen (z.B. Ernährung, Wasserkonsum und Erholung). Dennoch verschwanden in den letzten zwei Jahrzehnten weltweit 35% der Mangrovenvegetation, hauptsächlich durch direkte und indirekte menschliche Einwirkung. Resilienz von Mangroven zeigt sich in der Regeneration der Mangrovenvegetation, entweder durch natürliche Regeneration oder aber unterstützt durch Rekultivierungsmaßnahmen.

Der Golf von Mexiko und die Karibik sind repräsentativ für den Zustand der Mangroven weltweit. Die Dissertation umfasst fünf Fallstudien in Kuba, Mexiko und den USA. Diese Studien zeigen Beispiele für Mangroven-Resilienz sowohl durch natürliche Regeneration als auch durch Rekultivierung. Änderungen in der Vegetation, der Topographie bzw. dem Ausmaß der Überschwemmungen sowie institutionelle Aspekte der Mangrovenrekultivierung werden untersucht.

In der vorliegenden Studie werden insbesondere die Mangroven und Feuchtgebiete der Südküste der Havanna Provinz in Kuba betrachtet. In diesem Gebiet wurde zwischen 1985 und 1991 eine unterirdische Barriere errichtet, um die Versorgung von Landwirtschaft und Siedlungen, inklusive der Stadt Havanna, mit Frischwasser zu garantieren. Die Barriere ist 52 km lang und verlangsamt den Abfluss von Süßwasser ins Meer, indem sie das Wasser auf der landwärtigen Seite staut. Neben der Erhöhung der Grundwasserstands und der Verbesserung der Wasserqualität hat die Barriere aber auch das Mangrovensterben durch dauerhafte Überflutung verursacht.

Die Beurteilung der Mangroven-Resilienz wird durch eine historische Betrachtung unterstützt. Mit Satellitenbildern aus den Jahren 1985 und 2001 können enorme Änderungen in der Vegetationsdecke nachgewiesen werden, die durch die Barriere verursacht wurden. 2005 wurden auf Parzellen, die diese Veränderungen zeigen, die Vegetation und abiotische Faktoren, wie z.B. der Wasserspiegel und das Redox-Potential des Bodens, untersucht. Die Änderungen in der Vegetationsdecke konzentrieren sich auf die landwärtige Seite der Barriere, z.B. nahmen die Feuchtgebietswälder von 4847 auf 1206 ha ab (vorwiegend Pflanzungen) und die überfluteten Mangroven von 11 auf 1425 ha zu. Durch die Barriere, auf der eine Straße verläuft, ist ein Zugang in das Gebiet entstanden, der zu neuen Mangrovenpflanzungen (774 ha) auf der dem Meer zugewandten Seite führte. Wie die Satellitenbilder zeigen, ist sowohl in der Trocken- als auch in der Regenzeit des Jahres 2005 der durchschnittliche Wasserspiegel in den aufgestauten Feuchtgebieten höher (16 und 43 cm) als in den seewärts gelegenen Gebieten (-5 und 7 cm). Darüber hinaus hatten die landwärts gelegenen Feuchtgebiete im Durchschnitt ein negativeres Redox-Potential des Bodens (die saisonalen Extrema lagen bei -240 mV und -40 mV). Die größten Wasseransammlungen (d.h. der höchste Wasserspiegel) auf der landwärtigen Seite der Barriere wurde in den zwei Sektoren mit der höchsten Anzahl und Dichte an Abflusskanälen festgestellt (10 und 14 Abflusskanäle; 0.6 und 0.8 Abflusskanäle per km). Die Resilienz der Mangrove in den durch die Barriere dauerhaft überfluteten Flächen zeigt sich in der Regeneration der Mangrovenvegetation binnen kurzer Zeit (Deckung 60%; Bäume <4m) und der schnellen Entwicklung einer geschlossenen Kronendecke. Die regenerierten Mangrovenflächen wandeln sich nach und nach von permanent zu saisonal überfluteten Standorten. Die Studie zeigt, dass die Senkung des Wasserspiegels, hin zu nicht dauerhaften Überflutungen, die Entwicklung von Vegetation verhindern kann. Allerdings kann das Aufrechterhalten der Abflusskanäle die Regeneration der Mangroven verbessern. Managementeingriffe sind eine vielversprechende Methode für die Wiederherstellung der Mangrovenvegetation.

Es wird eine qualitative mathematische Modellierung (loop anlysis) vorgeschlagen, mit der die Beurteilung und das Management der Resilienz von Ökosystemen verbessert werden kann. Der methodische Ansatz wird anhand von Daten illustriert, die in den kubanischen Mangrovengebieten erhoben wurden.

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ACKNOWLEDGEMENTS

1 GENERAL INTRODUCTION

Mangrove ecosystems, together with other wetland ecosystems, are important for a wide range of ecosystem services that contribute to human well-being, e.g., food and water consumption, flood regulation, and recreational opportunities (MEA 2005a). However, since the 1980s mangrove degradation has been alarming. According to The Millenium Ecosystem Assessment released in 2005, "estimates of the loss of mangroves from countries with available multiyear data (representing 54% of total mangrove area at present) show that 35% of mangrove forests have disappeared in the last two decades" (MEA 2005a). To maintain the ecosystem services of mangroves, enhanced protection and restoration of these ecosystems is needed. Both protection and restoration are based on the resilience of mangroves, i.e., on their capacity "to remain as mangroves" (Capote-Fuentes and Lewis 2005; Dahdouh-Guebas and Koedam 2006; McLeod and Salm 2006; MEA 2005a).

When the concept of resilience was started in ecology by Holling (1973), the scientific community received it with enthusiasm (Berkes et al. 2003) although it has frequently been the origin of conceptual misunderstanding (Carpenter et al. 2001; Grimm and Wissel 1997). As in the case of other outstanding concepts, resilience has also jumped from the scientific side of ecology and environmental science to the political side (Folke et al. 2002; Golley 1993). Holling's definition takes resilience as "a measure of the persistence of systems and of their ability to absorb change and disturbance and still maintain the same relationships between populations or state variables".

Placing resilience among the most lively discussed concepts in ecology has received a strong input from the inclination of Holling to go "beyond the content of science", i.e., to pay special attention to "the way of doing science". In this regard, promoting formal scientific institutions has played a key role. An example is the launch of the journal Conservation Ecology (Holling 1997), and undoubtedly its turning into the current Ecology and Society (Folke and Gunderson 2004). Holling's coordination of the book Adaptive Environmental Assessment and Management (Holling 1978), and his directorship at the International Institute for Applied Systems Analysis (IIASA) (1981-1984) are outstanding, too.

Other recent concepts have not experienced such successful history, for example that of ecosystem health (Rapport et al. 1998). Currently, the scientific, policy and political strength of the resilience concept is heavily contested by those of vulnerability and adaptive capacity (Thywissen 2006; UNU-EHS 2006). A stable conceptual family including resilience, health, vulnerability, adaptive capacity, risk and disaster could emerge.

Resilience has been the framework in which long-time debated ideas have been re-challenged or simply presented again in the ecological and environmental literature (Gunderson 2000). The most controversial ones include the concepts of complexity, stability and non-linearity (Holling 1986, 2003; Phillips 1999; Pimm 1984; Walker et al. 2006a, b).

Interestingly, two of the strongest features of Holling's definition of resilience are about to, or have already, become the weakest features of the definition. The first is the very general way in which the definition was stated, which makes the definition rather an umbrella definition under which all concrete cases could be addressed. However, that general style of the definition sounded empty for some audiences, especially because clear methodological tools were not provided with the definition. The second feature regards the practical side of the definition, i.e., its suitability to address real-life situations in environmental management. This enters resilience into the discussion on the disciplinary/interdisciplinary and action-oriented components of ecology and environmental science. It widens the audience of the definition and thus the likelihood to link resilience to other important approaches, but also increases the chances for conceptual confusion. The most important confusion may be that related to the concept of stability, specifically in relation to what can be considered stable or not stable, and changing or not changing.

One interesting relation is that between resilience and ecological succession (Gehring et al. 2005). Ecological succession is deeply rooted in Holling's ideas. The succession approach is included in his benchmarking figure-eight diagram with the phases exploitation, conservation, creative destruction and renewal. Discussing that diagram has led to the so called adaptive cycle, a theory that can be used to examine the dynamics and resilience of a socio-ecological system (i.e., an environmental system) by addressing its collapse, reorganization, and recovery phases (Abel et al. 2006; Holling 1978, 1986; Walker et al. 2006a).

The Gulf of Mexico and the Caribbean region are representative of the worldwide mangrove situation. Countries in the region share mangrove plant species, natural influences on mangroves like tropical storms, and most important, major management challenges and opportunities in coastal development. Therefore, mangroves belong to the urgent issues needing enhanced regional scientific cooperation. This study contributes in that direction by addressing five study cases in Cuba, Mexico and United States of America (Capote-Fuentes and Lewis 2005; Doyle et al. 2003; Rivera-Monroy et al. 2004; Snedaker 1995; Suman 1994; Yáñez-Arancibia and Day 2004; Yáñez-Arancibia and Lara-Domínguez 1999).

For Cuba, the study focuses on mangroves and wetlands which are located on the south coast of Havana province. Across these mangroves and wetlands, a road-like freshwater barrier was built during 1985-1991 as part of the actions to guarantee an adequate freshwater supply for an important agricultural area and settlements like the national capital (Havana City) (IGT 1999). The manifold consequences of the barrier include enhanced quality and quantity of groundwater, and mangrove dieback and creation of new mangrove plantations (IGT 1999; Jiménez 2004; Menéndez 2000).

The study area has been considered as Cuba's littoral zone under highest risk mainly due to its high vulnerability to extreme meteorological events, which is influenced by local trends of long-term sea level rise and history of anthropogenic impacts (Hernández 2006; IGT 1999; Menéndez 2000; Mitrani et al. 2000).

The study area was covered by an Environmental Audit in 1999 to evaluate the positive and negative impacts of the freshwater barrier built across coastal wetlands (IGT 1999). That Audit was comprehensive enough as to leave little space for not explicitly action-oriented scientific work. A more detailed study on the recent local development of mangroves was one of the few pending issues.

Regarding resilience, this study builds on Holling's (1973) definition of resilience and main follow ups (Walker et al. 2006a, b) and conceptual discussions (Carpenter et al. 2001; Grimm and Wissel 1997). Two different approaches are applied. The first focuses on a concrete type of system, i.e., mangrove vegetation, and thus mangroves as a land-cover. This first approach rephrases Holling's definition into resilience of mangroves (Carpenter et al. 2001; Grimm and Wissel 1997). The second approach is more general, i.e., it addresses an environmental system and strictly follows Holling's (1973) definition of resilience without modifying the definition. A system is considered a network of partly opposing and partly reinforcing processes, observable as changes in their intersections at specified variables (Levins 1998).

The study takes resilience of mangroves as the capacity or tendency of the mangrove plants and vegetation, and so of mangroves as ecosystems and land covers, to recover. That capacity makes non-returnable changes of mangroves into another land cover a rare event, even when the components of mangrove ecosystems have markedly changed as is commonly the case when natural and human impacts influence mangrove areas. Components of ecosystems (Jørgensen and Müller 2000) refer to both abiotic and biotic components (Begon et al. 1996a). Land cover refers to the land's physical attributes (e.g., forest, mangrove) (Moran et al. 2004); its insertion in the above-stated definition of mangrove resilience enhances the spatial aspect of researching resilience of mangrove ecosystems.

The overall goals of the thesis are:

1. To assess the resilience of mangroves to the flooding induced by a freshwater barrier built on the south coast of Havana province (Cuba) during 1985-1991.
2. To propose a methodological framework for assessing and managing resilience of environmental systems.
3. To promote collaboration between practitioners of mangrove research and mangrove restoration.

The first goal is addressed in chapters 2 to 5. The second goal is addressed in chapter 6. The third goal is addressed in chapter 7, based on five study cases in Cuba, Mexico and United States of America.

The broad definition of mangrove resilience leads one to expect a broad methodological approach to mangrove resilience. Regarding the mangroves and wetlands of the south coast of Havana province, remote sensing (with satellite images) first allowed a general spatial comparison of the land covers existing before (1985) and two decades after the construction of the freshwater barrier (2001). Second, field ecological surveys in 2005 yielded information on changes in ecosystem components (mainly water level) that can be linked to the major land-cover changes, and which explain the resilience the mangrove cover has manifested (i.e., recovery of mangrove vegetation). The main spatial differentiation imposed by the freshwater barrier (i.e., dammed and non-dammed mangroves) was also addressed with field ecological surveys in 2005. Since not only regeneration of mangrove vegetation but also restoration can be indispensable for mangrove resilience, the five study cases in Cuba, Mexico and USA provide a wider understanding on mangrove resilience than the study case in Havana province in both a geographical and action (restoration) oriented sense. Finally, a methodological framework based on qualitative mathematical modeling is proposed for improving the assessment and management of environmental systems. Such general framework prepares the way to approach, in further work, resilience in an absolute sense (e.g., explicitly including socioeconomic information) and not just focused on the ecological aspects as in this study.

2 STUDY AREA

2.1 Location, hydrology and construction of a freshwater barrier

The wetlands in the south of Havana province in the Republic of Cuba occupy a coastal west-east oriented belt of 129 km length and 2-10 km width. The study area is the western part of these wetlands (Figure 2.1). Sectors I-VI extend north-south from the lanward of the wetlands until the sea. The sectors are limited west and east by the main roads accesing coastal towns, and correspond to the territorial units that are relevant for the implementation of local wetland management (IGT 1999; Jiménez 2004).

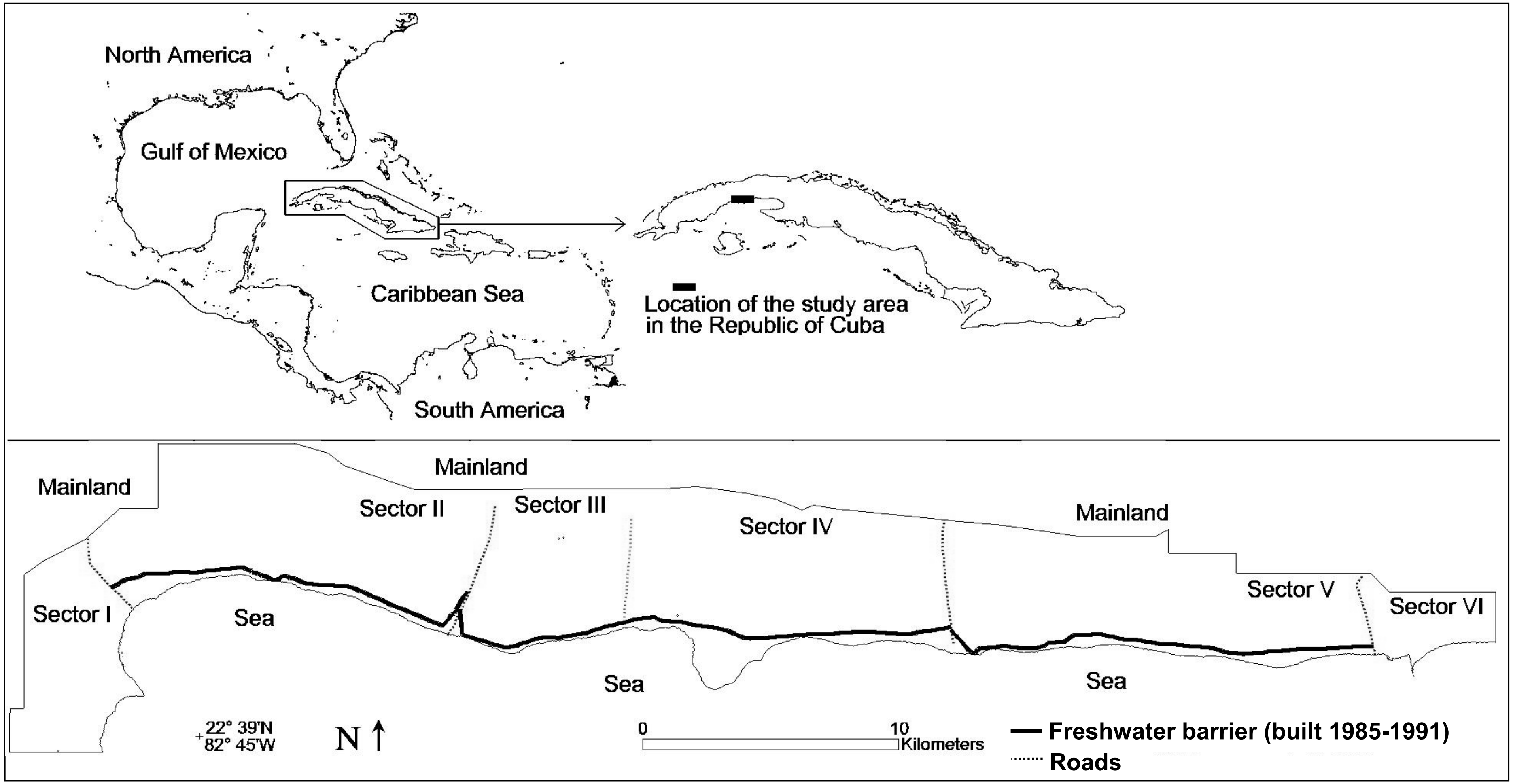


Figure 2.1 Study area: western part of the coastal wetland belt, on Havana province, Republic of Cuba

The study area is in the western coastal section of an 830 km^2 watershed named Artemisa – Quivicán, which is interconnected to the sea. The water originates by infiltration on the landward side of the wetlands. Water flows predominantly from north to south towards the sea, and the water level gradually becomes shallower from the upland (groundwater level up to 50 m) to the coast (0-1 m) (IGT 1999). Subsurface drainage predominates, corresponding to the plain relief and karstic geology; the low natural surface runoff reaching the wetlands spreads laterally and contributes to the swampy conditions (Menéndez 2000).

In the study area, a barrier to the freshwater flowing towards the sea was built along the coast during 1985-1991 (Figure 2.1 and Figure 2.2).

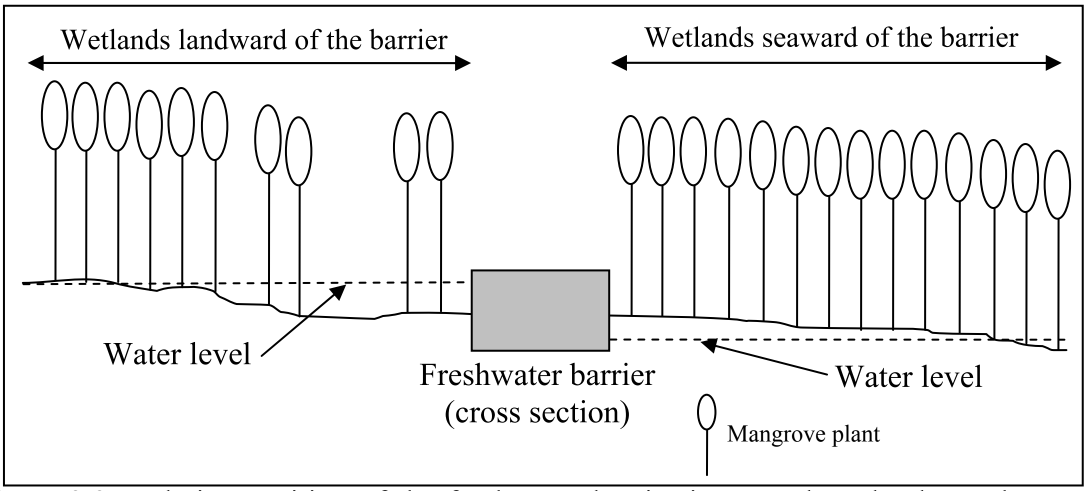


Figure 2.2 Relative position of the freshwater barrier in coastal wetlands on the south coast of Havana province; built 1985-1991

The barrier was part of the solutions implemented to keep guarantying adequate freshwater supply for an important agricultural area and settlements landward (north) of the barrier (including the national capital Havana City) (IGT 1999). Seawater intrusion was an argument too.

The barrier is an unpaved road, with gravel on top of clay layers. It is 52 km long, has a maximum width of 8 m, and is mostly placed about 500 m from the coastline. To its landward (north), the barrier retains the water and slows down the flow into the sea. It was built at elevations of 1-1.5 m above sea level and includes paved spillways (altitude lowered to 0.7-0.9 m) every 1-1.5 km to allow excess water to flow into the sea. The barrier was built in five phases during 1985-1991: 1985 (4.1 km in the western end of sector V), 1986 (11 km in the eastern end of sector IV), 1987 (the rest of sector IV, and sector III), 1989 (sector II), 1991 (rest of sector V) (Jiménez 2004).

The benefits brought by the barrier to the quality and quantity of groundwater have been related to: re-establishment of more natural conditions in the lowest coastal section of the aquifer via elimination of channels; elevation of the phreatic surface in the wells used for irrigation; improvement of water quality by decrease in salinity; increased depth of the boundary freshwater – saline water; increased amount of water in the aquifer; and improved conditions against seawater intrusion. However, those benefits are not homogeneous throughout the watershed and not completely clear, because in the period after the barrier was built (1990-95) the years were more humid, the operation of several water wells was stopped, and other water-use strategies were applied (IGT 1999; Jiménez 2004).

2.2 Geomorphology, soils and vegetation

The type of landscape in the study area is common on the south coast of Cuba, and is typically lower in altitude than the north coast. The wetlands in the study area are mainly over limestone and highly developed covered karst in a 2-10 km wide coastal belt (IGT 1999; Menéndez 2000). They develop on a very low and softly undulated (0-1m) littoral swampy plain of marine-palustrine origin. Soils are predominantly mineral swampy hydromorphic; they are deep humic marl and saline peat, not suitable for agriculture (Menéndez 2000). The agricultural area landward of the wetlands have well drained and highly productive red ferralitic soils (IGT 1999).

Beaches on this Holocene coast have muddy-sandy bottoms; sand deposits are scattered (IGT 1999; Menéndez 2000). The wetland vegetation comprises mangroves and wetland forests, herbaceous vegetation and forest plantations (mainly *Casuarina equisetifolia* J.R. et J.G.Forst., *Calophyllum antillanum* Britt., *Hibiscus pernambucensis* Arruda and *Rhizophora mangle* L.). The local mangrove species are *Rhizophora mangle*, *Avicennia germinans* (L.) L. and *Laguncularia racemosa* (L.) Gaertn. f. The pseodomangrove *Conocarpus erectus* L. is present, too. Hereafter, these species are respectively refered in this thesis as *R. mangle*, *A. germinans*, *L. racemosa* and *C. erectus*.

2.3 Climate

High evaporation and temperatures typify the local climate (Menéndez, 2000). The dry (November-April) and rainy seasons (May-October) account for 21 and 79%, respectively, of the annual average rainfall (800-1200 mm). The annual mean temperature reaches 24-26 °C (Menéndez 2000). The mean minimum temperature (20-22 °C) is typical of January-February (IGT 1999), while the mean maximum temperature (26-28 °C) is typical of July-August (IGT 1999).

Evaporation is high (2 000-2 200 mm) and the mean annual humidification is moderate. Relative humidity has its highest values during June-October (80-86%) and the lowest during March-April (about 78%). The daily oscillation is more pronounced than the seasonal one (IGT 1999); annual mean of humidity at 7:00 h is 90-95 %, and 60-70% at 13:00 h (IGT 1999; Menéndez 2000).

The wind predominantly blows from the east and its annual mean speed (3-10 km per h) is not high; the highest monthly mean speeds occur during the dry season, mainly in March and April (IGT 1999).

2.4 Tropical cyclones

The study area is located in the western part of Cuba, the one most exposed to tropical cyclones, extra-tropical depressions and cold fronts (IGT 1999). During 1902-2005 the central and eastern sections of the study area (sectors IV, V and VI) were directly influenced by more frequent and stronger tropical cyclones than the western section (sectors I, II and III). This is because in tropical storms rainfall and wind are mostly strongest in the east of the trajectory (Longshore 1999), and during 1950-2005, the central part of the study area (sector IV) was a point of common entrance for the cyclones: tropical depressions Alma in 1966 (turned into hurricane category 1-2) and Jenny in 1969, tropical storm Irene in 1999, and hurricane Charley (category 1-2) in 2004 (NOAA Coastal Services Center 2006).

2.5 Socio-economic setting

Agriculture and water use are the main historic source of anthropogenic change in the study area, peaking with the construction of the freshwater barrier in 1985-1991 (Figure 2.1). The high demand of water for agriculture and human consumption includes the demand of the capital Havana City (about 2 million inhabitants), located on the north coast opposite to the study area.

However, the wetland area itself has never hosted a high population, and in 1992 had 7 629 inhabitants (2 678 houses) in 7 settlements at a density of 33 inhabitants per km^2 (IGT 1999; Menéndez 2000). There is no scattered population in the wetlands. From its early occupation (already in 1576), the swampy conditions and scarce fluvial shelter for marine navigation lead to a pattern of isolated coastal settlements, mainly near roads to the coast. Most of the people (5441) live in the coastal town Surgidero de Batabanó, located in sector VI; the original location of the national capital was about 20 km east of sector VI in Spanish times at the beginning of the 16th century.

Before the Spanish colonization of Cuba in 1492, the local native population had a subsistence economy based on collecting natural products, hunting, fishing and primitive agriculture. They were settled landward of the coastal swamps and forests and did not change the territory much (IGT 1999).

Agriculture (both for national consumption and exports) was the main historical economic activity in the region north to the wetland belt, including potatoes, timber, fruits, pastures and cattle raising, tobacco, sugar cane (strongly related to the introduction of the railway in 1837), and coffee (the 200-year-old infrastructure still exists) (IGT 1999; Menéndez 2000).

The anthropogenic transformation of the wetlands in the study area included setting up a channel network for providing access, extracting wood and charcoal, draining to gain agricultural land and avoid floods. Other actions have been: building coastal infrastructure; sand extraction and cutting of *R. mangle* mangroves on the coastline (strongly related to coastal erosion); and peat extraction (Menéndez 2000). In 1932 there were already numerous channels, mainly in sectors IV and V (IGT 1999).

2.6 Marine area

Marine currents link the study area to eastern wetlands in Ciénaga de Zapata, of regional importance in the Caribbean and Cuba's largest wetland (Hernández 2006). Those currents flow to the west at a speed lower than 20 cm per s under the influence of the Alisios winds (Blázquez et al. 1988). The tides have a low amplitude (25-50 cm) (Hernández 2006; Rodríguez and Rodríguez 1983). However, events like tropical cyclones and front systems associated to extra-tropical depressions produce significative variation (increase or decrease) of the sea level during hours to weeks (commonly 3-15 days) (Hernández 2006).

There is evidence of a marked local long-term sea level rise (Hernández 2006; IGT 1999; Menéndez 2000). The retreat of the coastline is considerable (as much as 3 m per year in the period 1967-1998), and is influenced by less sediment reaching the coast after the freshwater barrier was built (IGT 1999).

The marine area to the south, the Gulf of Batabanó, is one of Cuba's richest marine platforms (IGT 1999) and is of high economic importance for fisheries (Hernández 2006). The transformation of wetlands and coastal zone in the study area could negatively influence the fisheries in the Gulf of Batabanó (Hernández 2006), although there is no definite proof, and mismanagement also seems to be important (e.g., overfishing) (IGT 1999). Local fishermen have documented positive effects on fisheries since the 1930s due to the construction of channels for draining the wetlands (IGT 1999). Less freshwater reaching the coast, due to the construction of the barrier, has been also interpreted as improving salinity values and having a positive impact on the marine biota close to the coast (IGT 1999).

3 CHANGES IN LAND COVER CAUSED BY THE FRESHWATER BARRIER

3.1 Introduction

Wetlands deliver a wide range of ecosystem services that contribute to human well-being (e.g., water supply, flood regulation, recreational opportunities), but the degradation and loss of these ecosystems is more rapid than that of others. Infrastructure development is among the primary direct drivers of wetland degradation and loss (MEA 2005a). However, the construction of infrastructure can be indispensable.

Worldwide, artificial damming structures (e.g., roads, dams) have led to highly negative impacts on mangroves (Field 1996; MEA 2005a; Yáñez-Arancibia and Day 2004) by changing the depth, frequency and duration of their inundation. The coastal wetlands in the south of Havana province are a typical example of direct use of wetland service (freshwater for agriculture and cities), infrastructure for warranting that use (a road-like freshwater barrier), degradation of wetlands linked to increased inundation and inadequate institutional settings in the past (e.g., legislation), and opportunities for restoration. Impacts on mangroves are among the most controversial influences of the freshwater barrier because of the relevance of the mangroves for forestry and fisheries (IGT 1999).

Analysis of land-cover change in impacted mangroves belongs to the research tools directed to prevent or alleviate further degradation of mangroves. The research findings can also support restoration based on the natural capacity of mangrove vegetation to recover (Capote-Fuentes and Lewis 2005; Lewis 2005). Remote sensing generated data (e.g., aerial photographs, satellite images) have been widely used for mapping wetlands (Dahdouh-Guebas et al. 2004; Jensen 1996). Bi-temporal change detection based on Landsat satellite images requires a careful matching of the processes of interest (in this study wetland change, massive mortality and further development of mangroves) and the spectral and spatial characteristics of the images so that actual and relevant changes between the two addressed time steps can be revealed (Coppin et al., 2004; Dahdouh-Guebas et al. 2005; Green et al. 1998; Wang et al. 2004).

The objective of this chapter is to present the changes in land cover, particularly in mangroves, caused by the construction of a freshwater barrier in 1985-

1991 in the coastal wetlands of Havana province in Cuba. For a more profound explanation of the pattern observed, water level was measured *in situ*. Also, for the two time steps addressed (years 1985 and 2001), maps of mangrove species were generated for one of the main mangrove areas.

3.2 Database and methodology

Two Landsat satellite images were acquired from 25 Jan 1985 (TM, dry season) and 4 April 2001 (ETM+, end of the dry season) for the study area (Figure 2.1). The seasonal variation of the vegetation (Capote et al. 1989) should not be a problem when interpreting the expected land-cover changes caused by freshwater barrier from 1985 to 2001 (IGT 1999). However, a Post-Classification Comparison Change Detection algorithm was selected (Coppin et al. 2004; Jensen 1996), because spectral differences between the images, probably induced by vegetation phenology (Capote et al. 1989), persisted after normalizing the 1985 image to that of 2001 (Hall et al. 1991a). The normalization was kept in the pre-classification process to increase the comparability of the classified images. The empirical calibration line was accepted with a coefficient of determination R^2=0.87, which resulted from water bodies and paved areas sets (1677 and 1542 pixels, respectively, for 1985, and 1567 and 1477 pixels for 2001). These sets represent similar percentages of the original images as required by the Tasseled-Cap-based method of Hall et al. (1991a).

The 2001 image was geometrically corrected with 26 ground control points collected in the field (October 2004-May 2005) with a hand-held eTrex-Legend Garmin GPS (Global Positioning System) (root-mean-square error RMS=0.57). The 1985 image was co-registered to the 2001 image using 33 reference points (RMS=0.64).

Knowing the land covers to be expected in the study area (IGT, 1999) and following a comprehensive field survey, the land covers were mapped via supervised classification (maximum likelihood algorithm) (Jensen 1996). A classification scheme (Capote et al. 1989) with a previous application in the study area (IGT 1999) was adopted (Table 3.1).

Table 3.1 Land-cover classification scheme

Land-cover class. Parenthesis: Number of pixels in training class for 1985 and 2001.	Description
Mangroves (46, each year)	Cover with predominance of the typical mangrove tree species *R. mangle*, *A. germinans*, *L. racemosa* and the associate species *C. erectus*. This class includes both natural and planted mangroves.
Flooded Mangroves (12 in 1985, and 46 in 2001)	Focused on inland flooded mangroves and not on coastline mangroves flooded by tides.
Wetland Forests (35, each year)	This class comprises both natural Swamp Forests (Capote et al., 1989) and forest plantations different from mangroves. The cover of natural forests is typically dominated by trees like *Tabebuia angustata* Britt. and palms like *Sabal parviflora* Becc. and *Roystonea regia* (Kunth) O.F.Cook. Plantations commonly have the trees *Casuarina equisetifolia*, *Calophyllum antillanum* and *Hibiscus tiliaceus*.
Herbaceous Swamp (46 , each year)	Typically dominated by *Typha domingensis* Pers. and *Cladium jamaicense* Crantz
Crops, Pastures and Ruderal Vegetation (46, each year)	Agricultural areas and vegetation associated with settlements.
Inland Open Water (16, each year)	Mainly natural lagoons and sites where vegetation has degraded.

The final land-cover maps include a further class for the sea, which was superimposed as a mask from a previous unsupervised classification of each image.

Local people with experience (10-70 years) in forest management and fisheries provided additional field knowledge to find training sites representative of the classes to be included in the supervised classifications. The first three bands (components) of the Principal Component Analysis of each image (years 1985 and 2001) accounted for 98% of the variability in the spectral data of these sites. Thus, high order components were not kept for further analysis. The training samples were spectrally separable (Transformed Divergence mostly higher than 1.900) (Jensen 1996).

The documented low extension of Flooded Mangroves in 1985 (IGT 1999) caused its training class to be rather small (12 pixels, Table 3.1) (Jensen 1996). That training class was solved with pixels on the edge of a lagoon reported by maps at least from 1976, and which was visited during the field survey. The classification of the 2001 image had labeled the pixels around that lagoon as Flooded Mangroves although those pixels were not in the training class Flooded-Mangroves of 2001. The situation for Inland Open Water was similar in 1985 and 2001 (16 pixels, Table 3.1). Pixels checked in the field as open water, and located to the lowest left extreme of the greenness-brightness histogram of the image Tasseled Cap transformation ensured the representativeness of these training classes (Hall et al. 1991a; Jensen 1996).

The field survey also provided the ground-truth data to assess the accuracy of the land-cover map derived for 2001. Available information about local historical distribution of land cover types (IGT 1999) allowed completing the ground-truth information for the 1985 map.

For complementing the findings from remote sensing in the wetlands landward and seaward to the barrier (sectors II, III, IV and V, Figure 2.1), the water level was measured *in situ* in 2005 in the dry season (March-April) and the rainy season (September-October). Water level refers to either the depth of flooding water or the water table; both were measured with a plastic ruler. The depth of the water table was found by drilling and waiting 30 minutes until the water level stabilized (Boto 1984).

For sector II, the dominance of mangrove species was mapped by addressing (masking) the areas labeled as Mangroves and Flooded Mangroves in the land-cover maps of 1985 and 2001. The classification procedure was similar to that yielding the land-cover maps, with supervised classification (maximum likelihood algorithm) based on the first three bands (components) of the Principal Component Analysis of each image. The training classes (18 pixels each) were found according to the pattern of species distribution observed in the field and previously reported for the study area (IGT 1999; Menéndez 2000): *R. mangle* along the coastline, *A. germinans* in a wide monospecific area, and *L. racemosa* towards the landward boundary of the mangroves. An additional class “Mangroves scattered in open water”, mainly for addressing newly flooded areas appearing in 2001, was trained with the same training class Inland Open Water (16 pixels) that had been used for obtaining the land-cover maps. The training classes leading to the maps were spectrally separable (Transformed Divergence 1.859, 1.907-2.000) (Jensen 1996).

A quantitative accuracy report is desirable when mapping via remote sensing (Jensen 1996) as it was done for the maps of land covers in the entire study area for

1985 and 2001. Similarly, when mapping species dominance in sector II, a quantitative accuracy assessment was conducted for the map of 2001. However, no assessment for the map of 1985 was performed due to the not spatially detailed ground-truth information available for this time step. The map obtained is used solely for finding general trends of change, and the correctness of the map is supported by field evidences of change associated to the barrier (e.g. dead stems of non-*R. mangle* trees in open water), publications on mangrove species composition before the construction of the barrier (IGT 1999), and personal communications of local people during the field survey.

3.3 Results

3.3.1 Land-cover maps

The land-cover maps for 1985 and 2001, resulting from the supervised classifications, are presented in Figure 3.1.

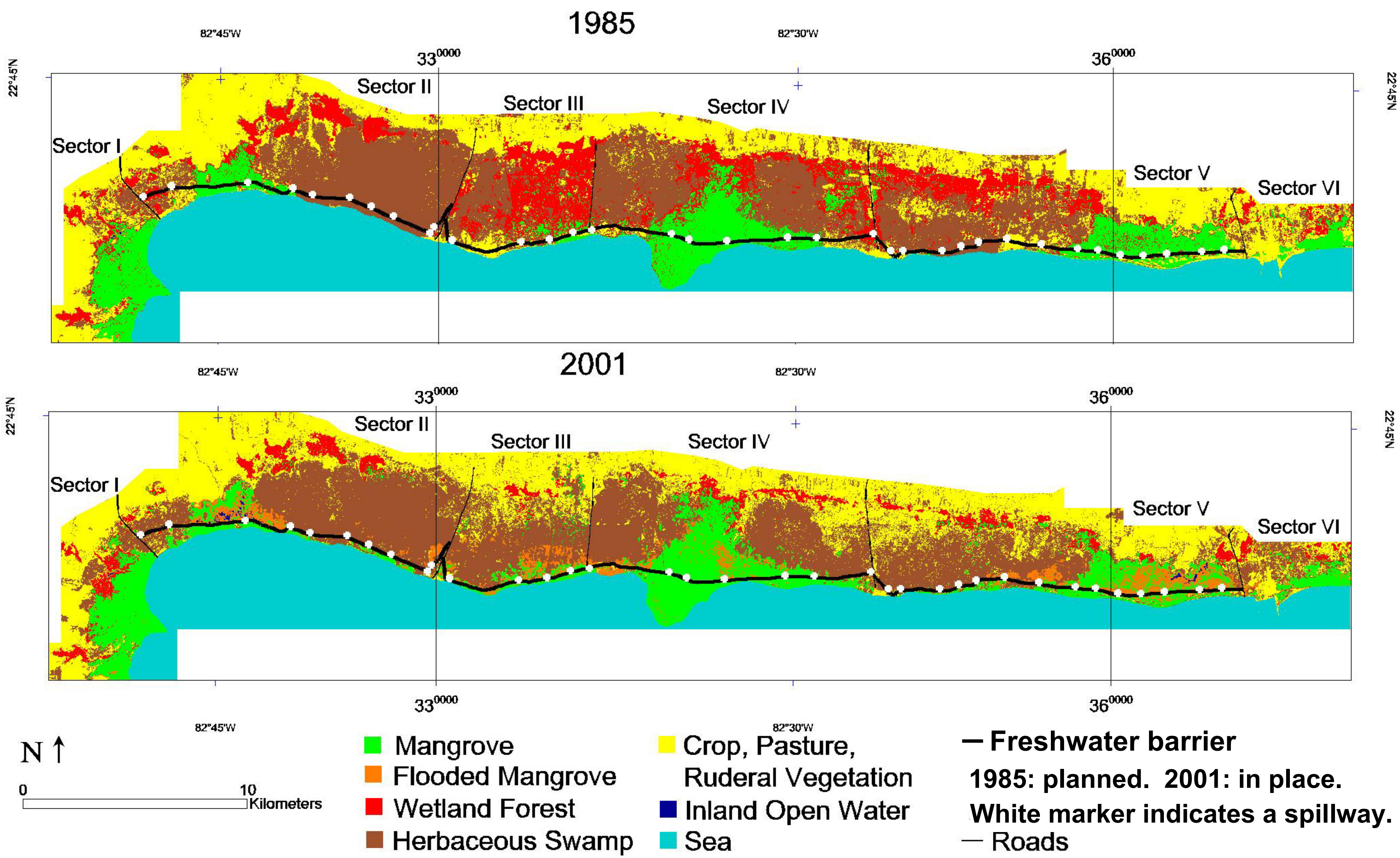


Figure 3.1 Land-cover maps for 1985 (before the construction of the freshwater barrier) and 2001 (with the barrier in place)

The independent accuracy assessment of the classifications, based on 228 and 266 ground-truth points for 1985 and 2001, yielded overall accuracies of 85% and 91%, respectively (Table 3.2). The accuracies of the land cover Mangroves are among the highest for both maps (87-96%), whereas for Inland Open Water in 1985 only 68% could be correctly assigned.

Table 3.2 Accuracy (error matrixes) for the supervised classifications of land cover. The head name of a column represents ground-truth information (the actual class of the pixels addressed in that column). The values in that column indicate to which classes the pixels were assigned by the classification. The bottom row and extreme right column summarize the matrix. The Producer Accuracy of a class indicates the probability that the classification has labeled an image pixel into Class A given that the ground truth is Class A. The User Accuracy indicates the probability that a pixel is Class A given that the classification has labeled the pixel into Class A (Jensen 1996)

a. Year 1985. Overall Accuracy = 196/228 = 85%

Class	Mangro-ves	Flooded Mangroves	Wetland Forests	Herbaceous Swamp	Crops, Pastures and Ruderal Vegetation	Inland Open Water	User Accuracy: Pixels (%)
Mangroves	48	1	4	0	0	2	48/55 (87)
Flooded Mangroves	0	9	0	0	0	3	9/12 (75)
Wetland Forests	1	1	43	1	0	0	43/46 (93)
Herbaceous Swamp	0	0	2	40	5	0	40/47 (85)
Crops, Pastures and Ruderal Vegetation	1	0	1	9	45	0	45/56 (80)
Inland Open Water	0	1	0	0	0	11	11/12 (91)
Producer Accuracy: Pixels (%)	48/50 (96)	9/12 (75)	43/50 (86)	40/50 (80)	45/50 (90)	11/16 (68)	196/228 (85)

b. Year 2001. Overall Accuracy = 244/266 = 91%

Class	Mangro-ves	Flooded Mangroves	Wetland Forests	Herbaceous Swamp	Crops, Pastures and Ruderal Vegetation	Inland Open Water	User Accuracy: Pixels (%)
Mangroves	48	0	2	0	0	0	48/50 (96)
Flooded Mangroves	2	44	0	0	0	1	44/47 (93)
Wetland Forests	0	0	45	0	0	0	45/45 (100)
Herbaceous Swamp	0	1	0	48	6	0	48/55 (87)
Crops, Pastures and Ruderal Vegetation	0	0	3	2	44	0	44/49 (89)
Inland Open Water	0	5	0	0	0	15	15/20 (75)
Producer Accuracy: Pixels (%)	48/50 (96)	44/50 (88)	45/50 (90)	48/50 (96)	44/50 (88)	15/16 (93)	249/266 (91)

For both years (Table 3.2), confusion between Herbaceous Swamps and Crops-Pastures-Ruderal-Vegetation occurred. For 1985, the wrong classifications included Wetland Forests assigned to Mangroves, and Inland Open Water assigned to Flooded Mangroves and Mangroves. For 2001, the errors included Flooded Mangroves classified as Inland Open Water.

3.3.2 Changes in land cover

The most obvious changes are the decrease in Wetland Forests, mainly plantations, and the increase in Flooded Mangroves (Table 3.3) landward of the barrier, in sectors III, IV, V, and sectors II, V, respectively (Figure 3.1).

Table 3.3 Land-cover change matrix for the period 1985-2001: Into which covers of 2001 each cover of 1985 developed (rows), and from which covers of 1985 each cover of 2001 developed from (columns). (Area of land-cover types in hectares)

		Land cover in 2001 (ha)						
		Man-groves	Flooded Mangroves	Wetland Forests	Herba-ceous Swamp	Crops, Pastures, Ruderal Vegetation	Inland Open Water	Sea
	Total area	4777.5	1425.2	1725.1	10724	10039.8	43.3	11870.3
Land-cover in 1985 (ha)								
Mangroves	4529.8	3235.6	671.8	173.1	216.5	146.4	39.7	46.7
Flooded Mangroves	11.2	2.7	3.8	1.9	1.5	0.5	0.8	0
Wetland Forests	4847	724.8	160.4	1206.3	1441.9	1310.5	1.4	1.7
Herbaceous Swamp	10045.6	422.2	325.9	141.9	7275	1865.2	0.2	15.2
Crops, Pastures, Ruderal Vegetation	9361.3	352.4	260.8	201.5	1787	6711.3	0.1	48.2
Inland Open Water	3.1	0.3	1.3	0.4	0	0	1.1	0
Sea	11807.2	39.5	1.2	0	2.1	5.9	0	11758.5

The road-like freshwater barrier facilitated accessing wetlands and creating new mangrove plantations. The net increase in Mangroves between 1985 and 2001 (from 4530 ha to 4777 ha) includes the conversion of Mangroves into Inland Open Water and Flooded Mangroves, and also new mangrove plantations had been created in Herbaceous Swamps and Crops-Pastures-Ruderal Vegetation (Table 3.3). Examples of new mangrove plantations in the 2001 map are: the southeast of sector II, the southwest of sector III, the southeast of sector IV, and the southwest of sector V (Figure 3.1). Between 1985 and 2001, 3236 ha of mangroves remained unchanged (Table 3.3).

The extension of Mangroves also increased from Wetland Forests (725 ha, Table 3.3). This was in areas landward of the barrier, in the northern borders of neighboring Mangroves and Wetland Forests (sector II) and also in small Wetland Forests located in mangrove areas (sector V) (Figure 3.1).

Coastline regression characterized by loss of Mangroves and sandy coast vegetation became evident only in the southeast extreme of the study area, outside the dammed sectors (Figure 3.1, sector VI).

In both the dry and the rainy seasons in 2005, the water level tended to be higher in wetlands located north (landward) of the barrier than south (seaward) of it (Figure 3.2). The differences between landward and seaward water levels were more marked in wetlands located in sectors II and V, where the conversion of Mangroves into Flooded Mangroves was more pronounced (Figure 3.1). In contrast, in these later two sectors, the barrier had the highest number and density of spillways (Figure 3.2).

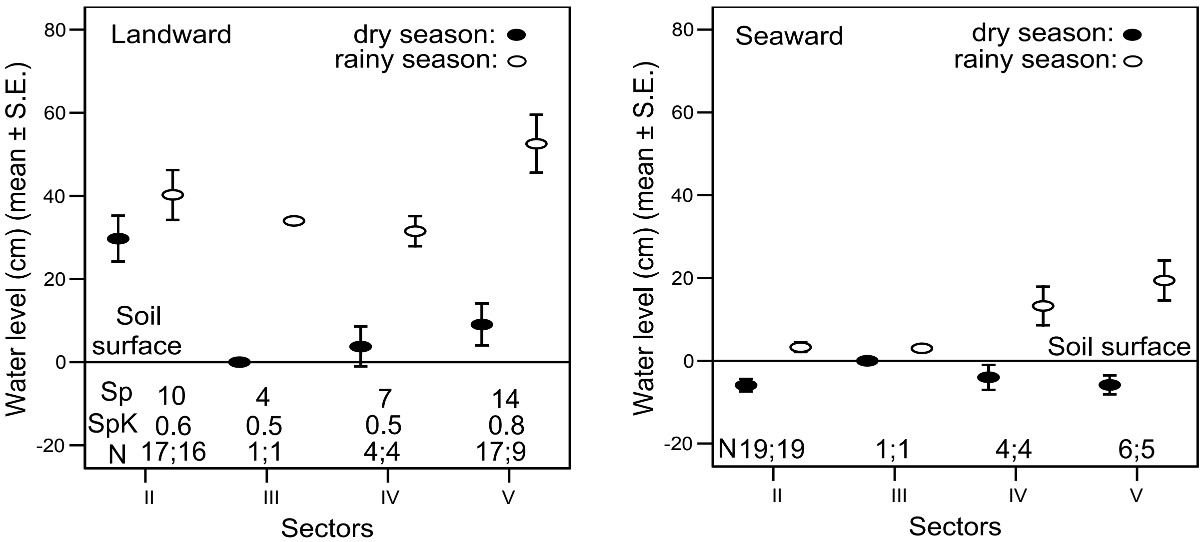


Figure 3.2 Water level (2005) landward and seaward of the freshwater barrier (dry and rainy season), number of spillways (Sp), and spillway per km (SpK) in each sector of the dammed area (sectors II to V) (N: number of plots)

Water was observed flowing over the spillways in both the dry and the rainy season in all sectors of the barrier. An exception was observed in sector V: in the second and third spillways east to west in the barrier (Figure 3.1), there was no water flow at the end of the dry season (April 2005). These spillways could not be visited during the rainy season. In the seven spillways located in the largest mangrove extension of sector V (Figure 3.1), vegetation (including mangrove plants) was observed blocking the flow of water across the barrier.

In sector II, although open water still existed where massive mangrove mortality had occurred (Figure 3.1), saplings and trees of *R. mangle* had increased their frequency and also dominated the regenerated vegetation cover (Figure 3.3). Truncated stems of the formerly predominant species *A. germinans* and *L. racemosa* were still found between the new plants or *R. mangle* canopy. They were also present as old trees or saplings in the shallowest sites. The species of the mangrove plants present in small mangrove patches (1-10 m^2) in the class “Mangroves scattered in open water” could not be distinguished when classifying the satellite images (Figure 3.3).

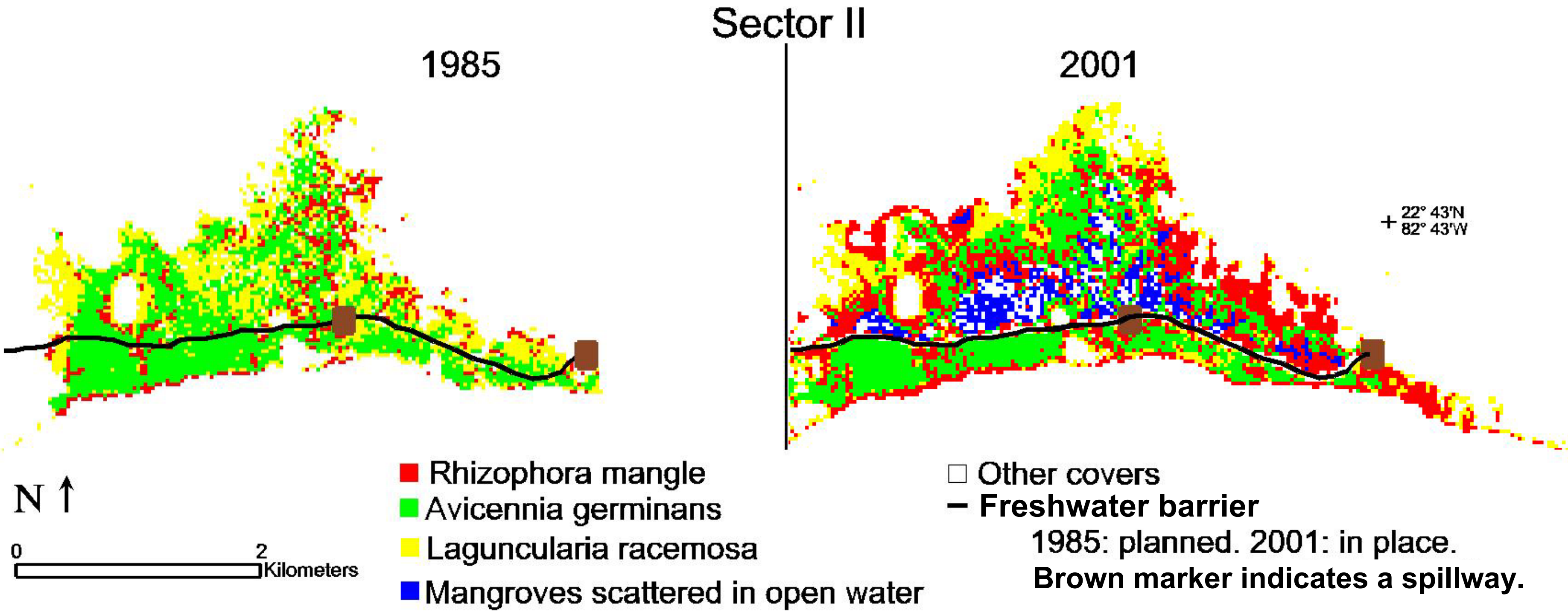


Figure 3.3 Dominance of mangrove species in sector II

Seaward of the barrier, the dominant species were the same as before the construction of the barrier, i.e., *A. germinans* was typically distributed in areas closer to the barrier, while *R. mangle* was found nearer to the coastline (Figure 3.3) and along freshwater channels. *Rhizophora mangle* is the species commonly planted, as evidenced in new mangrove areas appearing in the right extreme of the 2001 map (Figure 3.3).

Both maps (Figure 3.3) reproduce the local pattern of mangrove distribution observed during the field survey. For the map of year 2001, further support came from a quantitative accuracy assessment (Table 3.4).

Table 3.4 Accuracy (error matrix) of the supervised classification of dominance of mangrove species for the year 2001 for sector II (as in Figure 3.3). Overall Accuracy = 55/60 = 91%

Class	*R. mangle*	*A. germinans*	*L. racemosa*	Mangroves scattered in open water	User Accuracy: Pixels (%)
R. mangle	14	1	0	1	14/16 (87)
A. germinans	1	14	2	0	14/17 (82)
L. racemosa	0	0	13	0	13/13 (100)
Mangroves scattered in open water	0	0	0	14	14/14 (100)
Producer Accuracy: Pixels (%)	14/15 (93)	14/15 (93)	13/15 (86)	14/15 (93)	55/60 (91)

3.4 Discussion

3.4.1 Land-cover maps

The maps allow recognition and interpretation of the spatial features of the land covers in 1985 and 2001 in relation to the construction of the freshwater barrier.

As it is common when obtaining maps based on spectral data of satellite images, classes (land covers) sharing components tend to differ less spectrally, and thus classification errors can be detected by accuracy assessments (Jensen 1996). In the study area, Mangroves and Flooded Mangroves share plant species and water. Also as expected, the lack of synchronous ground-truth information for 1985 and 2001 led to a lower precision for the older time step (Green et al. 1998).

The Post-Classification Comparison Change Detection algorithm proved to be the right choice for comparing the maps of 1985 and 2001. The seasonal variation of vegetation phenology, which as expected, was not controlled by the conducted normalization (Hall et al. 1991a), could in further research be controlled with phenological data representative of the seasons where the spectral data of the images were acquired (Coppin et al. 2004; Hall et al. 1991b).

Seasonal variation in vegetation abundance could also have partially caused the errors found in the classification, since the 2001 image was acquired in a more advanced phase of the dry season than the 1985 image. That seasonal variation also prevented classification of both images with training classes (spectral data) selected on the 2001 image, which was closer in time to the ground-truth survey. When such classification was attempted to obtain a preliminary map of 1985, Wetlands Forests were misclassified as Mangroves.

Seasonal changes in both the cropped areas and the inundation regime of Herbaceous Swamps (Capote et al. 1989) could have also been the reason for the confusion between Herbaceous Swamps and Crops-Pastures-Ruderal Vegetation in the final maps of both years. A further class might be added to address karst spots classified as Crops-Pastures-Ruderal Vegetation in Herbaceous Swamp areas.

The small areas of classes like Flooded Mangroves (in 1985) and Inland Open Water (in 1985 and 2001), documented by field surveys (IGT 1999), caused their training classes to be rather small (12 to 16 pixels). Also, these small areas neither demanded nor allowed increasing the number of ground-truth points for assessing the accuracies of the class mapping, and reduced the importance of some of these accuracies, which were the lowest of all classes (e.g., 68% for Inland Open Water in 1985) (Jensen 1996).

3.4.2 Changes in land cover

Despite classification errors when obtaining the land-cover maps for 1985 and 2001 as discussed in the previous section, the interpretation of the maps allows stating that the freshwater barrier caused the increase in Flooded Mangroves and Inland-Open-Water, and the decrease in Wetland Forest and Mangroves landward of the barrier. The present study empirically documents that those changes in land cover correspond to higher water level in 2005 in the wetlands located landward of the barrier. These higher water levels are a result of the barrier slowing down the flow of freshwater into the sea, which was the aim of the barrier in order to guarantee adequate freshwater supply for the agricultural area and settlements located lanward of the wetlands of the study area (IGT 1999; Jiménez 2004).

The effectivity of the spillways of the barrier could be improved and thus the area of Flooded Mangroves reduced. Since the different segments of the barrier were built in different years of the period 1985-1991, different construction standards may have been applied in the different segments of the barrier. Correcting the application of the standards can increase the recovery of wetlands and mangroves. In sector II (Figure 3.1), in its largest area of Flooded Mangroves, there are hardly any spillways. The organizations in charge of managing the barrier should consider constructing at least three new spillways here. Two new spillways should be located in the area between the existing second and third spillway. The third spillway should be located between the existing third and fourth spillway.

In sector V, lowering the altitude of spillways seems imperative for effective flow of excess water from the landward to the seaward side of the barrier. Currently, that flow is absent in the largest area of Flooded Mangroves in the sector.

The Environmental Audit for the barrier in 1999 already recommended the maintenance of the spillways (e.g., removal of plants blocking the flow of water) as a simple measure to manage the negative impacts on the local wetlands (IGT 1999). Maintenance of the spillways can also increase the mangrove cover of the new mangrove plantations that were created after the barrier increased the access to wetlands. Mangrove cover was only up to 60% when the plantations remained flooded in both the dry and the rainy season (chapter 4).

Other spatial features show that the barrier is an important source of change. Close to it, where there was a greater increase in the water level (IGT 1999), Mangroves and Wetland Forests turned into Herbaceous Swamps, a vegetation type adapted to permanent and periodic flooding (Capote et al. 1989). Outside the new flooded sites and closer to the landward border of the wetland belt, Mangroves and Wetland Forests changed into Crops-Pastures-Ruderal Vegetation.

Not all the changes in the study area can be directly linked to the barrier. For instance, the changing of Mangroves into Wetland Forests occurred mainly outside the dammed areas in the boundaries of Mangroves and Wetland Forests. This change might correspond to natural successional changes although exploitation practices, modified after the barrier affected forest plantations, may also be an influence. Errors in the two classifications cannot be ruled out either.

Coastline regression has been a secondary effect pointed out as being very relevant after the barrier was built, and diagnosed as interacting with other cumulative effects as those of logging (IGT 1999). For a coastline highly dynamic by nature, detecting and characterizing permanent changes that are already acute when spreading 30 m landwards may demand: accurate co-registration of images with pixels smaller than 30 m x 30 m, and further training sites for classifying the images with a stronger focus on vegetation types of reduced extension (e.g., sandy coast vegetation) (Coppin et al. 2004; Dahdouh-Guebas et al. 2004; Green et al. 1998).

The observed increased dominance of the more flood-tolerant species *R. mangle* in Flooded Mangroves landward of the barrier indicates that mangrove vegetation can recover from the barrier-induced flooding. This manifestation of mangrove resilience relates to the natural capacity of mangroves to recover from impacts and is a fundamental tool for restoration (Capote-Fuentes and Lewis 2005; Lewis 2005). The anchoring of seedlings in the ground seems to be a critical step for the regeneration of the mangrove cover in conditions of increased water level. Only exceptionally long seedlings of *R. mangle* can have anchored.

Mapping dominance of mangrove species with Landsat images of spatial resolution 30 m x 30 m was possible because the study area has larger sites where single mangrove species tend to dominate. That pattern is fundamental for finding plots serving as training classes for the mapping process (supervised classification). In other Caribbean islands with similar mangrove species but growing in relatively small patches and linear stands along inlets, Landsat TM data allowed distinguishing mangroves from non-mangrove vegetation, but they were not suitable for discriminating mangrove classes (Green et al., 1998). Accordingly, satellite images with spatial resolution higher than 30 m x 30 m can help to overcome the impossibility in the present study of distinguishing mangrove species in small patches (1-10 m^2), which indicate mangrove resilience in Flooded Mangroves.

Extension of mangroves consistently lower than the spatial resolution of the Landsat images used in this study (30 m x 30 m) is also probably the reason why mapping dominance of mangrove species for 1985 was not successful for sectors IV and V (Figure 3.1); the corresponding maps are not presented. In these sectors, the pseudomangrove species *C. erectus* was mainly distributed in scattered patches smaller than 30 m x 30 m. Mapping this species was further made difficult because the historical information on its local distribution was scarcer than for the species mapped in sector II, i.e., *R. mangle*, *A. germinans* and *L. racemosa*.

3.5 Conclusions

Achieving the important goals of the barrier of maintaining or increasing the availability of freshwater for agriculture and human consumption caused permanent land-cover changes. Major changes are concentrated north (landward) of the barrier and include decreases of forest wetland plantations and increases in flooded mangroves. The barrier has also promoted new mangrove plantations via increased access to wetlands.

Besides detecting major changes in land cover, Landsat satellite images can help detecting changes in species composition of mangroves. An increased use of combined remote sensing and ecological data (e.g., changes in water level) is encouraged to re-check for which types of mangroves Landsat images could be the adequate database. Although limitations can arise from the spectral and spatial resolution of Landsat images, these images can have advantages over more modern databases for wide public use. For instance, they are cheaper, and the relation between its spectral information and actual ground features (e.g., vegetation) can make use of abundant relevant literature available since the first Landsat satellite was launched in 1972.

The largest areas of mangroves flooded after the construction of the freshwater barrier (in sectors II and V) can be reduced by including new spillways in sector II, and checking whether the spillways of sector V have the correct altitude. In all sectors, the functionality of the spillways should be regularly checked, for instance to remove plants, which can block the water flow.

4 RESILIENCE OF THE MANGROVE COVER

4.1 Introduction

Changes in land cover involving mangroves are commonly preceded or promoted by changes in ecosystem components, e.g., amount of sediments. Changes in ecosystem components relate to changes in ecosystem processes, e.g., sedimentation, and thus influence the dominance of mangrove plants and the predominance of mangroves as the typical land cover in a specific location. Replacing or promoting a mangrove cover can also occur without previous changes in ecosystem components, for example via logging or planting (Berger et al. 2006; Dahdouh-Guebas et al. 2004; Green et al. 1998; Lewis 2005).

Accordingly, this chapter presents the study on the resilience of mangroves as the capacity or tendency of the mangrove plants and vegetation, and so of mangroves as ecosystems and land covers, to recover. That capacity makes non-returnable changes of mangroves into another land cover a rare event, even when the components of mangrove ecosystems have markedly changed, as is commonly the case when natural and human impacts influence mangrove areas.

Water is a key component of mangrove ecosystems. Mangrove hydrology, i.e., frequency, duration and depth of flooding, is of key importance for both mangrove functioning and management, because it naturally supports the ranges of the abiotic factors (e.g., salinity) in which mangrove plants can dominate. The relationship between mangroves and water bodies like rivers and seas influences important processes like sedimentation and soil dynamics, nutrient exchange and soil anaerobiosis (Drexler and De Carlo 2002; Kjerfve 1998; McKee 1993; Thom 1984; Tomlinson 1986).

At the same time, mangrove hydrology is directly influenced or manipulated by management. This is not only through specific action intended to influence mangroves, e.g., through drainage transforming mangroves into agricultural land, and restoration correcting ground altitude (i.e., topography) and thus flooding. Hydrology can also be influenced unintentionally, for instance when road construction creates artificial barriers and influences water distribution (McLeod and Salm 2006; MEA 2005a; Lewis 2005).

Anthropogenic causes of change in hydrology can lead to either increased or decreased flooding. These causes include construction of structures that change water distribution, e.g., roads and barriers blocking the flow of water, and channels actively diverting water. They may also involve large changes in ground altitude as when digging or filling occurs (Brockmeyer et al. 1997; Field 1996; Lewis 2005; Perdomo et al. 1998).

The natural functioning of mangrove ecosystems is highly influenced by seasonal differences in the input of freshwater into mangroves, which is commonly related to a rainy and a less rainy (or dry) season. During the rainy season, the water level is typically higher and the salinity is lower, while during the dry season the water level lowers and salinity increases (Cintrón and Schaeffer-Novelli 1983; Lugo and Snedaker 1974; Suman 1994; Tomlinson 1986). Corresponding to the changes in water level, the anaerobic conditions of the soil usually become more pronounced (as indicated by more negative redox potential values) when the soil becomes flooded, which typically happens during the rainy season (Boto 1984; Krauss et al. 2006; Middleton 2002; Romigh et al. 2006).

When the hydrology of mangroves is impacted, the natural alternation of flooding frequently changes. Hence, the components related to soil anaerobiosis and salinity change and can reach and remain in extreme values that are impossible for mangrove plant species to cope with, thus leading to temporal or permanent transformation of mangroves into other land covers.

The regeneration of mangrove vegetation is a key element in the resilience of mangroves. The return of mangrove vegetation is the ultimate element allowing saying that mangroves, as a vegetation cover, tend to persist in a specific location, i.e., that mangroves are resilient. The resilience of mangroves can take time to manifest (days to decades), and when the mangrove cover has been severely damaged (to the extent of massive mortality of mangrove plants), the establishment of seedlings and saplings will be of major importance in the re-establishment of the mangrove cover (Capote-Fuentes and Lewis 2005; Lewis 2005; Lugo 1998; Perdomo et al. 1998).

Studying impacted mangroves is crucial for the ecological basis of both protecting and restoring mangroves (Lewis 2005; Lugo 1998; Macintosh and Ashton 2004; McLeod and Salm 2006). Protection is about avoiding extreme impacts that exceed the capacity of mangrove to regenerate. Restoration is about correcting ongoing impacts (e.g., commonly those related to hydrology), promoting natural regeneration of the mangrove vegetation, or planting when natural regeneration does not take place.

The objective of this chapter is to document manifestation of resilience of the mangrove cover, i.e., recovery of mangrove vegetation, in relation to the changes in ecosystem components (mainly water level), which correspond to major land-cover changes caused by the freshwater barrier built in 1985-1991 on the south coast of Havana province. Based on major land-cover changes from mangroves and/or into mangroves (chapter 3), vegetation structure and abiotic factors (e.g., soil redox potential) were surveyed in plots representative of those land-cover changes.

4.2 Methodology

4.2.1 Study site

In the study area on the south coast of Havana province (chapter 2), seventy 10 m x 10 m plots were surveyed to address the ecological conditions in sites representative of major land-cover changes involving mangroves (chapter 3). The sites surveyed are characterized, for 1985 and 2005, by the covers: Mangrove-Mangrove, Mangrove-Flooded Mangrove, Herbaceous Swamp-Mangrove Plantation, Herbaceous Swamp-Mangrove, and Wetland Forest-Herbaceous Swamp.

4.2.2 Methods

The plots were located in sectors II to V in sites landward and seaward of the freshwater barrier (Figure 2.1 and Appendix 1), which is the major local source of change in wetlands and mangroves (see chapters 2 and 3). Mangroves in sectors outside the dammed areas will be addressed in Chapter 5 (sectors I and VI in Figure 2.1).

Seeking a more feasible documentation of manifestations of the resilience of mangroves to the barrier-induced flooding (i.e., evidence of recovery of mangrove vegetation), plots were arranged in seaward-landward transects in the three largest mangrove areas, which are located in sectors II, IV and V. The most extensive conversion of mangroves into flooded mangroves had occurred in sectors II and V (Figure 3.1).

Due to sometimes impossible access (rainy season) to the mangrove areas in sectors II, IV and V, and insufficient field staff, only the transect in sector II was comprehensively surveyed, i.e., the transect crossed the entire mangrove area from the seaward to the landward side of the mangroves, and all plots were surveyed in both the dry and the rainy season. In sector V, the transect did not reach the seaward side of the mangroves, and not all the plots were surveyed in both seasons. In sector IV, two transects were surveyed; each transect had only one plot landward and one seaward of the barrier, which were surveyed in both seasons.

The vegetation was surveyed in the dry season of 2005 (mainly during March-April); the abiotic factors were surveyed in both the dry (mainly March-April) and the rainy season (mainly September-October). The surveys allowed both a general characterization of wetland vegetation with a focus on mangroves (Berger et al. 2006; Cintrón and Schaeffer-Novelli 1984; Dahdouh-Guebas et al. 2004; Mueller-Dombois and Ellenberg 1974), and the ataining of basic information on water level, and soil salinity and redox potential as major components of abiotic seasonal change (Boto 1984; Hartig 2005; Inglett et al. 2005).

Water level refers to either the depth of flooding water or the water table; both were measured with a plastic ruler. The depth of the water table was found by drilling and waiting 30 minutes until the water level stabilized. Soil salinity at about 20 cm depth was measured with a hand-held refractometer (Atago ATC-S/Mill-E) from a soil sample extracted with a soil driller (Eijkelkamp) (Snedaker and Snedaker 1984). The soil redox potential was measured *in situ* at 1-50 cm depth (starting at 1 cm, and then every 5 cm for 5-50 cm) with a redox potential needle electrode and its corresponding Ag/AgCl reference electrode (Microscale Measurements). As a reference electrode was used, the measured values were corrected using the equations provided with the Microscale Measurements Redox equipment (DeLaune and Reddy 2005).

In all plots, the percentage of mangrove cover was visually estimated (Cintrón and Schaeffer-Novelli 1984; Mueller-Dombois and Ellenberg 1974). A more comprehensive vegetation survey was conducted in those plots arranged in the already mentioned transects of sectors II, IV and V. Besides the total vegetation cover (percentage), the cover per mangrove species was estimated. For each tree (diameter at breast height (DBH) > 3 cm) the species, DBH (with metric tape), and height (visual estimation) were recorded (Berger et al. 2006; Cintrón and Schaeffer-Novelli 1984; Dahdouh-Guebas et al. 2004; Mueller-Dombois and Ellenberg 1974).

The main data analysis includes graphs of water level, and soil salinity and redox potential landward and seaward of the barrier and outside the dammed areas in the dry and the rainy season; graphs were drawn of mangrove cover versus water level, and mangrove cover versus soil salinity in the dry and rainy season.

For the plots in the transects, the data of different plots were combined into summary vegetation profiles. The profiles allow recognition of phases of manifestation of resilience of the mangrove cover with regard to the barrier-induced flooding (i.e., evidence of recovery of mangrove vegetation). For each profile, graphs of water level and soil redox potential are included.

A diagram illustrates the anchoring of seedlings and development of saplings in conditions of increased flooding.

4.3 Results

4.3.1 Changes in ecosystem components corresponding to major land-cover changes

Water level, soil salinity (Figure 4.1) and redox potential (Figure 4.2) landward of the barrier deviate from the values measured seaward of the barrier and in the non-dammed wetlands.

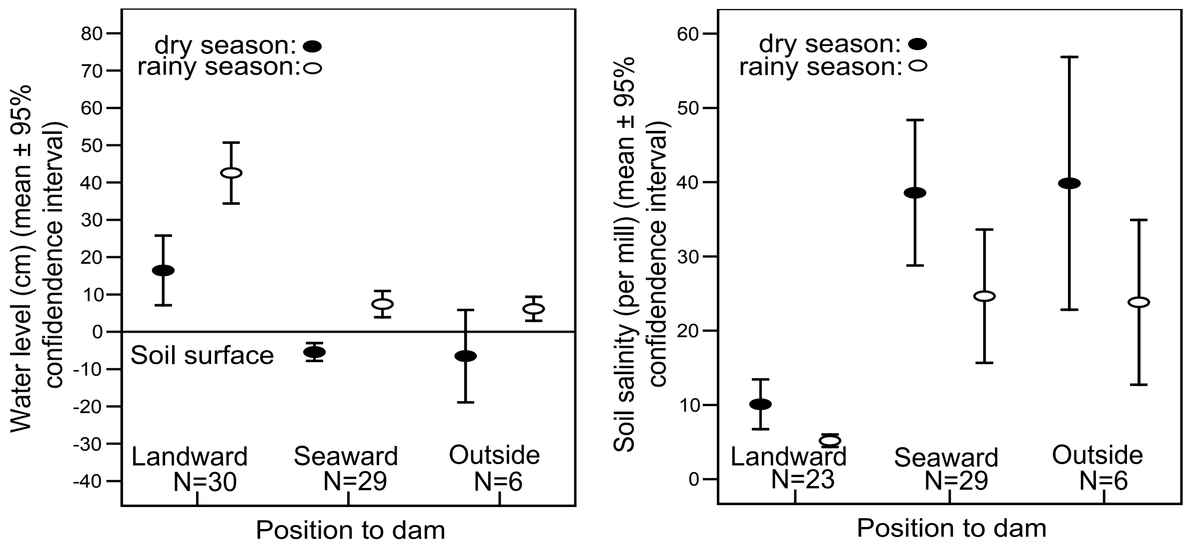


Figure 4.1 Water level and soil salinity (at 20-cm depth) in the dry and rainy season, landward and seaward of the barrier, and outside the dammed areas (N= number of plots)

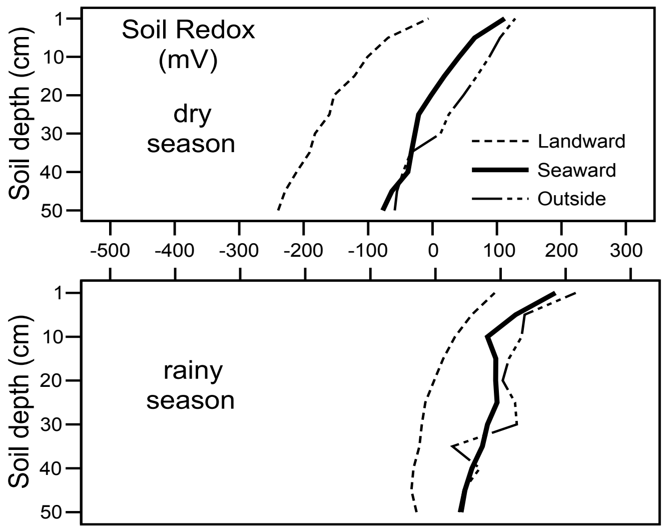


Figure 4.2 Soil redox potential at depth 1-50 cm in the dry and rainy season, landward (N=30) and seaward (N=29) of the barrier, and outside the dammed areas (N=6) (N= number of plots)

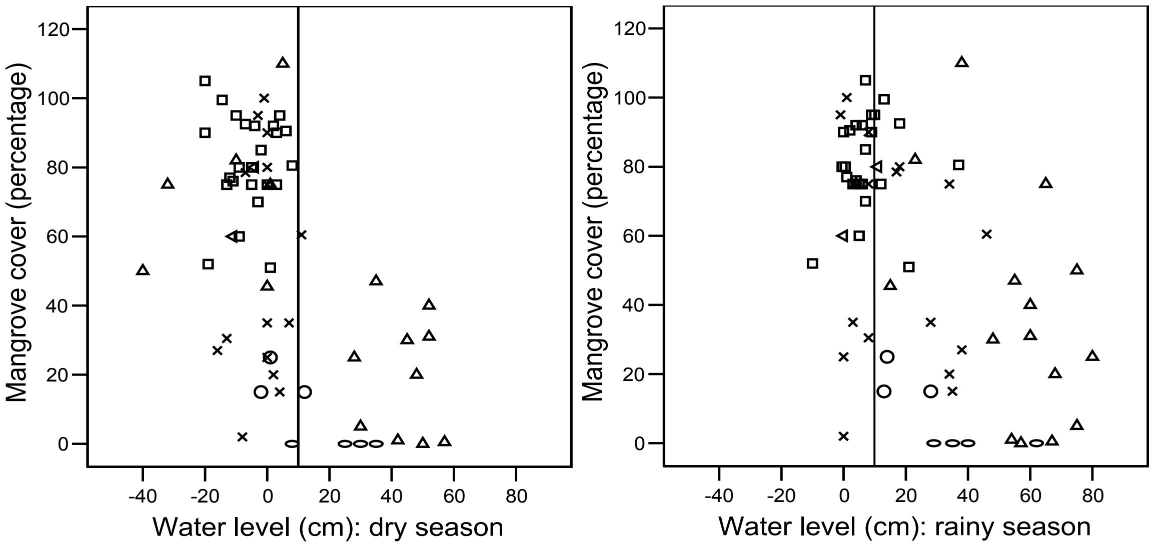
In 2005, wetlands landward of the barrier experienced more flooding (higher water level) than the seaward and non-dammed wetlands (Figure 4.1). In these wetlands, a seasonal flooding pattern predominates, i.e., the mean water level was below and above the soil surface, respectively, in the dry and rainy season (Figure 4.1). There are exceptions to that pattern, however. For instance, some wetlands outside the dammed areas, to be addressed in more detail in Chapter 5, experienced permanent flooding. This is indicated by the confidence interval of mean water level in the dry season, which includes values above the soil surface (Figure 4.1). Seaward mangroves also experienced natural permanent flooding, e.g. coastline mangroves located furthest seaward from the barrier, which are flooded by sea tides. Those coastline mangroves were not predominant in the areas seaward to the barrier and, consequently, the confidence interval of the mean water level of seaward wetlands does not reach values above the soil surface (Figure 4.1).

The difference between the mean water level of landward and seaward wetlands tends to be bigger in the rainy than in the dry season (Figure 4.1). In the rainy season, more freshwater enters the wetlands of the study area. Flow of water from landward to seaward wetlands not across the spillways of the barrier but over the barrier itself, was detected only in the case of extremely abundant rainfall in the rainy season (e.g., with tropical cyclones).

Soil salinity followed a pattern in which lower salinity corresponded to higher water level. The pattern was observed in all wetlands independent of their location with respect to the freshwater barrier, i.e., soil salinity decreased from the dry to the rainy season in all wetlands (Figure 4.1). This pattern was also observed in both the dry and the rainy season, i.e., in both seasons landward wetlands tended to have the highest water level and the lowest soil salinity (Figure 4.1).

Landward wetlands had the most reduced soils of all wetlands in both the dry and the rainy season as indicated by the more negative redox potential (Figure 4.2).

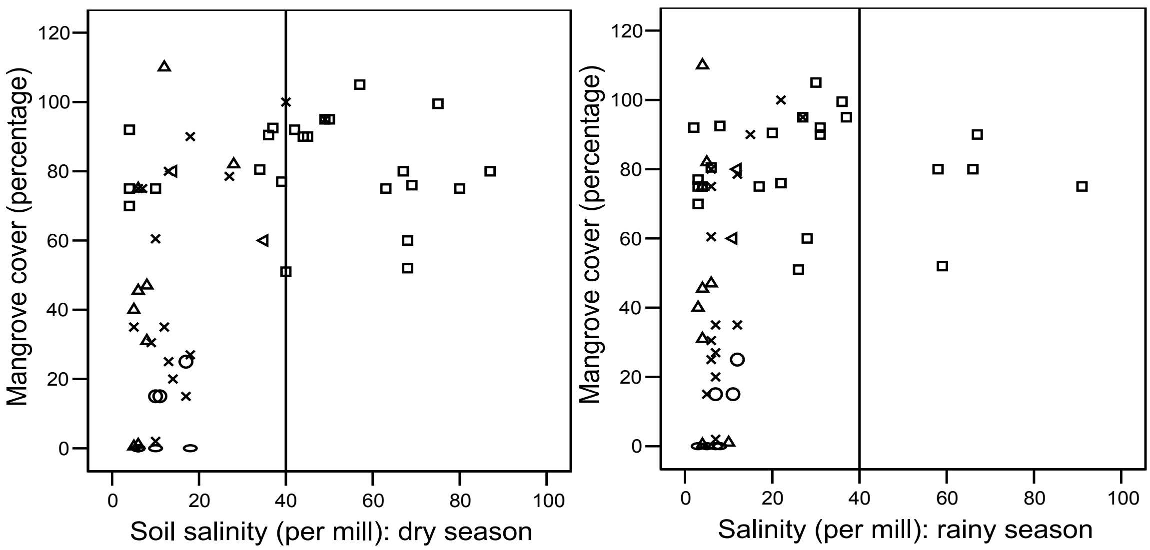
Most of the plots where mangroves were the characteristic land cover both before and after the construction of the barrier experienced flooding (water level higher than zero) during the rainy season but not during the dry season (Figure 4.3). Only a few of these plots remained flooded in the dry season and then always with a water level less than 10 cm. Mangrove plantations show this pattern too, i.e., those achieving mangrove cover higher than 60% are not flooded during the dry season (Figure 4.3). Plots where mangroves in 2001 had turned into flooded mangroves have the highest water level (Figure 4.3 and Figure 3.1), which is linked to the accumulation of water landward of the barrier (Figure 4.1).



Legend: Land-cover change in study plots from 1985 to 2001 (Number of plots).
□ Mangrove-Mangrove (24)
△ Mangrove-Flooded Mangrove (16)
× Herbaceous swamp-Mangrove plantation (16)
◁ Herbaceous swamp-Mangrove (2)
⬭ Herbaceous swamp-Herbaceous swamp (4)
○ Wetland Forest-Herbaceous swamp (3)

Figure 4.3 Mangrove cover vs. water level in the dry and rainy season (Reference line at water level 10 cm) (Symbols show land-cover changes from 1985 to 2001)

In contrast to the water level (Figure 4.3), mangroves with all levels of soil salinity showed a high mangrove cover of over 60% (Figure 4.4). This was in plots where mangroves were the characteristic land cover both before and after the construction of the barrier (1985 and 2001, respectively). The plots with the lowest mangrove cover had the lowest values of salinity during the dry and the rainy season (Figure 4.4), which was linked to the high water level (Figure 4.3).



Legend: Land-cover change in study plots from 1985 to 2001 (Number of plots).
□ Mangrove-Mangrove (24)
△ Mangrove-Flooded Mangrove (16)
× Herbaceous swamp-Mangrove plantation (16)
◁ Herbaceous swamp-Mangrove (2)
⬭ Herbaceous swamp-Herbaceous swamp (4)
○ Wetland Forest-Herbaceous swamp (3)

Figure 4.4 Mangrove cover vs. salinity in the dry and rainy season (Reference line at salinity 40 per mill) (Symbols show land-cover changes from 1985 to 2001)

4.3.2 Resilience of mangrove cover

The vegetation profiles representing the plots surveyed in the transects in sectors II, V and IV (respectively Figure 4.5, Figure 4.6 and Figure 4.7) illustrate the expected concentration of flooded mangroves landward to the barrier.

The transect in sector II (Figure 4.5) crossed the entire mangrove area seaward to landward. The main mangrove zones seaward of the barrier are defined as: *R. mangle* belt fringing the sea (profile H), *A. germinans* zone (profile G), and mixed zone (with *R. mangle*, *A. germinans* and *L. racemosa*) (profile F). Then comes the barrier and on the landward side (north) there are mangroves dominated by *R. mangle* (profile E), flooded mangroves with mixed lower cover (profiles D and C), and closed-canopy mangroves dominated by *R. mangle* (profile B), or *A. germinans* and *L. racemosa* (profile A) on the landward border of the mangroves towards pastures.

The transect in sector V (Figure 4.6) did not cross the entire mangrove area seaward to lanward, and started with an *A. germinans* zone immediately seaward of the barrier (profile H). Landward of the barrier, there were mixed flooded mangroves (with *R. mangle*, *A. germinans* and *L. racemosa*, profile G), flooded mangroves with low cover (profiles D1 and B) or no cover at all (profile F), and also more closed-canopy flooded mangroves dominated by *L. racemosa* (profile D) or *R. mangle* (profile C), or mixed (with all three local mangrove species and the pseudomangrove *C. erectus*, profile E). The most landward mangroves bordering pastures were abundant in *A. germinans* and *L. racemosa* (profile A).

In sector II and sector V, stumps of former mangroves were a remarkable element in profiles landward of the barrier. They were particularly abundant and noticeable in the flooded mangroves without closed canopy (e.g., profiles C and D in Figure 4.5; and B, D1 and F in Figure 4.6), which basically consisted of open water from which stumps emerged. The stumps were usually at densities of 7 to 14 per 100 m^2 with diameters of 4 to 22 cm and were the result of dieback due to the barrier and *ad hoc* management, i.e., mangrove stems were cut down and removed as reported by IGT (1999) (e.g., for firewood in bakeries). Stumps were more abundant in sites closer to the barrier (profiles D in Figure 4.5, and F in Figure 4.6), where transport of the stems out of the area would have been easier.

The two transects in sector IV (Figure 4.7) did not cross the entire mangrove area from searward to lanward. Transect 1 had dwarf mangroves both seaward and landward of the barrier (profiles 1A and 1B, respectively). In transect 2, there was an *A. germinans* zone immediately seaward of the barrier (profile 2B), and a mixed mangrove zone on the landward side (with *R. mangle*, *A. germinans* and *L. racemosa*, profile 2A).

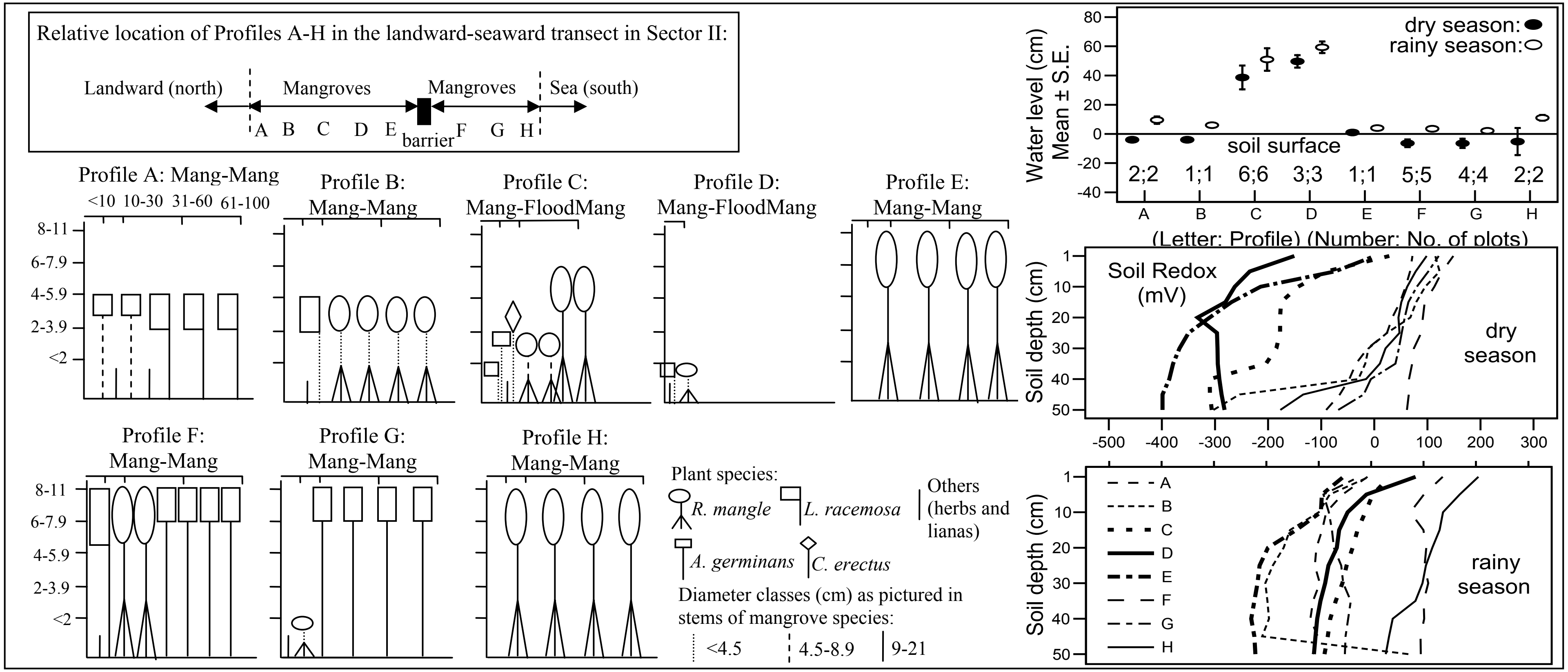


Figure 4.5 Vegetation, water level and soil redox potential in transect in sector II. Land-cover characterizing the site in 1985 and 2001: Mang-Mang = mangrove-mangrove, Mang-FloodMang = mangrove-flooded mangrove). Vegetation variables: plant height (in meters; y-axis of profiles), total vegetation cover (in percentage; top axis of profile; numbers as indicated in profile A), vegetation cover per species (same ranges as for the total vegetation cover: each range represented by one, two, three or four repeated symbols of species), plant species and species diameters. Each profile is a summary of the respective field-plots (Appendix 2)

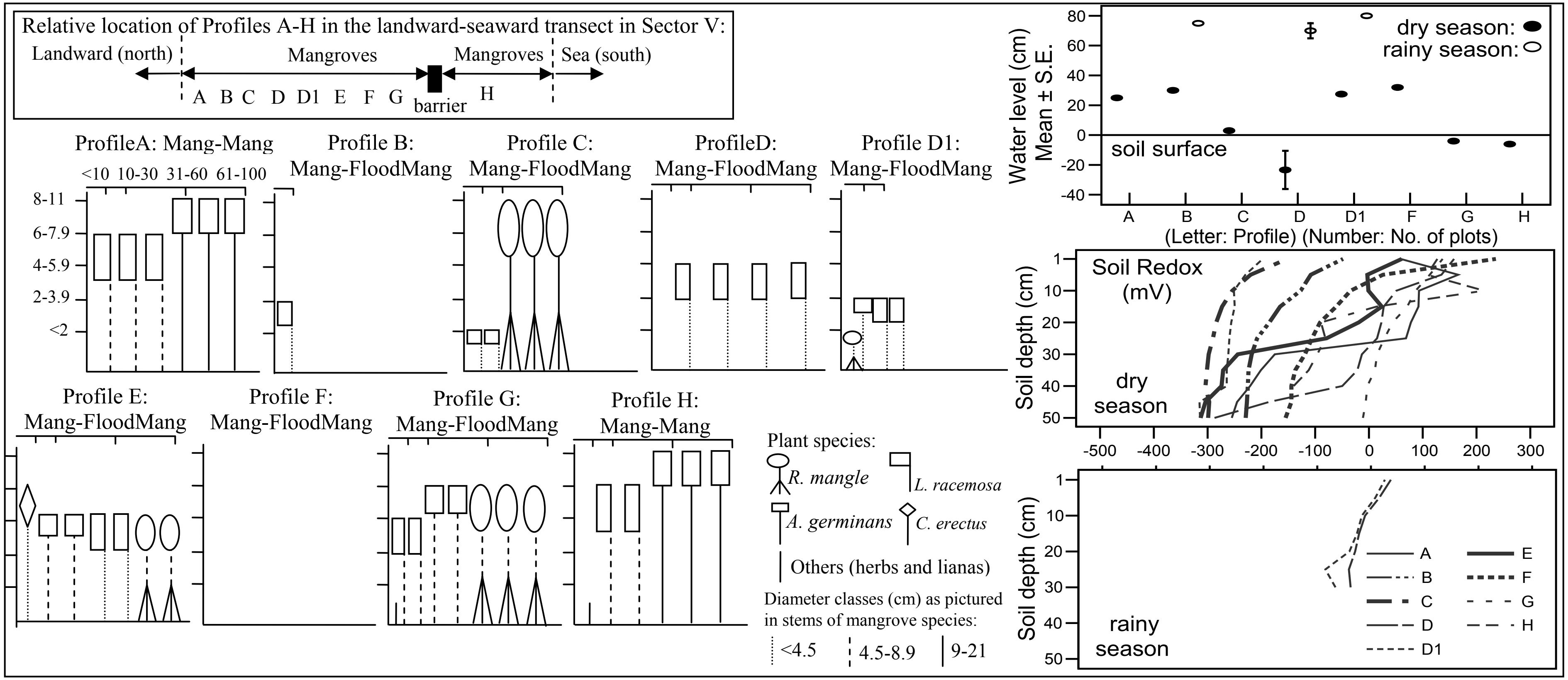


Figure 4.6 Vegetation, water level and soil redox potential in transect in sector V. Land-cover characterizing the site in 1985 and 2001: Mang-Mang = mangrove-mangrove, Mang-FloodMang = mangrove-flooded mangrove). Vegetation variables: plant height (in meters; y-axis of profiles), total vegetation cover (in percentage; top axis of profile; numbers as indicated in profile A), vegetation cover per species (same ranges as for the total vegetation cover: each range represented by one, two, three or four repeated symbols of species), plant species and species diameters. Each profile is a summary of the respective field-plots (Appendix 3)

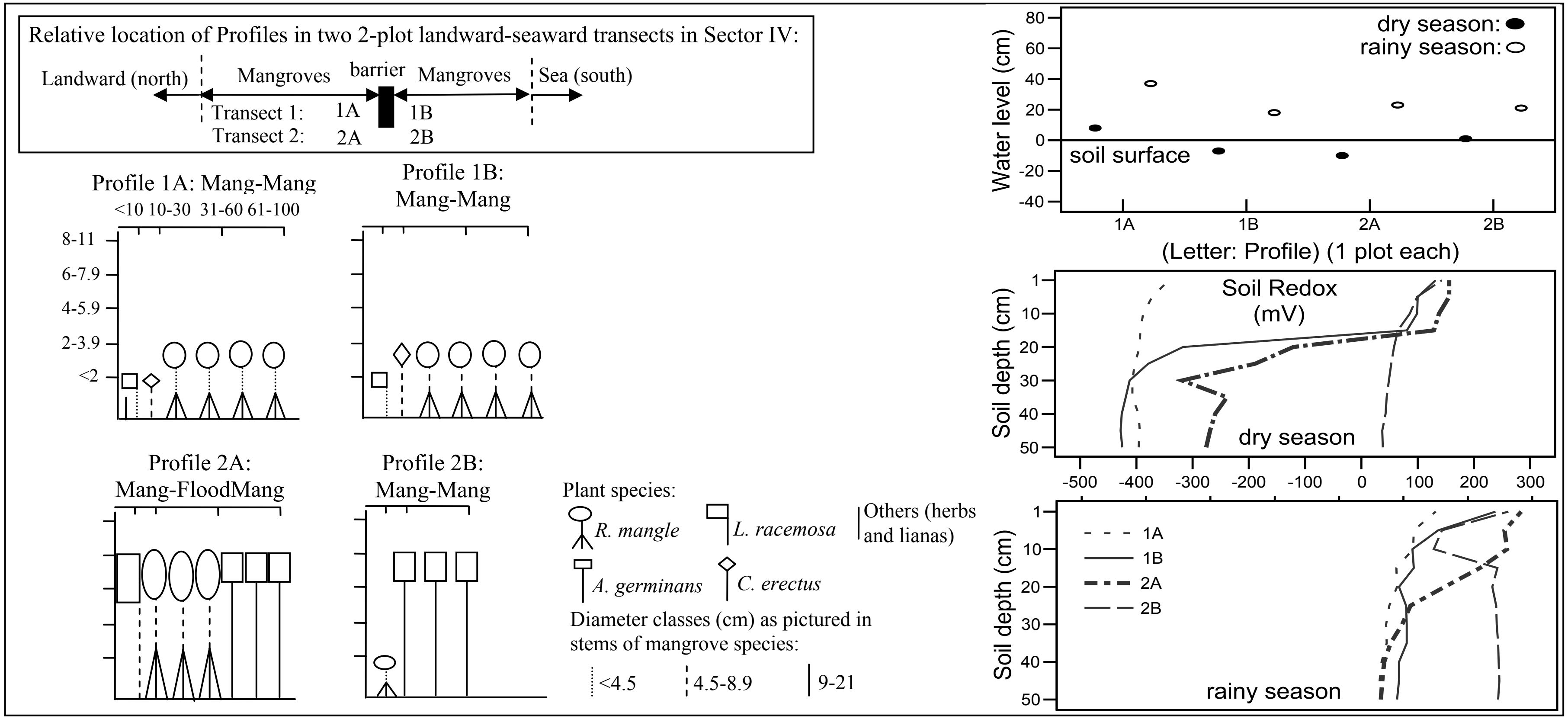


Figure 4.7 Vegetation, water level and soil redox potential in two transects in sector IV. Land-cover characterizing the site in 1985 and 2001: Mang-Mang = mangrove-mangrove, Mang-FloodMang = mangrove-flooded mangrove). Vegetation variables: plant height (in meters; y-axis of profiles), total vegetation cover (in percentage; top axis of profile; numbers as indicated in profile A), vegetation cover per species (same ranges as for the total vegetation cover: each range represented by one, two, three or four repeated symbols of species), plant species and species diameters. Unlike sector II (Figure 4.5) and sector V (Figure 4.6), in sector IV each profile represents a single field-plot (Appendix 4)

Sector II

In sector II, three groups of profiles can be distinguished. The first group has the profiles A, F, G and H. The second group has the profiles C and D. The third group has the profiles B and E.

The profiles A, F, G and H in the first group represent sites where mangroves were the characteristic land cover before and after the construction of the barrier as found with the remote sensing supported analysis (Figure 4.5). The water levels in these mangroves do not tend to remain the highest in any season in the entire transect (Figure 4.5). The only landward mangroves in this group of profiles (profile A) are those located furthest from the freshwater barrier, on the landward border of the coastal wetland belt of the study area.

Profile H is distinctive in that it is the only profile representing mangroves located on the coastline (Figure 4.5). Those mangroves, *R. mangle*-dominated and naturally flooded under the direct influence of sea tides, are the only mangroves in the first group of profiles where the curve of soil redox potential has more negative values in the dry season than in the rainy season. Profiles A, F and G are located more landward than profile H and tend to have either more positive redox potential values (more aerobic conditions) during the dry season when the water level is lower (profiles A, G), or similarly positive values in both seasons (profile F).

In the second group (profiles C and D), the profiles represent sites where the characteristic land-cover changed from mangroves into flooded mangroves after the barrier was constructed. In 2005, the vegetation cover was low (10-60 %) and was dominated by mangroves (Figure 4.5).

The profiles B and E, in the third group, are similar to those of the first group in the sense that mangroves were the characteristic land cover not only before the construction of the barrier but also in 2005 (Figure 4.5). Profile B represents sites that in 2005 had a continuous mangrove canopy, and which were located more inland than the mangroves represented in profile E. Those mangroves are spatially restricted to patches that in 2005 rarely exceeded 30 m x 30 m and were typically surrounded by open water.

The third group of profiles (profiles B and E), in general has higher tree diameter and height, and lower water level than the second group (profiles C and D). One profile of the third group (profile E) shares the pattern of soil redox potential with both profiles of the second group (profiles C and D), i.e., more negative values occured during the dry season, which is typical of the permanently flooded wetlands located landward to the barrier. In contrast, the soil redox curve of profile B in the third group is more similar to the soil redox curve of the seasonally flooded sites of the first group (profiles A, F and G).

Sector V

The patterns found in the transect of sector V are in general similar to those of sector II. The transect in sector V (Figure 4.6) could not be completely surveyed in both seasons as mentioned in section 4.2.2.

The most distinctive feature of sector V compared to sector II is a stronger presence of *L. racemosa* and *A. germinans* in the vegetation recovery of flooded mangroves (profiles B, D, D1, E and G in Figure 4.6), which in profile D have led to a monospecific *L. racemosa* cover.

Also distinctive in sector V is the absence of plants in the sites represented by profile F (Figure 4.6). The water level in the dry season (about 30 cm) is only slightly higher than in the other flooded mangroves (profiles B and D1). The soil redox potential of profile F is not more negative than that of the other flooded mangroves (profiles B and D1).

Similar to sector II, in sector V the lowest percentage of vegetation cover was observed in flooded mangroves with the highest water level and soil redox curves typical of permanent flooding conditions (profiles B, D1 and F). As in sector II (profile C; Figure 4.5), sites in sector V with well established mangrove canopies (up to 100% vegetation cover) that were not permanently flooded either have extreme negative values over the entire soil redox curve in the dry season (profile C; Figure 4.6), or not so negative values (profiles D, E, G and H; Figure 4.6).

Sector IV

In the two transects in sector IV (Figure 4.7), the findings do not contradict the findings made in the more intensively surveyed transects of sectors II and V (Figure 4.5 and Figure 4.6).

In sector IV, where the change of mangroves into flooded mangroves involved a smaller area than in sectors II and V and without dieback of mangroves, the water level was not as high as in sectors II and V, neither in the dry nor in the rainy season (Figure 4.5 and Figure 4.6). The dwarf mangroves in transect 1 (*R. mangle*-dominated) have smaller diameters landward of the barrier, where the most negative soil redox values of all transects in sectors II, V and IV were measured in the dry season.

Transect 2 of sector IV (Figure 4.7), in the dry season was the only transect in the study area where the water level in a seaward mangrove plot (profile 2B) was higher than in the landward flooded mangrove plot (profile 2A). That water level in the dry season (1 cm above the soil surface) would indicate conditions close to permanent flooding. However, the soil redox curve was more typical of non-permanently flooded sites, i.e., the soil redox potential did not show extreme negative values in any season. The sampled water level might not be representative of the plot and its surroundings, e.g., it may have been an exceptionally high water level measured in an exceptional depression.

4.3.3 Early mangrove establishment: anchoring of seedlings and development of saplings in increased flooding

In conditions of increased flooding and very low vegetation cover like in sectors II and V (e.g., Figure 4.5 profile D, and Figure 4.6 profiles B and F), dead stems and stumps of former mangroves provide important not permanently flooded spaces where seedlings and saplings involved in the natural regeneration of the mangrove cover can establish (Figure 4.8).

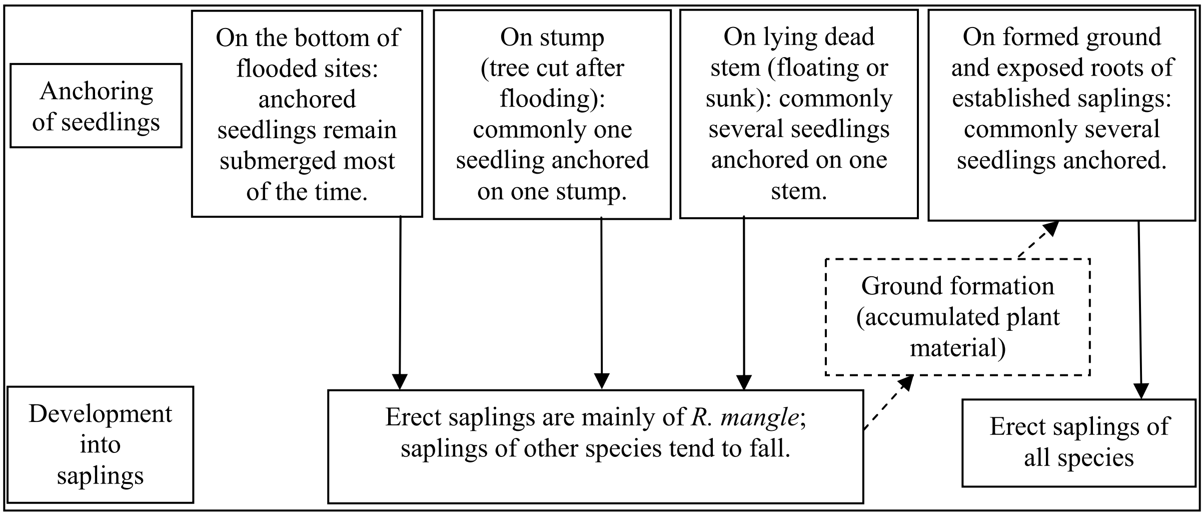


Figure 4.8 Anchoring of mangrove seedlings, and pseudo-mangrove *C. erectus*, and development into saplings in conditions of increased flooding and low vegetation cover (sectors II and V)

Seedlings anchor more frequently on stumps of former mangrove trees than on the bottom of flooded sites and lying dead stems (Figure 4.8). This is valid for seedlings of all mangrove species (*R. mangle*, *L. racemosa*, *A. germinans*) and the pseudo-mangrove species *C. erectus*. For seedlings anchored directly on the bottom of flooded sites, *R. mangle* was more frequently observed than *L. racemosa* and *A. germinans.* Seedlings of *C. erectus* were never observed anchoring directly on the bottom of flooded sites.

Only *R. mangle* saplings grow upright when rooting directly on the bottom of flooded sites, stumps or lying dead stems. Saplings of all other species tend to fall after the seedling stage. Those seedlings and saplings that do not develop, or die, provide plant material (stems, leaves, roots) and form elevated and non flooded ground where seedlings of all species including *R. mangle* establish and develop into erect saplings.

4.4 Discussion

4.4.1 Changes in ecosystem components corresponding to major land-cover changes

Due to the freshwater barrier, the landward wetlands differ from the seaward wetlands and wetlands outside the dammed areas. Wetlands landward of the barrier had become permanently flooded, and correspondingly had lower salinity and more reduced soils. The barrier had caused a distinctive functioning in the landward wetlands. For instance, the accumulation and partial stagnation of water in these wetlands in the dry season led to a considerable accentuation of the anaerobic and reduced conditions of these wetland soils (e.g., soil redox potential predominantly lowered to the range from 0 to -250 mV) (Hurt 2005; Lewis 2001).

The permanent increase in water level caused by the freshwater barrier prevents the complete recovery of the mangrove cover landward of the barrier. Other authors have documented the importance of understanding that mangroves are seldom permanently flooded, and that increased flooding due to natural phenomena (mainly tropical cyclones) and anthropogenic causes (e.g., diking) can kill existing mangroves, or make mangrove restoration fail (Cahoon and Hensel 2002; Capote-Fuentes and Lewis 2005; Doyle and Girod 1997; Lewis 2005).

Only in seaward wetlands and wetlands outside the dammed areas is seasonal flooding still common. However, high soil saturation due to a water level close to the soil surface in the rainy and the dry season is characteristic in all wetlands of the study area. This is because the study area, located in the coastal section of a watershed whose water predominantly flows towards the coastline, is one of the lowest coastal areas in Cuba with respect to the sea level (Boto 1984; Hartig 2005; Lewis 2001). Also, this coast is in the Cuban region with clearest evidences of sea level rise (IGT 1999).

The similarity between the wetlands seaward and those outside the dammed areas does not mean that the seaward wetlands are not impacted. IGT (1999), for instance, reported that the blockage of water landward of the barrier has disrupted the transport of sediments to the seaward wetlands and accelerated coastline erosion.

Despite the marked differences imposed by the barrier, the wetlands in the study area, independent of their location with respect to the barrier, still showed seasonal changes. These changes between the dry and the rainy season were indicated by changes in water level, soil salinity and redox potential. Since the occurrence of seasonal changes is a characteristic of the natural functioning of wetlands, the persistence of such changes in the study area indicates that if the freshwater barrier could be better managed, especially regarding diminishing the water level in the landward wetlands, then wetland vegetation (including mangroves) could recover further (Cintrón and Schaeffer-Novelli 1983; Hartig 2005; Lugo and Snedaker 1974; Middleton 2002; Tomlinson 1986; Romigh et al. 2006).

A major warning for management of the water level of the flooded landward wetlands is that trying to lower it to the former level below the soil surface can render the soils extremely acid and prevent the development of any vegetation, both natural and cultural, for at least four to five decades. Thomas et al. (2004) report soil pH above 7 to have been lowered to below 2.5 in coastal south Australia, where seawater was excluded in 1954 when a bund wall was constructed for industrial and agricultural land reclamation. Other examples exist for South East Asia and Africa (Boto 1984).

Such acidic soils are called acid sulphate soils (Boto 1984; Siciliano and Germida 2005). They could develop in the landward wetlands in the study area, where the long-term flooding caused by the barrier leads to highly reduced soils (redox potentials less than -100 mV). In such reduced wetland soils, the conversion of sulfate ($SO_4^{\ 2-}$) into sulphide ($S^{\ 2-}$) is possible, a pre-condition for acidic soils to develop (Boto 1984; Hartig 2005; Inglett et al. 2005). Iron pyrite (FeS_2) must also be present. The required concentration of iron is probably provided by the red ferralitic soils that predominate in the agricultural area landward of the wetland belt. If the flooded wetland soils containing sulphide and pyrite become aerated again, as it would be if the water level of landward wetlands is taken to former level below soil surface, then chemical reactions can produce sulphuric acid and thus very acidic soils (Boto 1984).

4.4.2 Resilience of the mangrove cover

The construction of the barrier did not transform all landward mangroves into flooded mangroves. Mangroves seaward of the barrier, and landward of the barrier in sites located on the landward border of the coastal wetland belt of the study area, were the characteristic land cover before and after the construction of the barrier. Therefore, the mangroves existing in those sites in 2005 do not correspond to recovery of the mangrove cover from the barrier-induced flooding. In these mangroves, the soils were not highly reduced, as it would be the case if permanent or long-term flooding had occurred. Predominantly, these mangrove soils were only moderatedly reduced to reduced (i.e., soil redox potential mainly in the range from -100 to 200 mV). The dominant mangrove species (*A. germinans* and *L. racemosa*) are typical of non-permanent flooding, which fits the non-permanent flooding indicated by the mean water level found respectively below and above the soil surface in the dry and the rainy season, respectively.

In some of the non-transformed mangroves, the increased landward flooding could have further stressed *R. mangle*-dominated dwarf mangroves and led them in 2005 to have smaller diameters (< 4.5 cm) than the seaward dwarf mangroves (4.5-8.9 cm). Those landward dwarf mangroves in the dry season of 2005 showed the most anaerobic and reduced soils found in any of the wetlands of the study area (Feller et al. 2002; Kozlowski 1997; Krauss et al. 2006).

Two phases of manifestation of resilience of the mangrove cover to the barrier-induced flooding (i.e., evidence of recovery of mangrove vegetation) were recognizable in 2005. Mangroves which had been turned into flooded mangroves by the construction of the barrier were in 2005 either still flooded mangroves, or had already recovered in the sense that mangroves were again the characteristic land cover. Those two phases can be called early/ongoing recovery, and advanced recovery.

The recovery, i.e., natural regeneration of mangroves, starts in open water surrounded by mangroves. The percentage of mangrove cover proceeds towards higher values (until 100%) and mangrove patches surrounded by open water appear. These patches, when expanding in area, can form a continuum mangrove canopy with mangroves sites which had never been transformed into flooded mangroves by the barrier-induced flooding.

In the regenerated mangrove patches, with the development of mangrove plants from seedlings into saplings and trees, roots and accumulated plant material contribute to form new ground. This means that vegetation recovery progressively promotes change from permanent flooded ground (below the water surface), to elevated ground (above the water surface) but still with characteristics of permanent flooding due to the influence of permanently flooded surroundings, to actually seasonally flooded ground without direct influence of flooded surroundings, e.g., soil redox potential becomes more positive, as typical of less flooded soils (Boto 1984; Lewis 2001).

Although the three local mangrove species and the pseudomangrove *C. erectus* were present in both the early/ongoing and the advanced recovery of the vegetation cover, *R. mangle* and *L. racemosa* tended to dominate.

The dominance of *R. mangle* was most likely favored by a differential dieback of mangrove trees caused by the initial sudden increase in flooding associated with the barrier construction. Trees of *A. germinans* and *L. racemosa* will have died more frequently. The respiratory structures in the aerial roots of these two species commonly develop above the soil surface up to a height of 40-60 cm. Thus, such structures would have become permanently covered by the new and higher landward water level (> 40 cm) and respiration completely prevented**.** In *R. mangle*, such structures commonly develop up to a higher position (around 2 m) in the aerial prop roots (Kozlowski 1997; McKee 1995; McKee and Mendelssohn 1987; McKee et al. 1988; Tomlinson 2006).

In sites where *R. mangle* dominated, differences in tree development should be influenced by different starting conditions. Sites where mangrove recovery did not start from near-to-zero mangrove cover but started from existing *R. mangle* trees, typically found along channels before the barrier was built (IGT 1999), in 2005 had taller trees with larger diameters (8-11 m and 9-21 cm). In contrast, mangrove recovery starting from open water until 2005 produced a closed canopy (100% vegetation cover), but trees were smaller and had smaller diameters (4-5.9 m and < 4.5 cm).

The dominance of *L. racemosa* in mangrove vegetation recovering in the more landward flooded mangroves was probably influenced by a lower relative importance of *R. mangle* in the vegetation structure of those sites prior to the construction of the barrier. The later species is naturally less frequent in such more landward located mangroves (Cintrón and Schaeffer-Novelli 1983; Menéndez 2000; Tomlinson 1986).

Differences in dominance of mangrove species in the recovered mangrove cover in some of the flooded sites were most likely influenced by the characteristics of mangrove seedlings. Mangrove seedlings are buoyant and water dispersed, and so the longer seedlings of *R. mangle* can reach the bottom of flooded sites more easily and anchor directly in such flooded sites with a higher water level (Ball 1980; Jiménez 1985a, b; Jiménez and Lugo 1985; Krauss et al. 2006; McKee 1995).

In sites where *L. racemosa* and *A. germinans* are more frequent in the recovered mangrove vegetation, a lower water level will allow the establishment of seedlings. These seedlings, typically reported in the literature as 1.5 and 3 cm long respectively, are much shorter than those of *R. mangle*, which are typically reported as 10-60 cm long (Ball 1980; McKee 1995; Menéndez 2000; Tomlinson 1986).

Given that the water level increased due to the barrier, the anchoring of seedlings on the not permanently flooded surface of the stumps of former mangrove trees, which remain above the water surface, is critical for the regeneration of the mangrove cover.

Rhizophora mangle seedlings have the advantage over seedlings of the other mangrove species and *C. erectus* in that they develop into erect saplings when anchoring directly on the bottom of flooded sites, or on stumps or lying dead stems. While saplings of all other species tend to fall when developing further from the seedling stage, the prop roots of *R. mangle* promote structural stability and allow exploring nutrient and water uptake directly in the flooded ground (Jiménez 1985a; Tomlinson 1986). Other seedlings and saplings, which cope less well with extreme permanent flooding, find more suitable conditions for developing into saplings when establishing in elevated ground formed by accumulated plant material (Cintrón and Schaeffer-Novelli 1983; Jiménez 1985a, b; Jiménez and Lugo 1985; McKee 1995; Tomlinson 1986).

Explaining the lack of mangrove regeneration in some sites, which could be surveyed only in the dry season, will demand surveying these sites in the rainy season. In these sites, the conditions due to flooding are not clearly more limiting for mangrove regeneration than in other flooded sites where such regeneration did occur. Where mangrove regeneration was lacking, the water level in the dry season (32 cm) was not much higher than where regeneration occurred (27-28 cm) and should most likely not prevent the establishment of at least *R. mangle* seedlings (Jiménez 1985a; McKee 1995). Also, the soils were not more anaerobic and reduced than where regeneration occurred.

That lack of regeneration can be related to limited seedling establishment or survival, because the sites where mangrove regeneration had not occurred belonged to the most extremely flooded wetlands of the study area. These wetlands showed the highest seasonal fluctuation of mean water level in the study area, i.e., they had the highest mean water level in the rainy season (52 cm), and the greatest difference between the mean water level of the rainy and the dry season (mean water level is 9 cm in the dry season).

Those extreme high water levels in the sites with no mangrove regeneration can make trees stumps, which have a key role in mangrove regeneration as discussed above, ineffective for seedling establishment. That lack of mangrove regeneration due to increased water level does not contradict but warns about the results of Krauss et al. (2006). These authors concluded that slightly increased long-term flood durations associated with hydrological rehabilitation may have little effect on neotropical mangrove seedlings or saplings if imposed over long time frames (i.e., measured in months). That conclusion can be always valid for mangrove physiology, the focus of Krauss et al. (2006). However, the results of the present study indicated that slightly increased long-term flood durations can also prevent the anchoring of mangrove seedlings, and thus completely prevent mangrove recovery or rehabilitation.

4.5 Conclusions

The mangrove cover has showed resilience (i.e., had partially recovered) to the flooding caused by the freshwater barrier (permanently increased water level). In the former mangroves flooded and turned into open water by the construction of the barrier, regenerated mangrove patches tended to form a continuous mangrove canopy and thus mangroves again became the characteristic land cover. All local mangrove species (*Rhizophora mangle*, *Avicennia germinans*, *Laguncularia racemosa*) and the pseudo-mangrove *Conocarpus erectus* were present in the recovered mangrove vegetation. Based on the capacity of the local mangroves to recover, restoration and management should support that capacity and thus further diminish the extension of mangroves flooded by the freshwater barrier.

If wetlands and mangroves transformed by the barrier are to be restored or actively managed, a comprehensive view should be taken, which was lacking when the barrier was built. Decreasing the water level towards non-permanent flooding can cause extreme soil acidification and thus prevent the establishment of natural and cultural vegetation for at least 4-5 decades. Also, in the flooded mangroves, the local population has developed intensive fishing, the importance of which has not yet been assessed.

5 CHANGES IN MANGROVES OUTSIDE THE DAMMED AREAS

5.1 Introduction

Natural and anthropogenic events like hurricanes and building of infrastructure cause change in mangrove areas, for example in extension and plant species composition (Ellison 2000; Jiménez et al. 1985; Lugo 1997). Those events modulate the trajectories of mangrove vegetation from early successional to mature stages, and so contribute to shaping the physiognomy and zonation patterns of mangrove ecosystems (Ball 1980; Berger et al. 2006; Fromard et al. 1998; Lugo 1980; McKee 1993).

When anthropogenic events are clearly identifiable as the major cause of impacts on mangroves, then comparing the impacted areas to others not subjected to the same anthropogenic events is a useful study strategy (Ball 1980; Berger et al. 2006; Ellison 2000; Lewis 2005). Following that approach for the south coast of Havana province would lead to compare wetlands and mangroves in the dammed part of the coast (see Chapters 2, 3 and 4) with those located outside the dammed areas.

A research alternative to the comparison of different impacted and non-impacted mangroves is to compare data from the same area, which then must be surveyed in times distant enough as to be relevant for the studied subject. In studies of mangrove vegetation involving mature forest stages, at least one decade is commonly necessary to observe relevant changes. Availability of historic data is rather the exception, but for a section of the south coast of Havana province, specifically that area located immediately west of the dammed wetlands (sector I, Figure 2.1), such reference data exist thanks to field plots established in the 1980s by Leda Menéndez et al. (Menéndez 2000). The four plots they surveyed were relocated during the present study; the permanent tagging of trees reported by Menéndez (2000) is still in place to a considerable extent.

Besides normal limitations of using data generated by different studies (e.g., due to different sampling strategies), the scarcity of long-term (decades) empirical data on mangrove vegetation makes those 1980s data reported by Menéndez (2000) of high value as a source of comparison for other mangrove areas. It is especially the case for the Gulf of Mexico and the Caribbean because of the widely spread natural features of mangroves, e.g., coastal karstic geology and plant species composition (Suman 1994;

Yáñez-Arancibia and Lara-Domínguez 1999), shared natural influences like tropical storms (Doyle et al. 2003; Smith et al. 1994; Snedaker 1995) and most important, major management challenges and opportunities in the tradeoffs between coastal development and conservation (Capote-Fuentes and Lewis 2005; Rivera-Monroy et al. 2004; Yáñez-Arancibia and Day 2004).

The objective of this chapter is to document the status of mangrove vegetation in the two areas located outside wetlands dammed by a freshwater barrier built in 1985-1991 on the south coast of Havana province. For the entire region, land-cover changes involving wetlands and mangroves have been reported (see Chapter 3), as well as manifestations of resilience of the mangrove cover in the dammed areas (see Chapter 4). In the present chapter, trends of change in the two non-dammed areas are interpreted from the current structure of the mangrove vegetation. For one of the non-dammed areas, the analysis is supported by comparing data surveyed in 2005 with data available from the 1980s (reported by Menéndez 2000). As an example of the usefulness of the data available in the study area, a recent theoretical framework (Berger and Hildenbrandt 2003) for a long-time unsolved problem in ecology (since 1933), i.e., the biomass-density trajectories of self-thinning, was tested (Shaw 2006).

5.2 Methodology

5.2.1 Study site

In the study area on the south coast of Havana province, mangroves are reported as the natural coastal vegetation outside the dammed areas both west and east of the dammed areas (sector I and VI respectively, Figure 2.1) (Menéndez 2000). Correspondingly, remote sensing diagnosed mangroves as the characteristic land cover of these sectors for 1985 and 2001 (see Chapter 3). In mangroves west of the dammed areas (sector I), direct anthropogenic impacts (e.g., via logging) have been limited at least since the 1980s by a local coast guard station that regulates access of persons to these mangroves. Mangroves east of the dammed areas (sector VI) are also close to a coast guard station. Nonetheless, less regulated access to the eastern mangroves than to the western mangroves has been promoted, because in sector VI the most highly populated coastal town in the study area is located and a beach considered among the best (Figure 2.1) (IGT 1999; Menéndez 2000; Hernández 2006).

5.2.2 Methods

Two transects were surveyed in mangroves outside the dammed areas, one west of the dammed areas in sector I, the other east of the dammed areas in sector VI (Figure 2.1). Each transect had 3 plots of 10 m x 10 m. Hereafter, the plots of each transect will be named after the numbering of the sector in which the transect was placed. In sector I (western mangroves) the plots are named I1, I2 and I3. In sector VI (eastern mangroves), the plots are named VI1, VI2 and VI3 (Appendix 5).

Both transects started on the seaward side of mangroves in a landward direction. Each plot was placed in a different mangrove zone, recognized in the field by taking into account species composition and vegetation structure (Cintrón and Schaeffer-Novelli 1984). For the transect in sector I, in order to compare the surveyed data with available data, the plots were placed in the 20 m x 50 m plots surveyed in the 1980s (Menéndez 2000).

The vegetation was surveyed in the dry season of 2005 (mainly during March-April); the abiotic factors were surveyed in both the dry (March-April) and the rainy season (September-October). The surveys allowed both a general characterization of mangrove vegetation (Berger et al. 2006; Cintrón and Schaeffer-Novelli 1984; Dahdouh-Guebas et al. 2004; Mueller-Dombois and Ellenberg 1974), and collection of basic information on water level, and soil salinity and redox potential as major components of abiotic seasonal change (Boto 1984; Hartig 2005; Inglett et al. 2005).

The total vegetation cover (percentage) was estimated, as well as the cover per mangrove species and for mangrove saplings (DBH < 3 cm) and seedlings (typical two-leaf stage once the seedling has anchored) (Tomlinson 1986). For each tree (diameter at breast height (DBH) > 3 cm), the species, DBH (with metric tape), and height (visual estimation) were recorded. Standing stems of dead mangrove trees were also recorded (species, DBH, height); it was not always possible to identify the corresponding species because features relevant for the identification were absent due to decomposition (e.g., tree architecture, leaves). The cover of aerial roots of the mangrove species *A. germinans* and *L. racemosa* was estimated with the same method that as for the vegetation cover, which was possible due to the vertical growth pattern of these roots (different from the prop roots of *R. mangle*) emerging from the soil in an understory mostly free of plants (Berger et al. 2006; Cintrón and Schaeffer-Novelli 1984; Dahdouh-Guebas et al. 2004; Mueller-Dombois and Ellenberg 1974).

Water level refers to either the depth of flooding water or the water table; both were measured with a plastic ruler. The depth of the water table was found by drilling and waiting 30 minutes until the water level stabilized. Soil salinity at about 20 cm depth was measured with a hand-held refractometer (Atago ATC-S/Mill-E) from a soil sample extracted with a soil driller (Eijkelkamp) (Snedaker and Snedaker 1984). The soil redox potential was measured *in situ* at 1-50 cm depth (starting at 1 cm, and then every 5 cm for 5-50 cm) with a redox potential needle electrode and its corresponding silver/chloride (Ag/AgCl) reference electrode (Microscale Measurements). As a a reference electrode was used, the measured values were corrected using the equations provided with the Microscale Measurements Redox equipment (DeLaune and Reddy 2005).

The main data analysis includes comparison of the vegetation of the plots in a comparative way using vegetation profiles, for which the vegetation data were ranked. The profiles were complemented with graphs addressing water level, and soil salinity and redox potential in the plots in the dry and the rainy seasons.

For the transect in sector I, the vegetation data surveyed in 2005 was compared with data surveyed in the 1980s (Menéndez 2000). The height of trees (m), tree density (trees per 100 m^2) and basal area (m^2 per ha) obtained from both surveys were included in a single table. Two other tables respectively address the diameter range of trees, and the number of trees the present survey (2005) found tagged by the 1980s survey of Menénedez (2000).

The tree density (trees per 100 m^2) and basal area (m^2 per ha) were kept in different units in order to facilitate comparison of the data on the number of trees obtained by the 2005 and 1980s surveys. The tree density reported for the 1980s as trees per ha (Menéndez 2000) was converted into trees per 100 m^2. Correspondingly, the basal area obtained from the survey in 2005 was converted into m^2 per ha after summing up the basal area of all trees (DBH > 3 cm). The basal area of each tree was calculated as 3.14 x $(DBH)^2$ / 4 (Cintrón and Schaeffer-Novelli 1984).

The comparison of data from 2005 and the 1980s included testing the existence of a single linear segment in the biomass trajectory of forests according to the theoretical framework proposed by Berger and Hildenbrandt (2003). The biomass-density trajectory of forests relates to the linearity and slope of the part of such trajectory where self-thinning occurs. The topic is of great interest for both basic ecology and forestry, since self-thinning is a progressive decline in density in a population of growing plants. The problem originated when the works of Reineke (1933) and Yoda et al. (1963) led to the establishment of the so-called self-thinning rule, which proposes that a linear self-thinning trajectory (logarithms of average biomass and plant density) with slope of -1.5 is of general validity. Several empirical findings and theoretical explanations in support of or against the rule followed without consensus (Begon et al. 1996b; Reynolds and Ford 2005; Sackville Hamilton et al. 1995; Shaw 2006; Westoby 1984).

In their theoretical framework on the biomass trajectory of forests, Berger and Hildenbrandt (2003) assume that all different empirical findings regarding self-thinning are significant. The framework proposed by these authors is partially based on empirical evidence from mangrove forests, and is expected to account for empirical findings on the emergence of the different segments of a self-thinning trajectory, on the nature (i.e., linear or not) of the relationship between the average biomass and plant density in each segment of the trajectory, and on the slope of that relationship in the linear segment. The framework defines the segments in the biomass trajectory by looking (on the trajectory) at reference points where the skewness of the stem diameter distribution of the forest changes (Figure 5.1).

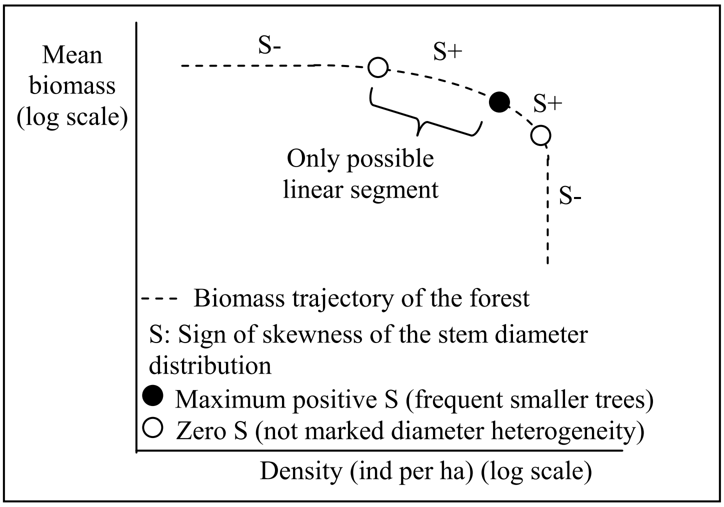


Figure 5.1 Theoretical framework of Berger and Hildenbrandt (2003)

The theoretical framework of Berger and Hildenbrandt (2003) (Figure 5.1) proposes that the biomass trajectory of the trees of a cohort in a forest can have a linear segment only after the skewness of the stem diameter distribution has had a maximum positive value. The framework proposes that the linear segment would end once the trajectory reaches its second zero in the skewness of the stem diameter distribution. The framework also explains why the value of the slope of that linear segment is not always -1.5 (Berger and Hildenbrandt 2003).

The empirical data obtained in the present study were graphed as described by the theoretical framework proposed by Berger and Hildenbrandt (2003) (i.e., mean biomass of a tree versus density of trees; log scale). By graphing the data, it was tested whether the biomass trajectory corresponding to these data fulfills the expectations of the theoretical framework, i.e., the biomass trajectory can have a linear segment only after the skewness of the stem diameter distribution has a maximum positive value (Figure 5.1). The empirical data graphed came from 10 surveys. Two of the 10 surveys correspond to plot I2 in the 1980s and 2005. The other 8 surveys were conducted in 2005 in 8 plots of sector II. These 8 plots are those with mangroves similar to the mangroves of plot I2, i.e., *A. germinans monospecific* (included in profiles F and G Figure 4.5). For the graph of mean biomass of a tree versus density of trees, the total biomass of the trees of each plot was calculated from the total basal area of the corresponding plot. The equation used in the calculation corresponds to the trendline of the data reported by Fromard et al. (1998), who studied mangroves ranging from pioneer to mature and declining mangroves. The equation is biomass = 13.179*basal area-136.39 (coefficient of determination R^2 = 0.90; N=8).

When testing the theoretical framework proposed by Berger and Hildenbrandt (2003) it is important to know the skewness of the stem diameter distribution (Figure 5.1). Therefore, for each of the 10 surveys, the skewness of the stem diameter distribution was explored by examining the position of the mean diameter with respect to the minimum and maximum (extreme values) diameters and by examining the relative size of the left and right tail of the normal curve in the histograms of stem diameter distribution (Moore and McCabe 1998).

5.3 Results

5.3.1 Differences between mangroves west and east of the dammed areas

Mangrove vegetation was highly developed in both the western and the eastern mangroves, and mangrove cover ranged from 70 to 95%. Non-mangrove species, when present, accounted for less than 10% of the total vegetation cover (Figure 5.2).

The western mangroves (sector I) had a lower tree density (for both alive and dead trees) and larger diameters than the eastern mangroves (sector VI).

In the western mangroves (sector I), from the seaward to the landward side, the mangrove zones were: a *R. mangle*-dominated belt fringing the sea (plot I1), an *A. germinans*-dominated zone (plot I2), and a *L. racemosa*-dominated zone in the most landward plot I3 (Figure 5.2).

In the eastern mangroves (sector VI), in contrast to the western mangroves (sector I), the seaward belt was not dominated by *R. mangle* but by *A. germinans* (plot VI1). The more inland plot VI2 was a mixed zone with all mangrove species (*R. mangle*, *A. germinans* and *L. racemosa*) and the pseudomangrove *C. erectus*. The most inland plot VI3 was dominated by *R. mangle* and *C. erectus*. Although not surveyed or represented in the transect in sector VI, in some sites of the coastline of this sector natural accumulations of sand (up to 10 m wide) between the mangrove belt and the sea exist. In these sand accumulations, species typical of sandy coast vegetation like *Sesuvium portulacastrum* (L.) L., *Canavalia maritima* (Aubl.) Thouars, *Batis maritima* L. and *Ipomoea pes-caprae* (L.) R. Br. mix up with mangrove plants. Such sand formations were not observed west of the dammed areas (sector I).

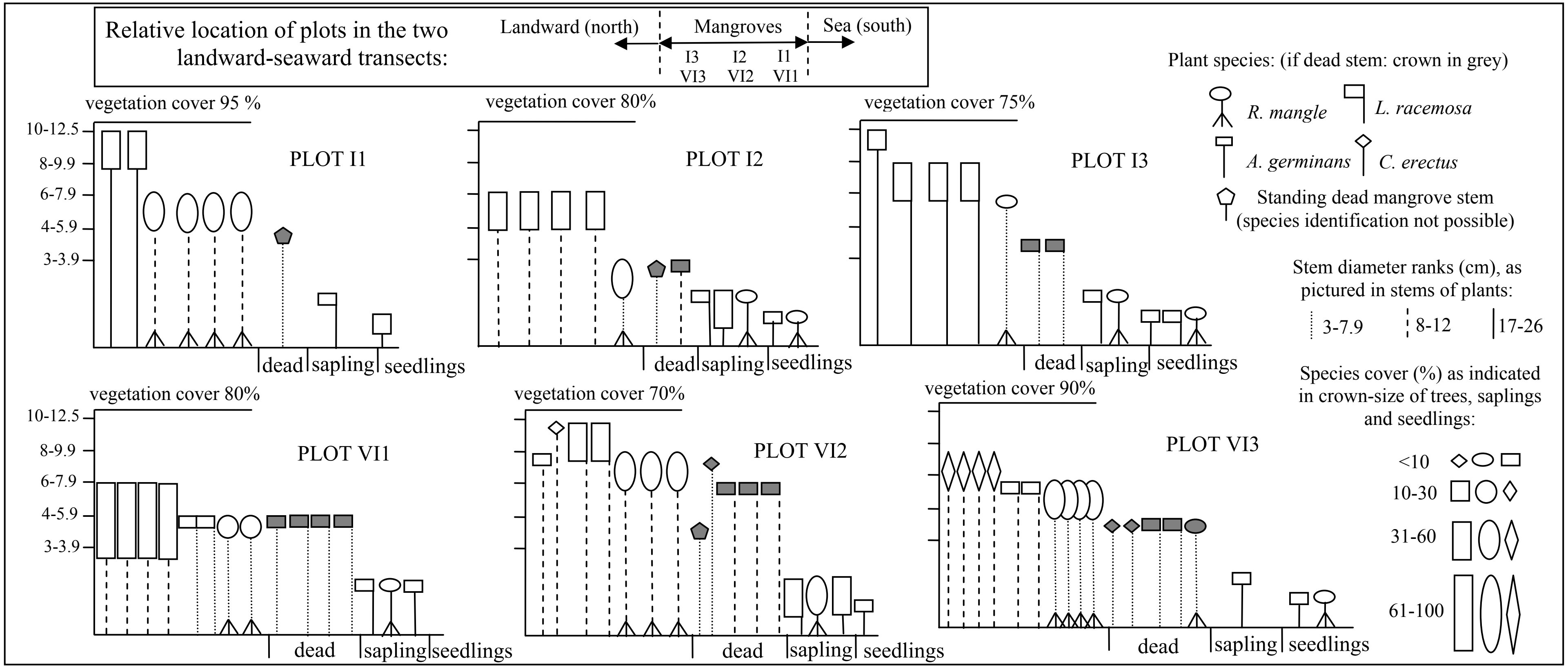


Figure 5.2 Vegetation profiles of two 3-plot transects outside dammed areas in sector I (western mangroves) and sector VI (eastern mangroves). Plant height (m, y-axis). Tree density of species (trees per 100 m^2 in 4 ranges: 1-3, 5-10, 14-18, 23-26; each range is represented by 1, 2, 3 or 4 plants of the species) Each profile represents a field-plots (Appendices 6 and 7)

In the mangroves both west and east of the dammed areas (plots I3 and VI2), *L. racemosa* had aerial roots (Table 5.1) where the trees were most highly developed (highest values for cover, diameter and height) (Table 5.1 and Figure 5.2). In contrast, *A. germinans* had aerial roots not only where the trees were most highly developed (i.e. plots I1, I2 and VI1), but also where trees of the species were merely present (i.e., plots I3 and VI2).

Table 5.1 Aerial roots (percentage cover) per species

Species	Plots sector I			Plots sector VI		
	I1	I2	I3	VI1	VI2	VI3
L. racemosa	0	0	30	0	25	0
A. germinans	75	0	39	100	25	0.5

The water level differed in the transects, i.e., in the western mangroves (sector I) the water was above the soil surface in both the dry and the rainy season, while the plots in the eastern mangroves were flooded only during the rainy season (Figure 5.3). In both transects, the water level was lower in the dry season than in the rainy season, except for plot I1, the plot closest to the sea in the western mangroves. Following the direction of seasonal change in water level, soil salinity tended to be higher in the dry season than in the rainy season (Figure 5.3).

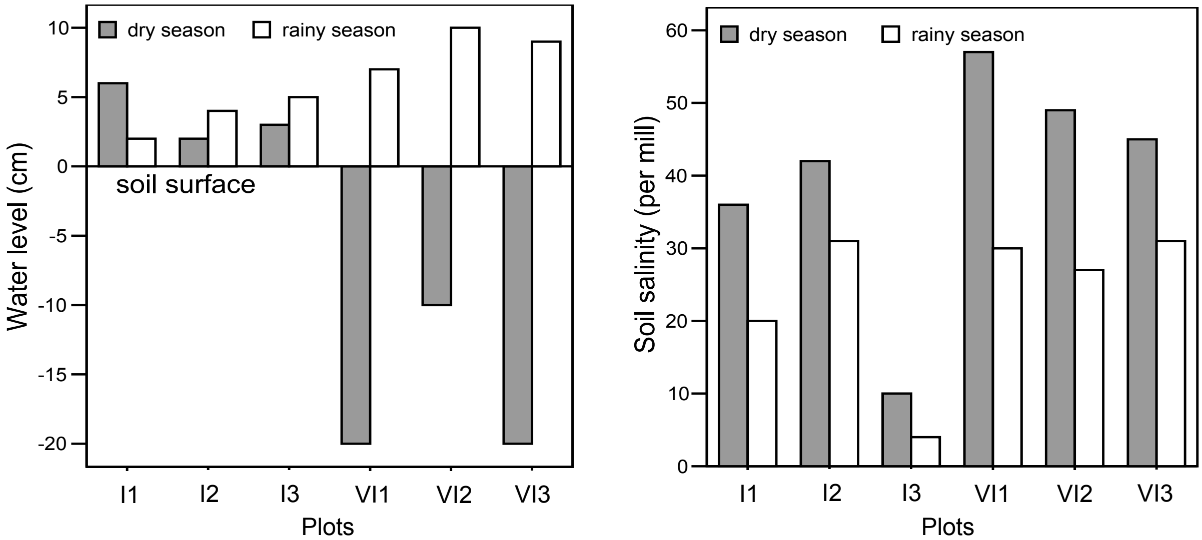


Figure 5.3 Water level and soil salinity in two 3-plot transects located west (sector I) and east (sector VI) of the dammed areas

The soil redox potential had more negative values during the dry season (Figure 5.4).

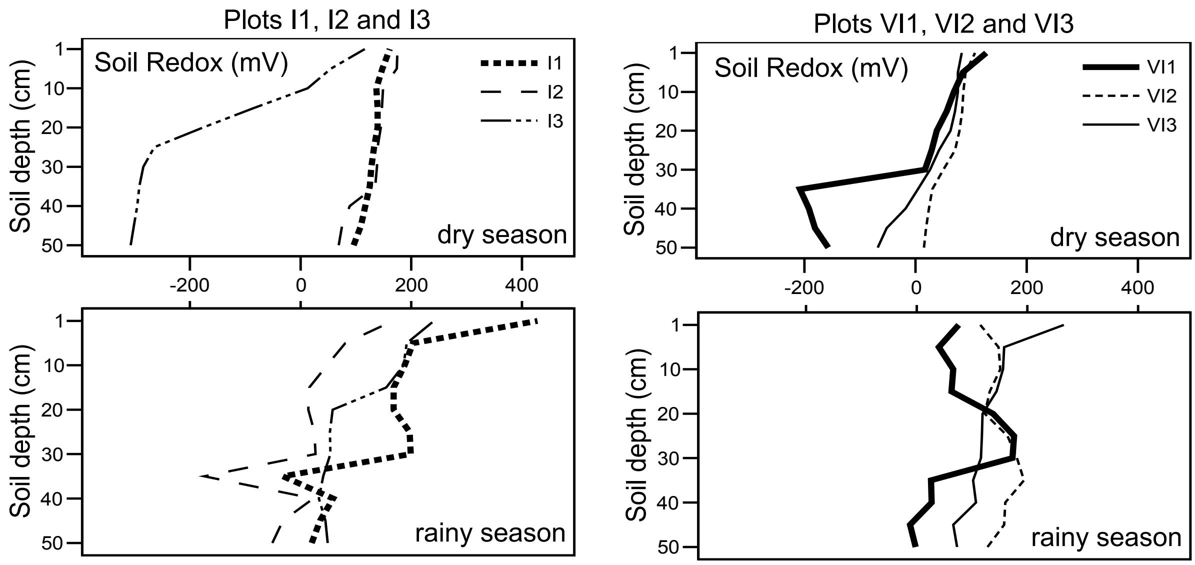


Figure 5.4 Soil redox potential in two 3-plot transects located west (sector I) and east (sector VI) of the dammed areas

5.3.2 Change in mangrove vegetation between the 1980s and 2005 west of the dammed area

In comparison with the survey conducted in the 1980s (Menéndez, 2000), the tree height surveyed in 2005 west of the dammed area (plots I1, I2 and I3; Figure 5.2) showed similar or higher values (e.g., *A. germinans* in plot I1, *L. racemosa*, respectively, in plot I3) (Table 5.2a). In 2005, tree densities tended to be lower than those in the 1980s (Table 5.2b), while the basal area tended to be bigger (Table 5.2c). The exceptions to this general pattern were species which represent a small part of the trees and the basal area of their respective plots (*R. mangle* in plot I2; and the same species and *A. germinans* in plot I3) (Table 5.2b and c).

Table 5.2 Mangrove vegetation in the 1980s (Menéndez 2000) and 2005: trees

a. Tree height (m). For the 1980s, range and/or maximum values (as reported by Menéndez 2000), and for 2005 mean and range are presented

Plot	Year	*R. mangle*	*A. germinans*	*L. racemosa*
I1	1980s	4-8(10)	5-10	0
	2005	7.6 (4-10)	12.3 (10-14)	0
I2	1980s	short thin trees	5-10(12)	0
	2005	3.5 (3.5-3.5)	7.5 (3-12)	0
I3	1980s	0	till 10 m	till 10 m
	2005	6.7 (6-8)	12 (12-12)	9.4 (5-14)

b. Tree density (trees per 100 m^2)

Plot	Year	*R. mangle*	*A. germinans*	*L. racemosa*	Total
I1	1980s	30.9	10.6	0	41.5
	2005	26	6	0	32
I2	1980s	0.2	29.1	0	29.3
	2005	1	24	0	25
I3	1980s	0	7.2	29.8	37
	2005	3	1	16	20

c. Tree basal area (m^2 per ha)

Plot	Year	*R. mangle*	*A. germinans*	*L. racemosa*	Total
I1	1980s	5.9	17.3	0	23.2
	2005	16.9	33.1	0	50
I2	1980s	0.2	20.8	0	20.8
	2005	0.1	37.7	0	37.8
I3	1980s	0	7.3	10.8	18.1
	2005	0.7	4	49.6	54.3

Tree diameters increased from the 1980s to 2005, except for *A. germinans* in plot I2. In 2005, the diameters of dead trees corresponded to the lowest range of live trees (Table 5.3).

Table 5.3 Diameter mangrove trees (alive, dead and stumps) in sector I for 2005 and the 1980s (Menéndez 2000)

Plot	Year		*R. mangle*	*A. germinans*	*L. racemosa*	Species not identifiable in the field (i.e. some of the dead stems and stumps)
I1	1980s	alive	3-13	3-27	0	not reported
	2005	alive	3.5-16.7	15.5-30.8	0	0
		dead	0	0	0	3.8-5
		stump	0	22.6	0	9.9
I2	1980s	alive	<5	3-32.9	0	not reported
	2005	alive	3.5-3.5	3.1-25.4	0	0
		dead	0	4.6-11.6	0	5.1
		stump	0	0	0	0
I3	1980s	alive	0	3-9	3-13	not reported
	2005	alive	4.9-5.7	22.6	5.1-45.1	0
		dead	0	0	4.4-6.3	0
		stump	0	11.9	0	0

Most of the trees surveyed in 2005 in the plots I1 and I3 had been already tagged in the 1980s (Table 5.4).

Table 5.4 Total number of trees in 2005. Of that total, number of trees with tag of the 1980s Menéndez (2000)

Plots	Total number of trees in 2005 (live trees + dead stems + stumps)	Number of trees with tag of the 1980s (live trees + dead stems + stumps)
I1	37	33 = 29 + 3 +1
I2	29	9 = 6 + 3 + 0
I3	26	22 = 16 + 5 + 1

Data of 2005 addressing the entire 1000 m^2 area of the 1980s plots (Table 5.2 and Table 5.3) indicate a greater change in vegetation structure than the 2005 data spatially restricted to the parts of the 1980s plots covered by the smaller 100 m^2 plots of 2005 (Table 5.4). For plot I1, 2005 data indicate a change from 33 live trees in the 1980s (Table 5.4) to 32 live trees in 2005 (Table 5.2b). It is a smaller change than taking the 41 live trees reported for the entire corresponding plot of 1980s as reference (Table 5.2b). In plot I3 the situation was similar, with a change from 22 live trees in the 1980s (Table 5.4) to 20 in 2005 (Table 5.2b), a smaller change than taking the 37 alive trees present in the entire plot of 1980s as reference (Table 5.2b).

In plot I2, the 2005 survey detected only 9 trees tagged in the 1980s (Table 5.4). Besides probable less effectiveness of the devices used for tagging the trees in plot I2, that low number of tagged tress found in 2005 could be due to the fact that, when placing the plot for the 2005 survey, a gap (apparently formed by lightning) had to be avoided. Avoiding that gap could have meant that the 2005 plot was not included in the 1980s plot but partially overlapped it. This could also be the reason why only in plot I2 did the diameter of *A. germinans* not increased as it was the case with the other species in plots I1 and I3 (Table 5.3). It is also possible that *A. germinans* with large diameter(s) in plot I2 fell over and decomposed between the 1980s and 2005. Plot I2 of 2005 can be considered representative of the corresponding 1980s plot, i.e., 29 live trees per 100 m^2 in the 1980s (Table 5.2b) and 25 alive trees (Table 5.2b) plus 3 dead tagged stems in 2005 (Table 5.4). Also, the mangrove vegetation of the zone represented by plot I2 is less heterogeneous than that in plots I1 and I3, as it is almost monospecific (*A. germinans*) (Table 5.2). Furthermore, plot I2 showed greater similarities between the data obtained in 2005 and the data from the 1980s (Table 5.2, Table 5.3).

Plot I3 showed the greatest differences in tree density and basal area between the 1980s and 2005 (Table 5.2). The species *L. racemosa* became more dominating in this plot, where the lowest salinity value of all plots was measured (Figure 5.3) under the probably increased influence of an inland freshwater lagoon located near this plot. This plot is also the only one where a new mangrove species, *R. mangle*, was reported in 2005. This species is still limited to 3 trees with a basal area of only 0.7 m^2 per ha (Table 5.2). Also, this is the only plot where one species, *A. germinans*, decreased in tree density without having increased in basal area (Figure 5.5).

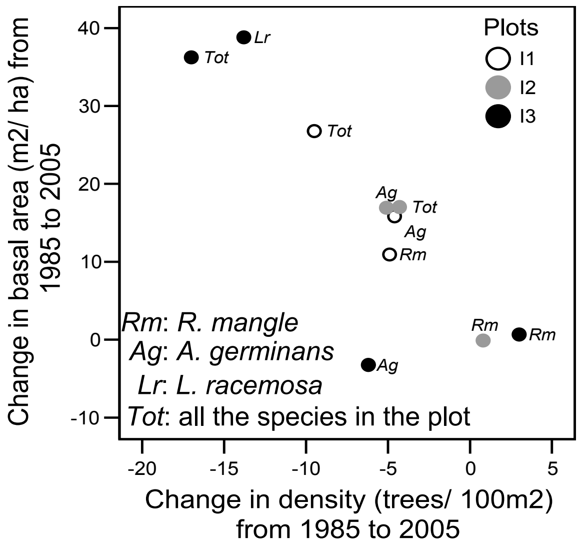


Figure 5.5 Change in mangrove vegetation west of the dammed areas. Each point represents the change in basal area versus change in tree density from the 1980s (Menéndez 2000) to 2005

5.3.3 Theoretical framework on self-thinning

The data of plot I2 (Figure 5.2), complemented with data surveyed in monospecific (*A. germinans*) plots of sector II (mainly from profile H, Figure 4.5), allowed testing one of the main aspects of the theoretical framework proposed by Berger and Hildenbrandt (2003) regarding the biomass trajectories of forests. This aspect relates to the existence of a single linear segment in the biomass trajectory. From the data, a linear segment can be distinsguished; and the slope of that segment is different from -1.5 (Figure 5.6).

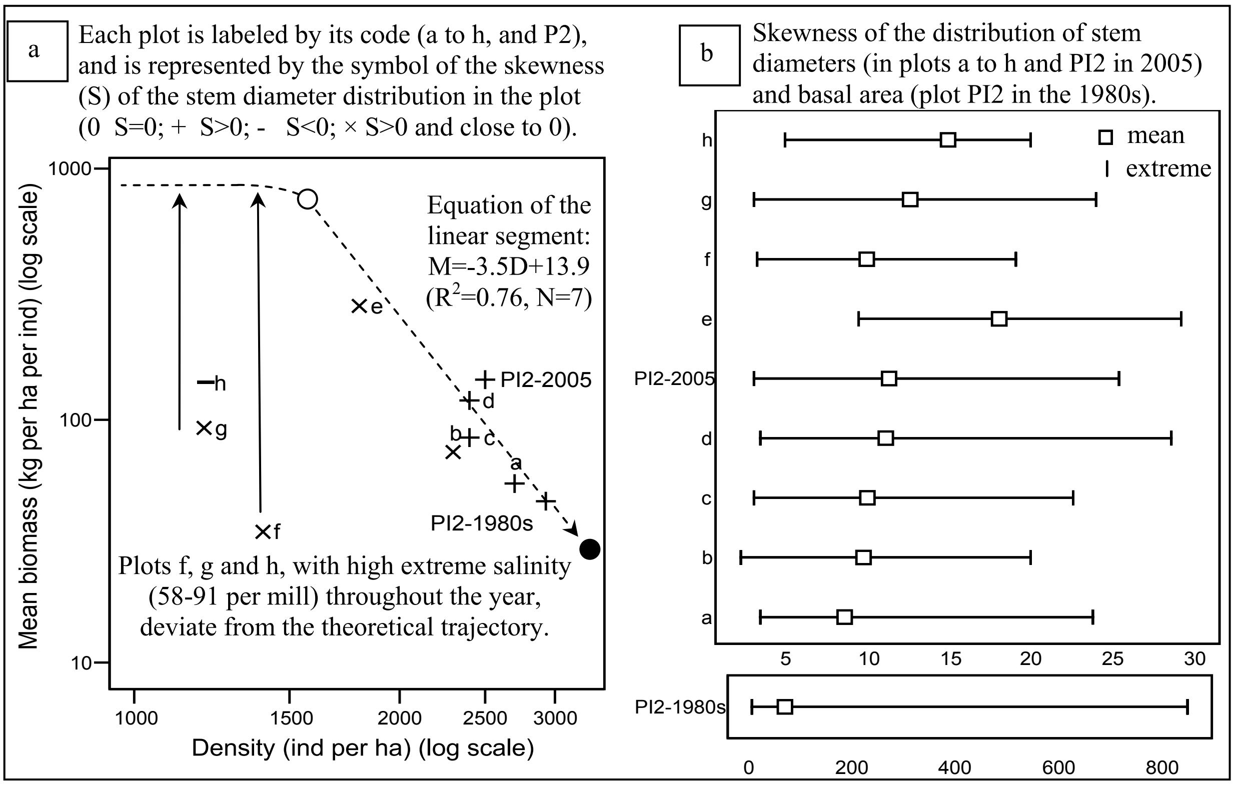


Figure 5.6 Testing the linearity of the biomass trajectory of self-thinning proposed by the theoretical framework of Berger and Hildenbrandt (2003). a: Arrangement of the data obtained in the study area on the biomass trajectory expected by the theoretical framework of Berger and Hildenbrandt (2003), with a focus on the only possible linear segment proposed by the framework. The labels PI2-2005 and PI2-1980s refer to data from plot I2 of sector I, respectively from 2005 and the 1980s (data from the 1980s calculated from Menéndez 2000). Plots a to h were surveyed in sector II in 2005. b: Skewness of the stem diameter distribution in the plots, as evidenced by the position of the mean diameter with respect to the minimum and maximum (extreme values) diameter in the plot.

The linear segment of the emprirical biomass trajectory was obtained between a plot with the highest positive skewness in the stem diameter distribution (P2 in the 1980s) and a plot with skewness close to zero (plot e) (Figure 5.6a, Appendix 8). In the empirical biomass trajectory, this plot e with skewness close to zero is followed by two other plots with skewness close to zero (plots f and g) and a plot with negative skewness (plot h) (Figure 5.6a). Three out of ten mangrove plots have much lower mean tree biomass than expected from the framework (plots f, g and h). These are the only plots where salinity is extreme throughout the year (Appendix 9).

5.4 Discussion

5.4.1 Mangroves west and east of the dammed areas

In the two areas located outside the wetlands dammed by the freshwater barrier, as a result of different exposure to major natural impacts (mainly tropical cyclones), the mangroves located west of the dammed areas were structurally better developed than those located in the east. Due to the less frequent strong tropical cyclones in 1902-2005, the trees in the western mangroves have a larger diameter and tree density is lower than in the eastern mangroves (Baldwin et al. 1995; Ball 1980; Berger et al. 2006; Doyle et al. 2003; Jiménez et al. 1985; Lugo 1980; Smith et al. 1994).

The development of the mangroves in the west is also favored by local marine currents. These currents, which flow to the west, carry sediments that stabilize the larger trees in these mangroves, which are located in a sheltered part of the coast. Load of sediments into mangrove areas enhances mangrove development as long as extremes do not occur, e.g., when sediments suddenly and permanently bury mangrove aerial roots and cause mangrove dieback (Blasco et al. 1996; Blázquez et al. 1988; Fromard et al. 1998; Snedaker 1995; Woodroffe 1990).

Less extensive anthropogenic impacts after the 1960s in the mangroves west of the dammed area have also favorably influenced the development of the western mangroves, e.g., occasional but uncontrolled mangrove logging and sand extraction have negatively affected the eastern mangroves more than the western mangroves. The reasons include more regulated access to the western mangroves thanks to the active role of a local coast guard station, a beach in the east attracting more people and leading to unregulated access to the eastern mangroves, and an institutional weakening of the local organizations involved in forestry in these mangroves (Hernández 2006; IGT 1999).

For western mangroves, comparison of data from 2005 and the 1980s shows the development of mangrove vegetation towards phases of greater maturity, i.e., increased tree diameters (and thus increased basal area) concomitant with a decrease in tree density, and higher tree height (Ball 1980; Berger et al. 2006; Fromard et al. 1998; Jiménez et al. 1985; Lugo 1997).

For the western mangroves, analyzing the degree of change in vegetation structure between the 1980s and 2005 faced challenges which are to be expected when comparing data surveyed by different studies and researchers (Snedaker and Snedaker 1984). For instance, evidence of less or greater change in vegetation structure depending on which 2005 data are taken (i.e., 2005 data addressing the entire 1000 m^2 plots of the 1980s, or 2005 data spatially restricted to the parts of the 1980s plots surveyed by the smaller 100 m^2 plots of 2005) emphasizes the need for caution when dealing with the horizontal variability of the apparently homogenous mangrove vegetation (Ball 1980; Berger et al. 2006; Dahdouh-Guebas et al. 2004; Fromard et al. 1998; Lugo 1997). Achieving meaningful multitemporal vegetation surveys and data comparisons also faces practical challenges. For instance, the field plots must be easily found in future surveys (GPS georeferencing can help), the devices used for tagging plots and trees must last until later surveys, and the surveyed data must be properly carried into future (i.e., data sharing between researchers and research organizations is fundamental).

Despite all care taken to ensure a meaningful comparison of vegetation data collected in the two time steps (1980s and 2005), the interpretation of the data can become more difficult or have no relevance if vegetation has changed its development trend (successional trajectory) due to disturbances (e.g., gap due to lightning) or less abrupt causes. For instance, in the most landward part of the western mangroves (plot I3; Figure 5.2), lowered soil salinity probably due to the increased influence of an inland freshwater lagoon has enhanced the previous path change of vegetation, i.e., increase in the dominance of the species *L. racemosa*. The increased influence of the inland freshwater lagoon could have lead to the major differences (e.g., in species dominance) to that part of the western mangroves. This is the only part of the western mangroves where a new mangrove species (*R. mangle*) was reported in 2005. The development of *R. mangle* can improve when salinity decreases, and the dispersion of *R. mangle* seedlings can improve with more frequent flooding, a probable consequence of the increased lagoon influence (Medina et al. 2001; Tomlinson 1986; Wolanski et al. 1992).

The structurally better developed vegetation of the mangroves located west of the dammed areas than mangroves located eastern to the dammed areas is in contrast to the flooding pattern, i.e., the more permanent flooding conditions (water level above the soil surface in both the dry and the rainy season) can lead to the incorrect interpretation that eastern mangroves, seasonally flooded, are less damaged by the long-term sea level rise (Blasco et al. 1996; Cahoon et al. 2006; Hernández 2006; IGT 1999; Medina et al. 2001; Menéndez 2000; Snedaker 1995; Wolanski et al. 1992; Woodroffe 1990).

However, the seasonal flooding of the eastern mangroves merely indicates that the former coastline mangroves in this area have been eroded. Although the *A. germinans*-dominated coastline of the eastern mangroves could be natural, a *R. mangle*-dominated coastline, as found both in the western mangroves and the dammed areas, should also be the typical natural status in the eastern mangroves. Here, the erosion of a former *R. mangle*-dominated coastline is indicated by the retreat of the coastline from 1985 to 2001 (remote sensing-based evidence; section 3.3.2), fallen *A. germinans* trees on the coastline, and the presence of *C. erectus* in the mangrove zone fringing the sea. The species *C. erectus* naturally occurs in zones less directly exposed to the sea, i.e., commonly towards the landward border of mangroves and always where flooding is exceptionally infrequent (Blasco et al. 1996; Fromard et al. 1998; Menéndez 2000; Snedaker 1995; Tomlinson 1986; Woodroffe 1990). Although anthropogenic impacts on eastern mangroves have enhanced coastal erosion, the extended landward presence of *R. mangle* in these mangroves might also indicate that change in the position of the coastline is a natural local process (Blasco et al. 1996; Gilman et al. 2006; Hernández 2006; Snedaker 1995; UNEP 1994; Wells et al. 2006; Woodroffe 1990).

Despite the active coastal erosion, mangroves can be expected to remain as the characteristic land cover in the coastline east of the dammed areas. It could mean that the ongoing changes associated to erosion will only change the relative density of the mangrove species in these mangroves, where all the local native mangrove species (*R. mangle*, *A. germinans* and *L. racemosa*) and the pseudomangrove *C. erectus* are present. The species *R. mangle* is expected to be the future dominant species, since it can cope with the two main expected sources of mangrove degradation due to coastal erosion, i.e., increased mechanical influence on the trees, and accentuated soil anaerobiosis (Ball 1980; Blasco et al. 1996; Snedaker 1995; UNEP 1994; Woodroffe 1990). The species can cope with this kind of degradation through the structural stability and aerial respiratory structures (lenticels) provided by prop roots. Aerial roots, of the type prop root, are always present in *R. mangle* plants. In contrast, the aerial roots of *A. germinans* and *L. racemosa*, of the type pneumatophore, are not always present, i.e., are more facultative (Kraus et al. 2006; McKee 1993, 1995; McKee and Faulkner 2000; McKee and Mendelssohn 1987; McKee et al. 1988; Thibodeau and Nickerson 1986; Tomlinson 1986).

Coastal erosion causes the sea to directly reach landward mangrove zones, and thus mangrove species that would otherwise remain farther from the sea, and which are naturally less adapted to cope with the direct influence of the sea. The species *C. erectus* cannot develop aerial roots at all, so in the mangroves east of the dammed areas, coastal erosion would affect *C. erectus* via both accentuated anaerobic conditions and increased mechanical influence (Cintrón and Schaeffer-Novelli 1983; Tomlinson 1986). The species *A. germinans* and *L. racemosa* would be damaged mainly through lessening structural stability, i.e., the aerial roots, when present, allow coping with accentuated anaerobic conditions but much less with increased mechanical influence (Boto 1984; McKee 1993; McKee and Faulkner 2000; Tomlinson 1986).

Before a *R. mangle*-dominated coastline can develop in the eastern mangroves, further coastline retreatment is to be expected, i.e., degradation of the existing *A. germinans*-dominated coastline through damage to the structural stability of *A. germinans* trees. However, the degradation of the existing coastline can give the time needed for existing *R. mangle* trees to develop further and form a *R. mangle*-dominated coastline. Environmental management can take advantage of that time, too, and manage coastal erosion (Blasco et al. 1996; Krauss et al. 2003; Ning et al. 2003; Snedaker 1995; Woodroffe 1990).

For the degradation of the existing *A. germinans*-dominated coastline to happen, coastal erosion will have to change the existing conditions that are suitable for the dominance of *A. germinans* over the other species *R. mangle*, *L. racemosa* and *C. erectus*. The existing vegetation cover and tree density of *A. germinans* is higher, tree diameter are larger, aerial root cover is 100%, and there are no dead individuals. Favorable conditions for *A. germinans* include a combination of high salinity (respectively 57 and 30 per mill in the dry and the rainy season, respectively), and highly reduced soil but without permanent flooding (water level below and above the soil surface, in the dry and the rainy season, respectively, and marked negative soil redox potential < -150 mV deeper than 30 cm).

Damage to *L. racemosa* close to the coastline through both increased mechanical influence on trees and accentuated soil anaerobiosis, should occur more rapidly than damage to *A. germinans*. The former species already accounts for most of the dead trees in the eastern mangroves, especially in the zones located closest to the coastline, where this species lacks aerial roots. If *L. racemosa* starts developing aerial roots and thus becomes more able to cope with accentuated soil anaerobiosis, the expected increase in mechanical influence of the sea will remain a source of damage for this species (Kraus et al. 2006; McKee 1993; McKee and Mendelssohn 1987; McKee et al. 1988; Thibodeau and Nickerson 1986; Tomlinson 1986).

In spite of coastal erosion in the eastern mangroves, sand accumulation around with species typical of sandy coast vegetation shows a natural potential towards the development or re-constitution of a catena formed by sandy coast vegetation and mangroves (Blasco et al. 1996; Snedaker 1995; Woodroffe 1990). That natural potential can be used by management and restoration to combat coastal erosion. However, coastal infrastructure located to the east of the study area makes the formation of the catena difficult, because such infrastructure blocks sediments that would otherwise be transported by local marine currents (Hernández 2006).

5.4.2 Theoretical framework on self-thinning

The empirical biomass-density trajectory of self-thinning obtained in this study can be explained by the theoretical framework proposed by Berger and Hildenbrandt (2003). As expected from that framework, the linear segment of the biomass trajectory was observed between a plot having the highest positive skewness in the stem diameter distribution, and a plot having skewness close to zero. Also as expected from the theoretical framework, the slope of the empirical linear segment obtained in the present study differs from -1.5, i.e., it equals -3.5.

In the present study, empirical deviations from the theoretical framework occurred, i.e., 3 out of 10 mangrove plots had much lower mean tree biomass than expected from the framework (plots f, g and h; Figure 5.6). The biomass values in these mangrove plots would be higher, and thus the plots would better fit the expectations of the theoretical framework, if these plots were not the only plots where extreme salinity (58-91 per mill) throughout the year is preventing mangroves from achieving higher biomass (Cintrón and Schaeffer-Novelli 1983).

The empirical deviation from the theoretical framework due to extremely high salinity values exemplifies the spatial heterogeneity in the abiotic factors of a mangrove forest, i.e., heterogeneity detected when comparing mangrove plots surveyed at the same time (Cintrón and Schaeffer-Novelli 1983; Tomlinson 1986). If spatial heterogeneity in abiotic factors causes deviations from the theoretical trajectories of biomass, then temporal heterogeneity would also cause deviations. Temporal heterogeneity can be detected, for instance, by surveying the same forest over time (monitoring). Data from such multitemporal surveys could show even greater deviation from the theoretical trajectories proposed by Berger and Hildenbrandt (2003), since multitemporal surveys usually show less good control of changes in abiotic factors than when all plots are surveyed at the same time.

The observed empirical deviations suggest that the theoretical framework proposed by Berger and Hildenbrandt (2003) could be expected to encompass all empirical trajectories, but also that the specific theoretical trajectories themselves are difficult to observe in nature, for instance due to deviations caused by heterogeneity in abiotic factors (e.g., salinity). However, beyond its theoretical importance, the practical relevance of the framework remains, since deviations from the theoretical trajectories can help identifying thresholds in abiotic conditions (e.g., in salinity) relevant for mean biomass and thus for forest functioning and forest services.

The empirical deviations from the theoretical framework manifested in the mangrove plots with high salinity throughout the year also suggest that explaining empirical biomass trajectories may need to incorporate ecological factors different from those on which the theoretical framework is built. The framework is mainly built on the strength of competition between the trees. The deviating plots in this study (plots f, g and h; Figure 5.6) might not deviate from the trajectory along which the other plots seem to fit better, but might also correspond to an alternative biomass trajectory. The alternative trajectory for the deviating plots would go through lower values of mean biomass, as can be expected by the extreme salinity of these plots throughout the year. The existence of different trajectories is explicitely addressed by the theoretical framework of Berger and Hildenbrandt (2003). However, since the possibility of having more than one empirical trajectory was detected in this study in concomitance with extreme salinity values, then explaining the existence of different trajectories may not need to be restricted to differences in the strength of competition between plants as basically assumed in the theoretical framework, but also incorporate differences in other factors that affect the values of mean biomass (e.g., salinity). The need to incorporate factors different from the strength of competition among trees to explain biomass trajectories would not be a limitation of the theoretical framework. On the contrary, it would evidence that the framework can address empirical situations that are beyond the assumptions on which the framework has been built.

5.5 Conclusions

Mangroves located outside the dammed areas changed in structure and floristic composition, although the changes were not related to a single major event, as was the case for the mangroves in the dammed areas and the construction of the barrier.

East of the dammed areas, restoration of mangroves and management of coastal erosion must be considered, since the coastline mangroves are retreating, aggravated by a combination of natural and anthropogenic events. The natural events included the influence of more frequent and stronger tropical cyclones than in the mangroves east of the dammed areas, and location in a less sheltered part of the coast, which is also less favored regarding the local pattern of coastal sediment movement. The anthropogenic events relate to sand extraction on the coastline, logging, existing coastal infrastructure and less efficient mangrove protection.

The research station established in the mangroves west of the dammed areas in the 1980s, and relocated during the present study, should be reactivated. The station can provide results relevant both at the local and the non-local level. In this study, data was used for testing a recent (2003) theoretical framework for a long-time unsolved problem in ecology (since 1933), i.e., the biomass-density trajectories of self-thinning. Given the mixture of spatial and temporal aspects of vegetation addressed by that framework, both the framework and the local research station are suitable for helping to solve the long-lasting discussion in mangrove ecology on what the zonation of mangrove vegetation means in the context of mangrove succession.

6 A MODELING FRAMEWORK FOR ASSESSING AND MANAGING RESILIENCE

6.1 Introduction

For a given territory, ecosystem or environmental system in general, a regime shift refers to the change between two fundamentally different statuses (named regimes) (Kinzig et al. 2006; Walker et al. 2006a). Regime shifts are commonly addressed regarding the change between well established efficient environmental management, and inefficient environmental management. For example, Kinzig et al. (2006) explore potential regime shifts and future outcomes for the Causse Méjan region of France, where the observed decline of its native grasslands might undesirably transform the local culture and gastronomy, which are characterized by the production of Roquefort and Fedou cheeses made from sheep milk.

A regime shift can manifest as a sudden collapse or can take longer, but it is always a drastic change in the properties of an environmental system. The regimes preceding or resulting from the shift have different feedbacks and internal controls regarding their ecological, social, economic, cultural and environmental components in general (Kinzig et al. 2006).

These changes or regime shifts are usually examples of surprises in environmental management and include cases in which the outcomes of management differ from its goals (Abel et al. 2006; Walker et al. 2006b), for instance when fire suppression seeking forest improvement leads to forest degradation. Since regime shifts involve marked changes, they have been interpreted as discontinuities and so as manifestations of non-linearity in environmental systems (Abel et al. 2006; Carpenter et al. 2001; Holling 1973, 1986; Walker et al. 2006a). Thus the methods assessing regime shifts are expected to deal with non-linearity.

A regime shift involves the notion that the regime preceding the shift, and/or the one resulting from it, can be persistent and tend to last, and so be resilient (Kinzig et al. 2006). Therefore, addressing regime shifts and assessing resilience are intimately related. At the same time a regime can be desired or not, as some specific forms of environmental management can be considered adequate or not (Carpenter et al. 2001; Kinzig et al. 2006).

So, in environmental management, promoting the desired status for a territory or environmental system can be interpreted as trying to promote the resilience (persistence) of desired regimes, and avoiding their shifts into non-desired ones. It gives resilience assessment an important place in environmental management. The definition of resilience in the present study is according to Holling (1973), i.e., "a measure of the persistence of systems and of their ability to absorb change and disturbance and still maintain the same relationships between populations or state variables". This definition shows that assessing resilience of an environmental system is basically an exploration of the relations in the system. The core of such an exploration is the discussion of hypotheses about the interaction of the system components and analysis of what makes the environmental system persist (Grimm and Wissel 1997; Holling 1973; Walker et al. 2006b).

The assessment of resilience has remained an unsolved challenge (Abel et al. 2006; Carpenter et al. 2001; Holling 1973, 2003; Kinzig et al. 2006; Walker et al. 2006a, b). Critical is the definition of a suitable methodological framework for articulating the diverse theoretical and empirical insights available in a particular resilience assessment.

Fundamental demands of such a framework include arranging or portraying the relations between the components (variables) considered relevant in the environmental system. However, these relations cannot always be immediately studied through exact equations or numerical methods (e.g., due to theoretical or practical constraints). It should be possible to address relations between components (variables) commonly different (e.g., natural, social), numerous, which might not be readily measured or whose available data is partial and not always quantitative. A framework for resilience assessment should also provide operational support for addressing changes in the values of the variables considered relevant, or at least in their trends.

These demands reflect the complexity of environmental systems, and suggest the use of qualitative mathematical modeling to help achieving a framework for resilience assessment. Similar demands have increased the relevance of qualitative modeling in other fields of environmental science and ecology (Bodini and Giavelli 1989; Levins 1974, 1998; MEA 2005b; Petschel-Held et al. 1999; Salles et al. 2006). The most influential literature in resilience research has also recognized that more quantification will not always increase understanding (Holling 1973, 1986, 2003; Walker et al. 2006a, b).

The objective of this chapter is to propose how to ascertain if a particular regime of an environmental system is resilient, and how management can use the assessment of resilience to promote the persistence or change of a regime. The methodological approach regards the feasibility of using qualitative mathematical models for representing a regime or situation under study. The empirical evidences come from three regimes observed in wetlands on the south coast of Havana province (Cuba), where a freshwater barrier was built in 1986. For these wetlands, previous chapters addressed the local changes in land cover (from 1985 to 2001) (Chapter 3) and wetland vegetation, mainly mangroves (chapters 4 and 5).

6.2 Methodology

6.2.1 Characterization of three regimes observed in a wetland environment

In the western part of the south coast of Havana province in Cuba, three regimes were observed (Figure 6.1).

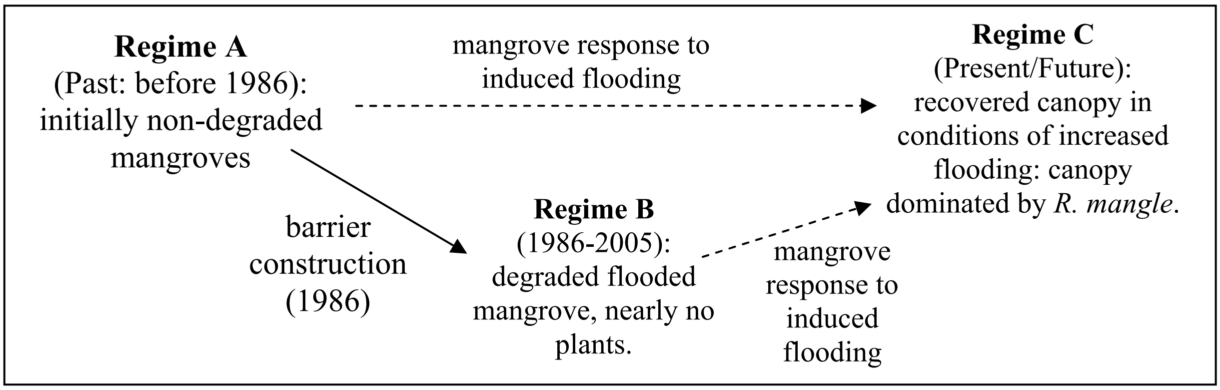


Figure 6.1 Three regimes in the western part of the south coast of Havana province with respect to coastal freshwater barrier constructed in 1986

Regime A (Initially non-degraded mangroves) refers to a mangrove forest without any major event leading to a permanent decrease in its cover. The construction of the barrier caused mangroves of Regime A to turn into Degraded flooded mangroves (Regime B). Regime A had persisted for years, as mangroves form the local natural vegetation (IGT 1999). Regime B had persisted since the barrier was built (1986-2005). In some sites, the barrier had not lead to a total dieback in the mangrove cover but to an increase in more flood-tolerant species (Regime C). The degraded flooded mangrove (Regime B) also shows tendencies towards Regime C.

6.2.2 Assessing resilience with loop analysis

In this chapter, the methodological approach for assessing the resilience of regimes of an environmental system will give priority to the practical relevance of such assessment in environmental management. The methodological approach builds on Holling's (1973) definition.

Although formal (mathematical) tools available in the literature will be used in the methodological approach, the practical relevance of resilience assessment makes the so-called intuitive notion of stability a suitable initial reference to start formalyzing a methodological handling of Holling's (1973) definition of resilience. The formal (mathematical) foundations of such intuitive notion of stability are presented by Svirezhev (2000) in the context of stability concepts in ecology. He states that the notion of stability can be intuitively clear, where stability is "an ability to persist in the course of a sufficiently long time in spite of perturbations". However, and as will also be seen in the following, Svirezhev (2000) recognizes that such intuitively clear notion of stability poses major formal (mathematical) challenges and, therefore, that notion has so far not been formally (mathematically) defined in a unique and unambiguous way.

Svirezhev (2000) states that even in mathematics, a science characterized by the avoidance of conceptual and methodological incongruities, the theory of stability uses about thirty different definitions of stability. Nonetheless, Svirezhev (2000) points out that some consensus exists about the Lyapunov stability, which appears to be inherent in or substantial for any further notion of stability. According to Svirezhev (2000), Lyapunov stability from an ecological point of view means an ability to persist in the course of a sufficiently long time in spite of perturbations.

The methodological approach in the present study builds on that seemingly universal mathematical treatment of stability, i.e., the Lyapunov stability. This is because the methodological approach will be based on loop analysis, a qualitative mathematical modeling method adopted by Levins (1974) and whose stability criteria build on Lyapunov stability.

Assessing resilience of regimes of an environmental system will be based on the feasibility of representing a resilient regime with a stable loop model. If the regime is resilient, the loop model representing the regime is stable. Correspondingly, if the regime is not resilient, the loop model representing the regime is not stable.

A loop model is the type of model to be developed when using loop analysis Levins (1974, 1998). The loop model representing a regime of an environmental system will work as analytical framework (platform) to assemble (represent) the components (variables) considered relevant in the regime, e.g., considered relevant in environmental management. The loop model will also allow proposing and exploring influences (relations) between the variables represented in the loop model. Adopting the methodological focus of loop models on variables and relations between variables implies that an environmental system is approached with a notion of system according to Levins (1998), i.e., a system is considered a network of partly opposing and partly reinforcing processes, observable as changes in their intersections at specified variables.

Developing and using a loop model does not start with a ready-made model provided by software. Therefore, a fundamental step is to select the variables considered relevant, and to define the interactions that represent how the variables influence each other (Levins 1974, 1998; Puccia and Levins 1985). This goes along with the explorative nature of resilience assessment (Abel et al. 2006; Carpenter et al. 2001; Holling 1973, 1986; Kinzig et al. 2006; Walker et al. 2006a, b).

Loop analysis takes the problem of how to address specific variables and interactions and still give relevance to other framework variables and interactions. For example, when studying the dynamics of plant establishment in a forest (e.g., seed germination): how to decide whether forestry policies that regulate access to forests and thus influence further development of germinated seeds (via paths of cattle and people) should also be addressed and then implemented? Therefore, the method was developed to help dealing with complex systems as wholes (Levins 1974, 1998), a common challenge in resilience assessment and environmental management.

To avoid confusion, it should be mentioned that loop analysis is followed in this study as defined by Levins (1974), and not as it had been previously developed or used to study electrical circuits and breeding systems (Puccia and Levins 1985). Although these different approaches share part of their mathematical foundations, Levins (1974) puts a stronger emphasis on qualitative analysis (Puccia and Levins 1985).

6.2.3 Basic aspects of loop analysis

Most of the applications of loop analysis have regarded dynamics of ecological communities and populations (Briand and McCauley 1978; Giavelli and Bodini 1990; Giavelli et al. 1988; Levins 1975; Lane 1986b), one of the main original fields of study of Levins, who developed the method. More environment-oriented applications have addressed the conditions for stability of management options in riverine wetlands (Bodini et al. 2000), and cumulative effects (Lane 1998).

Introductory or comprehensive overviews of the method are available (Levins 1974, 1998; Puccia and Levins 1985; Wright and Lane 1986), as well as less detailed explanations on some of its parts (Bodini 1998; Bodini and Giavelli 1989; Lane 1986*a*; Lane and Collins 1985; Lane and Levins 1977).

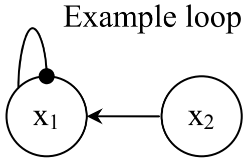
In this study, the following requirements or tools of loop analysis will be used: developing a loop model (also named signed graph), alternative loop models, stability of loop models, and response of variables to parameter change (Levins 1974, 1998; Puccia and Levins 1985).

Developing or proposing a loop model is a basic starting step in loop analysis, since all other tools will commonly use the loop model as a platform for analysis. While developing a loop model, not achieving the loop model itself but gaining insight into the relations in the studied environmental system will be commonly the most fruitful part of applying loop analysis. In this regard, developing a loop model will commonly lead to alternative loop models. This is closely related to the explorative nature of both loop analysis and resilience assessment.

In loop analysis, addressing the stability of a loop model means addressing how the variables, and the interactions between variables in that loop model, lead the variables to have persistent values or to tend towards persistent values. Thus, the variables considered for an environmental system represented by a stable loop model are interpreted as having persistent values or tending towards persistent values. The mathematical notion of stability in loop analysis will be addressed in Figure 6.3 and Appendix 10.

In what is relevant for the present chapter, the qualitative nature of loop analysis manifests explicitly as explained in this paragraph and the next one. When developing or proposing a loop model, the influence of one variable on another variable is characterized by the sign of that influence; the method does not demand a complete equation to represent how a variable influences another variable (Figure 6.2). The sign (positive or negative) of the link can be proposed after empirical evidences or more theoretical propositions about the behavior of the variables involved in the link (Lane 1998; Puccia and Levins 1985).

Example loop model with two variables x_1 and x_2 :



: Link of negative sign, also known as decreasing effect.

Link of positive sign, also known as enhancing effect.

If a link is represented in a loop model, it indicates that the following kind of influence is present: if the variable source of the influence (where the link starts) changes, then the influenced variable (where the link ends as pictured by the arrow head) responds by changing faster or slower. An absent link means that such an influence does not exist; for example variable x_1 does not influence variable x_2.

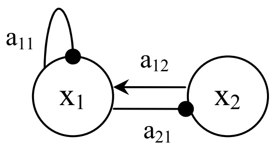
The sign of a link characterizes the similarity between the change in the variable source of the influence and the change in the speed of variation of the influenced variable. A positive link means both changes are qualitatively similar (either decrease or increase), while a negative sign means they are qualitatively different (one decreases and the other increases).

Figure 6.2 Loop analysis: links between variables in a 2-variable loop model

For the variables represented in a loop model, the tool analysis of parameter change indicates whether their values will increase, decrease, or remain unchanged in response to a change in the speed of variation of any of the variables (Puccia and Levins 1985).

A user of loop analysis can choose to ignore the mathematical basis of the method, but it is indispensable that the above described qualitative nature of the influences between variables and speed of variation of variables applies. Only when it applies, is developing a loop model and interpreting results in loop analysis then correctly relying on the mathematical background of the method: the equivalence of a loop model, a system of qualitative differential equations near a stable equilibrium point, and a square matrix (Figure 6.3).

Example: System under study represented by a loop model (also named signed graph) with two variables x_1 and x_2:



a_{ij}: link (influence) between variables.

Influence (link) of negative sign.

Influence (link) of positive sign.

A link (influence) in a loop model indicates that: if the variable source of the influence changes, then the influenced variable responds by changing faster or slower. In other words: its speed of variation (rate of change) is influenced. The loop model represents each influence (link) only qualitatively, with the sign of the influence. It corresponds to the sign of the following differential:

$$\frac{\partial}{\partial x_i}\frac{dx_i}{dt}$$

For each variable x_i in the loop model, $\frac{dx_i}{dt}$ represents the speed of variation (rate of change) of the variable, and depends on all the variables in the signed graph:

$$\frac{d\,x_i}{dt} = f_i\,(x_1, x_2, \ldots, x_n)$$

Each f_i represents near stable equilibrium situations:

$$f_i \rightarrow 0 \text{ for all } i$$

Square matrix: its elements are the links a_{ij} among variables in the signed graph:

$$A = \begin{vmatrix} a_{11} & a_{12} \\ a_{21} & a_{22} \end{vmatrix}$$

Matrix A for the example signed graph with variables x_1 and x_2:

$$A = \begin{vmatrix} - & + \\ - & 0 \end{vmatrix}$$

The matrix notation postulates that the relations between the variables are linear (originally or after linearization).

Loop analysis addresses stability by using the stability criteria for the zero solution of a system of differential equations.

Figure 6.3 Loop analysis: equivalence of a loop model, a matrix and a system of simultaneous qualitative differential equations near a stable equilibrium point (after Lane 1986a,b 1998; Lane and Collins 1985; Lane and Levins 1977; Levins 1974, 1998; Puccia and Levins 1985)

6.2.4 Loop models and resilience of the regimes in a wetland environment, and exploration of a future scenario

As previously mentioned, developing a loop model is a fundamental starting step in loop analysis. In this study, the loop models to be obtained for representing the three observed regimes (Figure 6.1) will include two variables: mangrove cover and infrastructure, because in the territory of interest, all processes can be interpreted as the intention of mangroves or infrastructure to persist (IGT 1999). Mangrove cover refers to the extension of mangrove vegetation as understood in vegetation science. Infrastructure refers to man-made structures, which usually replace or influence vegetation during land-cover and land-use change; those structures include roads, freshwater barriers, buildings.

Taking only two variables will make clearer how loop analysis is proposed to assess resilience. Increasing the number of variables does not impose any special limitation in loop analysis besides those commonly present in most modeling efforts (Puccia and Levins 1985). Lane and Collins (1985) had 7-14 variables in their loop models; Lane (1986*a*) used between 14 and 18 variables.

The Regimes A (Initially non-degraded mangrove) and B (Degraded flooded mangrove) (Figure 6.1) have lasted for years (section 6.2.1). Regime C (Recovered canopy) (Figure 6.1) is not yet spatially extended in the study area (see Chapter 3), but can be expected that the mangrove canopy will recover and persist in conditions of increased flooding (see Chapter 4).

Since the persistence and hence resilience of the studied regimes is not neglected by their long actual (Regimes A and B) or potential durations (Regime C), the notion of stability explained by Svirezhev (2000) and taken in this study applies (see 6.2.2). Therefore, the three regimes (A, B and C) will be represented with stable loop models.

For the Regime A, the links to be proposed between the variables mangrove cover and infrastructure will be found by stepwise reconstruction of loop models (Puccia and Levins 1985) using empirical evidence of how these variables responded to a direct influence on one of them, i.e., to infrastructure (Table 6.1).

Table 6.1 Evidence used in the stepwise reconstruction of loop models to find the 2-variable loop model(s) representing Regime A: decrease in mangrove cover and increase in infrastructure in response to a direct influence on infrastructure (construction of a freshwater barrier in 1986)

		Phase 1: 1970-1985	Phase 2: Impact introduction. Part of 1986.	Phase 3: 1986-2001
Variable: mangrove cover (m)	Value	High (Mangroves as natural vegetation)	High	Low (Mangrove cover decreased)
	Speed of variation	About 0 (variable m about constant)	Negative (variable m started decreasing)	Negative (variable m changed from high to low)
Variable: infrastructure (i)	Value	Low	Medium (Barrier construction)	Medium (Barrier already built)
	Speed of variation	About 0 (variable i about constant)	Positive (variable i started increasing)	About 0 (variable i about constant)

The stepwise reconstruction of loop models will indicate the loop models whose arrangement of variables (links or influences between variables) could have produced the observed tendencies of the variables. This is supported by the loop analysis tool of parameter change (Puccia and Levins 1985) (section 6.2.3).

For the regimes B (Degraded flooded mangrove) and C (Recovered canopy), finding the links between the variables mangrove cover and infrastructure for the loop models will not be based on observed responses of the variables, as was the case for regime A (Table 6.1). Proposing the links of the loop models for regimes B and C will reflect an earlier explorative phase than for Regime A, and will come from discussing less systematic evidence available about the variables. These evidences are provided by previous studies in the study area (IGT 1999; and previous chapters in this thesis).

While finding the stable loop models that could represent the three regimes, the pool of candidates for stable 2-variable loop models will be conveniently restricted to those diagnosed as stable without ambiguity (without needing to add more information on the links represented in the loop models) (Levins 1998; Puccia and Levins 1985). This will avoid non-essential arithmetic, which would unnecessarily complicate the explanation of how to apply loop analysis in resilience assessment.

However, that ambiguity is sometimes essential, and will be taken up again when addressing a future scenario.

In a future scenario, the potential outcomes of influencing specific components in a given territory or environmental system are investigated, a common task in environmental management. The future scenario will explore, for Regime B (Degraded flooded mangrove) (Figure 6.1), how mangrove cover and infrastructure could respond if management actions intend to increase both mangrove cover and infrastructure. It will stress on the need for coordination in environmental management, based on interactions that take place between the variables in the environmental system of interest. The scenario exploration will use the tool of loop analysis called analysis of parameter change, which is the same tool used by the stepwise reconstruction of loop models (Puccia and Levins 1985).

6.3 Results

6.3.1 Assessing and managing resilience with the help of loop analysis

In environmental management, loop models can be a working reference to promote the persistence (resilience) of desired statuses of an environmental system (Figure 6.4). Similarly, for non-desired statuses of an environmental system, loop models can be a working reference to promote that non-desired statuses will not persist, i.e., to avoid the resilience of non-desired statuses.

Having loop models as such a working reference means that environmental management would have to represent the regimes under analysis, the desired and/or non-desired regimes, with loop models. Operationally, for each regime under analysis, environmental management would have to select which variables and links between variables should be included in the loop model(s) representing the regime.

The policy implication when dealing with a desired regime is that a desired regime will need to become, and remain, representable by stable loop model(s). In that way, the desired regime would tend to persist in real life, i.e., to be resilient. Thus, dealing with a desired regime in environmental management would include using loop analysis for assessing the resilience of the desired regime in order to manage the regime towards the manifestation of resilience.

Similarly, the policy implication when dealing with a non-desired regime is that a non-desired regime shall not become, or remain, representable by stable loop model(s). In that way, the non-desired regime would not tend to persist in real life, i.e., would not be resilient.

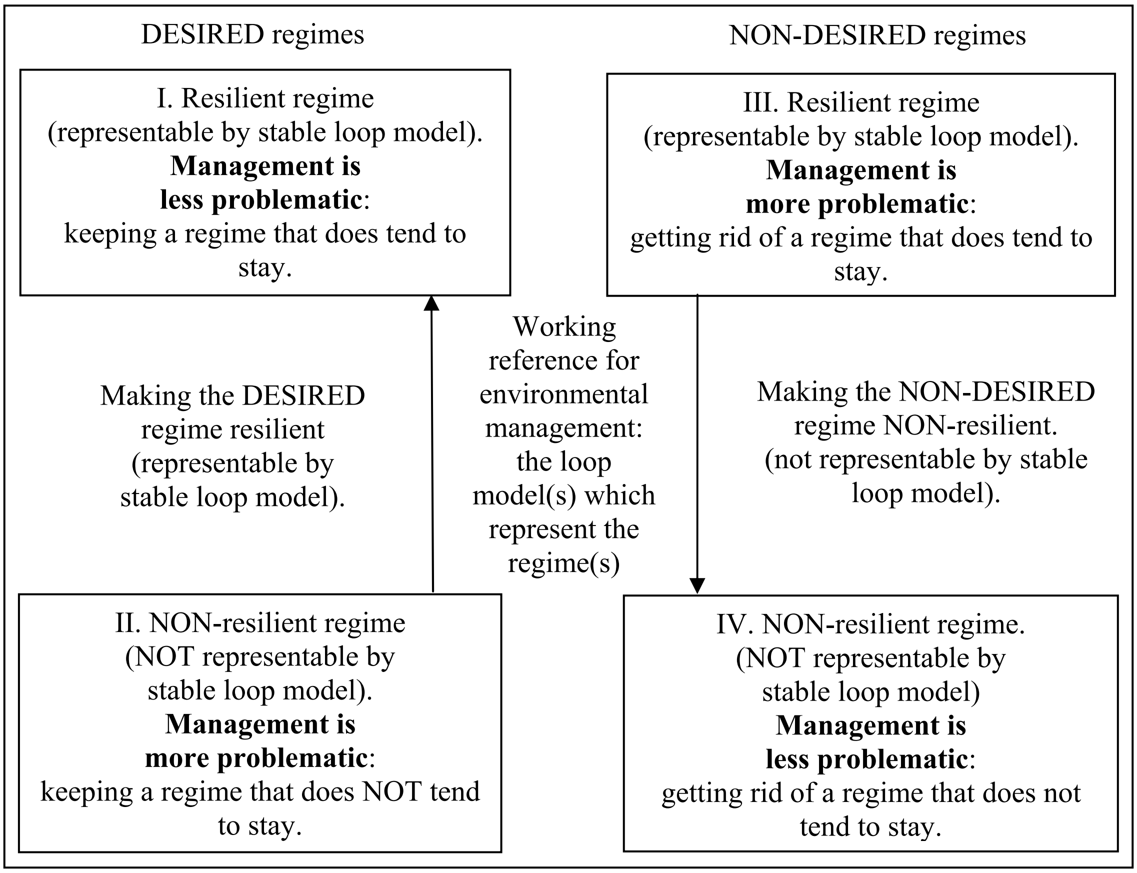


Figure 6.4 Loop models as working reference for assessing and managing the resilience of desired and non-desired regimes of an environmental system

6.3.2 Loop models representing the regimes observed in a wetland environment

Regarding the three regimes observed in a wetland environment (Figure 6.1), the stepwise reconstruction of loop models shows that models 1 and 2 can represent Regime A (Figure 6.5). Both models 1 and 2 could have produced the observed tendencies of mangrove cover and infrastructure before and after the construction of the barrier (Table 6.1). These two models differ in that Model 2 lacks the negative self-feedback on infrastructure which is present in Model 1. For the links in the models representing regimes B and C (Figure 6.5), not obtained via stepwise reconstruction of loop models but proposed one by one while considering less systematic evidence, obtaining the links and thus the models is in itself a discussion. Discussing the links in loop models is crucial in order to gain insight into the relations in the studied regimes. The discussion on the links will be presented in section 6.4.1 (Table 6.2, see also Appendix 10).

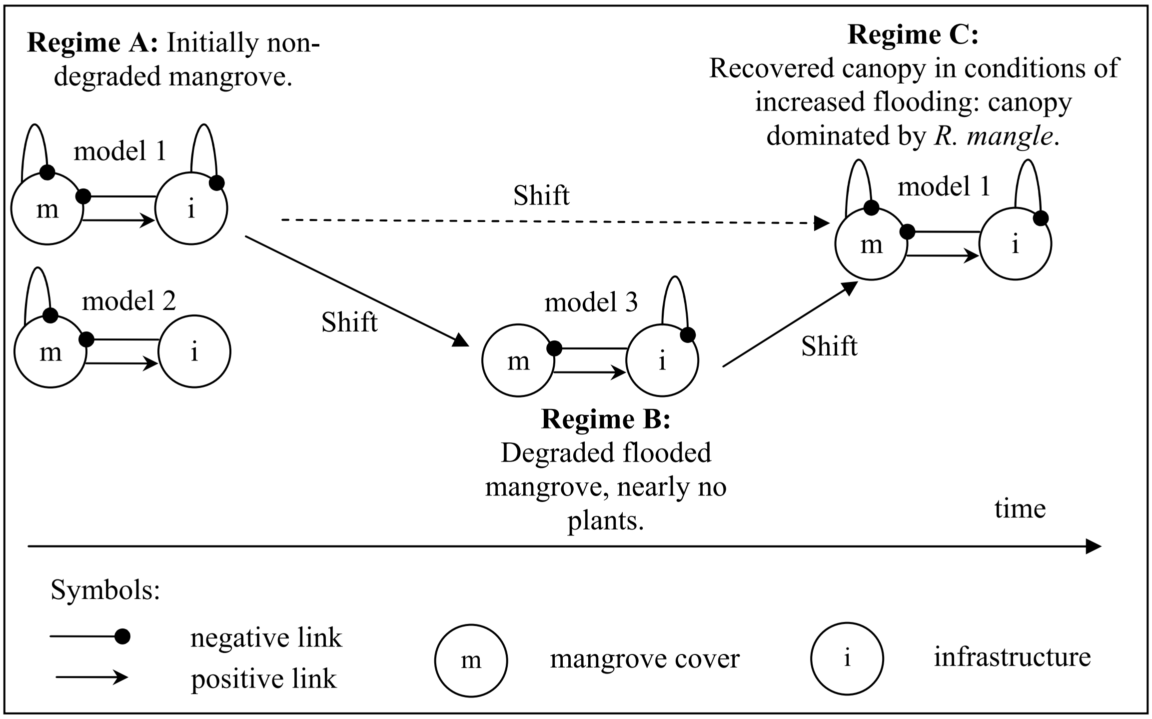


Figure 6.5 Stable loop model(s) for the three regimes of mangroves on the south coast of Havana province, Cuba

The shifts between regimes in the studied wetland environment involve two manifestations of non-linearity (Figure 6.6). The first manifestation, actually only a potential manifestation of non-linearity as later discussed, refers to the existence of more than one regime for the same territory under study, i.e., regimes A, B and C (Figure 6.6a). The second manifestation of non-linearity refers to the non-linear nature of the relation between variables in a given regime, i.e., the variables mangrove cover and infrastructure in the case of regimes A, B and C (Figure 6.6b).

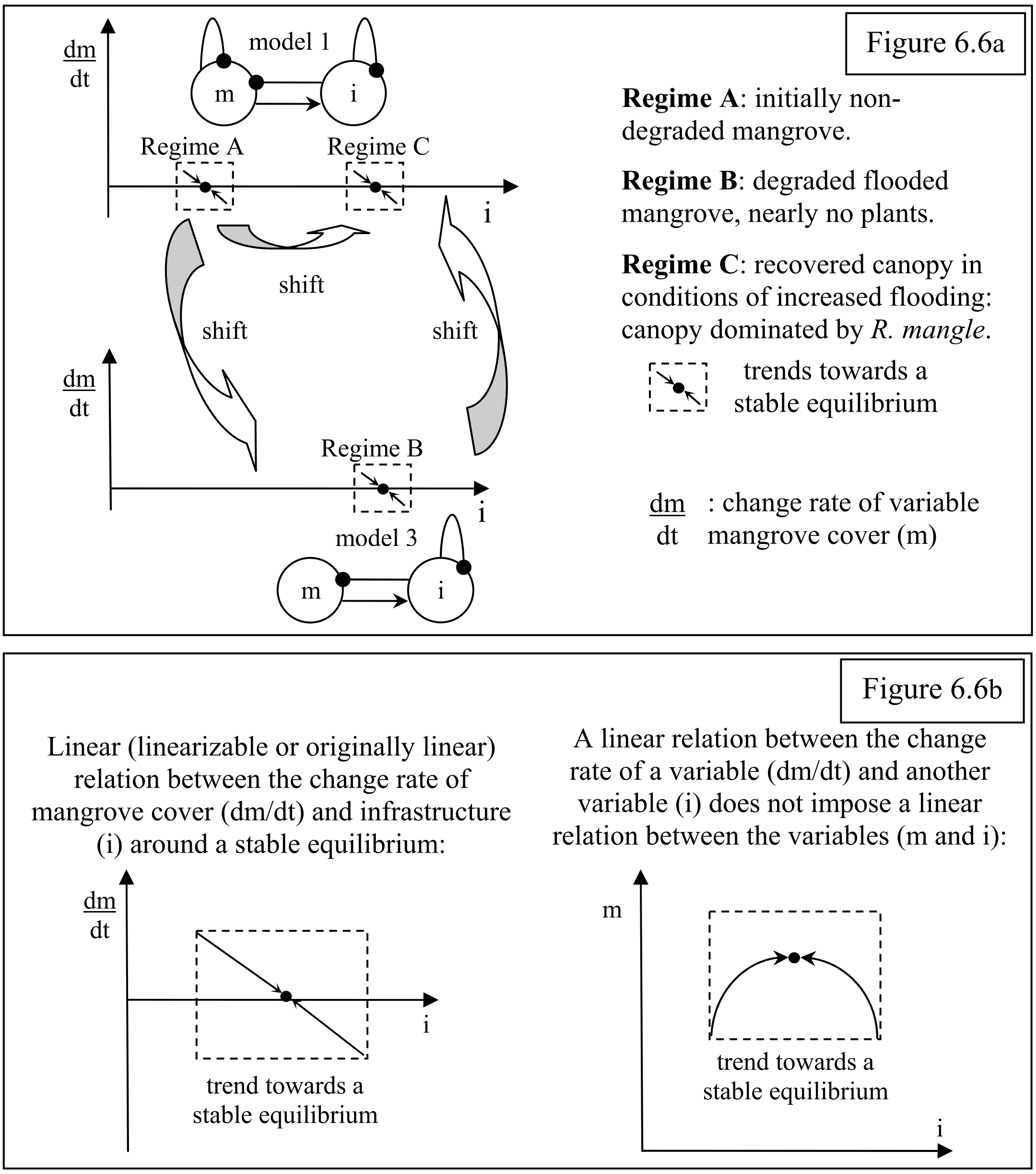


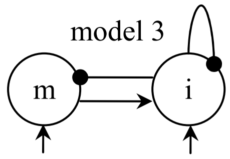
Figure 6.6 Two manifestations of non-linearity in the shifts addressed with loop analysis for three regimes in a wetland environment on the south coast of Havana province, Cuba. a: Existence of more than one regime or stable equilibrium for the same territory under study (m=mangrove cover; i=infrastructure). b: The non-linear nature of the relation between two variables in a given regime (after diagrams by Carpenter et al. 2001 and Puccia and Levins 1985)

Of the three regimes, Regime B (Degraded mangroves), is problematic for environmental management, not only because it is non-desired (IGT 1999), but also because it is resilient and so tends to persist (case III in Figure 6.4). For that resilient non-desired regime, following the loop model approach means that environmental management would try to get rid of that regime by implementing management actions that modify the qualitative structure of the environmental system. This implies that environmental management would modify the links between the existing variables so that such non-desired regime will not remain representable by a stable loop model(s) and thus will stop being resilient and will not persist in real life (e.g., the value of the variable mangrove cover would increase). If such changes put in place by environmental management promote achieving a resilient (persistent) desired-status of the environmental system, then the environmental system would have changed from a non-desired resilient regime to a desired resilient regime. Such change would be an example of change from case III to case I (Figure 6.4).

Paradoxically, implementing management actions intending to increase the value of a variable can lead to a decrease in the value of that variable. This can happen, for instance, with the variable mangrove cover of Regime B (Figure 6.7). The paradox can occur if, when environmental management attempts a change from case III to case I, the qualitative structure of the environmental system is not actually changed. The paradox of a variable changing in the opposite direction of the expectations of the actions managing the variable is captured in the qualitative structure of the environmental system as represented in loop model(s). In the case of Regime B, the paradox of a decreased mangrove cover as a result of management actions directed to increase the value of mangrove cover can occur if other actions try to directly increase the value of the other variable of the environmental system, i.e., infrastructure (Figure 6.7).

To avoid such paradox and surprise (Figure 6.7), the influence of infrastructure on itself (link from infrastructure to infrastructure, ii) should be stronger that the influence of infrastructure on mangroves (link from mangrove cover to infrastructure, mi). This can happen if the increase in infrastructure that took place in the wetland environment with the construction of the freshwater barrier would have made the construction of new infrastructure more difficult. This seems to be the case, since after the construction of the barrier, no new infrastructure influencing mangroves has been installed.

Degraded mangroves
(Regime B)



Management actions
intending to increase both mangrove
cover and infrastructure

Possible outcomes
(as indicated by the loop analysis tool response
of variables to parameter change):

infrastructure will: - increase

mangrove cover could:
- increase,
- not change!
- decrease!!!

The relations (qualitative structure of the system) represented by loop model 3 ensure that infrastructure will increase. In contrat, mangrove cover can decrease, increase, or not change, depending on the relative magnitude of the self-link of infrastructure, and the link from infrastructure to mangrove cover. This result for mangrove cover is indicated by the corresponding expression of loop analysis (Puccia and Levins 1985) as follows:

$$\text{Change in m} = \frac{(\partial f_m/\partial c)(p_{mm})(F_1^{(comp)})+(\partial f_i/\partial c)(p_{mi})(F_0^{(comp)})}{F_2}$$

Where:

1. $(\partial fm/\partial c)$ and $(\partial fi/\partial c)$ are both positive: they represent the management actions intending to increase the mangrove cover and infrastructure, respectively.
2. $(p_{mi}) = -mi < 0$: link from i to m in model 3.
3. $(F_0^{(comp)}) = -1$ feedback at level 0 is -1: definition in loop analysis (Puccia and Levins, 1985).
4. $F_2 < 0$: loop model 3 is stable (according to the stability criteria in loop analysis) (Appendix 10).
5. $(p_{mm}) = 1$: path of a variable to itself equals 1: definition in loop analysis (Puccia and Levins, 1985).
6. $(F_1^{(comp)}) = -ii < 0$: link from i to i in model 3 (explained in Table 3).

Substituting in the above expression, and making the corresponding arithmetic:

$$\text{Change in m} = \frac{(+)(1)(-ii)+(+)(-mi)(-1)}{(-)} = \frac{-ii + mi}{(-)}$$

So, the mangrove cover (m) will change as:

will diminish if ii < mi

will not change if ii = mi

will increase if ii > mi

Figure 6.7 Paradox of a variable that decreases its value although management actions had been implemented to increase its value

6.4 Discussion

6.4.1 Assessing and managing resilience with the help of loop analysis

Giving priority to the practical relevance of resilience assessment and management is fundamental. A biased (non-practical) scientific approach to resilience assessment and management can certainly yield a theoretically perfect solution, but too late, leaving people and environmental management with mostly degraded environmental systems, and thus persistent (resilient) undesirable regimes of those environmental systems. Such examples exist, for instance the very low immediate positive impact on nature conservation (use and protection) of the 1970s wide International Biological Program (IBP) (di Castri 1986; Golley 1993, 2000).

Although not obvious, adressing regime resilience with loop analysis has a very close relation to the views that predominate in the resilience literature (Walker et al. 2006a, b), i.e., the views derived from Holling (1973, 1986). The bases of such close relation, both methodological and theoretical, appear in Svirezhev's (2000) very extensive work on stability concepts in ecology.

This study has not clarified the above mentioned relation. The relation would have been more clear if in his treatment of digraphs (i.e., signed graphs and thus loop models) Svirezhev had explicitely explained how digraphs use Lyapunov stability. Lyapunov stability is an important element of the approach this thesis has taken to address resilience by choosing loop analysis (6.2.2). When dealing with qualitative stability, Svirezhev (2000) focuses on the linear algebra aspects of digraphs. Nonetheless, his style is similar to a major synthesis of loop analysis (Levins 1998).

The most important resilience literature has repeteadly explicitely supported the importance of qualitative understanding in resilience issues (Holling 1973, 1986, 2003; Walker et al. 2006a, b). In one of Holling's most influential works (Holling 1986), the author based such support on the same mathematical tools used by loop analysis, i.e., qualitative differential equations. He proposed a qualitative mathematical approach when addressing forces of change in ecosytstems that have been subjected to management (e.g., suppression of forest fire, semiarid savanna ecosystems turned into productive cattle-grazing systems, etc.).

Holling (1986) basically relied on a previous study (Ludwig et al. 1978), which is mathematically less qualitative than loop analysis. In that study, in contrast to loop analysis, equations are used to represent the influences among the studied variables. The mathematical approach of those authors remains qualitative in the sense that the values of the parameters in the equations are considered not known. In loop analysis, equations are not required and only the signs of the influences between the studied variables are typically used (Figure 6.2 and Figure 6.3). However, loop analysis can also straightforwardly incorporate equations and become a more quantitative approach (Puccia and Levins 1985; Levins 1998). Ludwig et al. (1978) could have full equations, usually a methodologically desirable situation, because that paper deals with a field of ecology with established quantitative methods, i.e., insect ecology. Indeed, having established quantitative methods and equations to represent influences between variables is not the general situation in environmental management.

Loop models as a working reference to address the persistence (resilience) of a regime of an environmental system is conceptually equivalent to addressing such regime as tending to stay in a domain of attraction in the sense of Holling (1973, 1986). It applies to both desired and non-desired regimes. For a desired regime, it means managing the regime so that it becomes persistent (resilient), i.e. representable by a stable loop model. For a non-desired regime, it means avoiding its persistence (resilience), i.e., avoiding that it is representable by a stable loop model.

Interpreting the domain of attraction in the resilience literature with the local stability regions and fixed point concepts of the more mathematical literature (Hirsch and Smale 1974; Kaplan and Glass 1995; Puccia and Levins 1985) has been implicit since Holling (1973). However, the author’s stress on the need to assess change, and on differentiating stability from resilience (Grimm and Wissel 1997; Holling 1986, 2003; Walker et al. 2006b), might have influenced the resilience literature to misunderstand the mathematical concept of stability. Loop analysis uses the concept of stability to address change (Puccia and Levins 1985). Addressing regimes with stable equilibriums as a reference with loop analysis, does not mean at all that the variables considered relevant in a regime strictly remain in fixed specific values where the stable equilibrium stands.

Assessing a regime by taking a stable loop model as reference allows addressing if the relations between the components (variables) considered relevant in that regime change or not. More important, assessing the stability of the loop model representing a resilient regime allows addressing which components (variables), and relations between them, influence these variables to have persistent values or to tend towards persistent values.

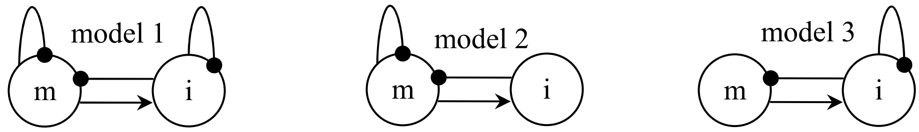
Stating that an environmental system can have different resilient regimes, each of them representable by an equilibrium (very possibly a stable equilibrium), corresponds in the resilience literature to the welcomed idea of addressing environmental systems with a multiple-equilibrium approach (Carpenter et al. 2001; Holling 1986, 2003; Kinzig et al. 2006). The changes from regime to regime are thus examples of changes from domain of attraction to domain of attraction as addressed in the resilience literature (Holling 1973; Kinzig et al. 2006). Both the existence of more than one regime in an environmental system, and the multiple-equilibrium approach of the resilience literature, have to do with manifestations of non-linearity. For the sake of clarity, the discussion on non-linearity is postponed for the next section of this chapter.

Loop analysis for addresing the resilience of three regimes in a wetland environment illustrates the relation between essential features of resilience assessment and management, and of loop analysis (Abel et al. 2006; Carpenter et al. 2001; Holling 1973, 1986; Kinzig et al. 2006; Levins 1998; Walker et al. 2006a, b). For example, having obtained two candidate models to represent the regime “Initially non-degraded mangroves” (Regime A, Figure 6.5) fits the need of resilience assessment and management to challenge different hypotheses about the interactions between the components of a regime. Different hypotheses can emerge from controversial scientific views, or from stakeholders with different perceptions.

Operationally, such hypotheses can be challenged with loop analysis by discussing the links that are in the loop models representing the regime. Discussing the links of the loop models is a must in all steps of loop analysis application. When developing loop models to represent regimes, the explorative nature of loop analysis will typically yield more than one candidate loop model to represent the regime, independent of how much empirical evidence or theoretical knowledge about the influences between the variables is available (Lane 1998; Levins 1974, 1998; Puccia and Levins 1985).

Discussing the links present in loop models is fundamental, not only when the links, and so the loop model(s), have been obtained via stepwise reconstruction as for the above mentioned regime “Initially non-degraded mangroves” (Regime A), but also in cases like the regimes “Degraded flooded mangroves” (Regime B), and “Recovered canopy in conditions of increased flooding” (Regime C), whose loop models are obtained as a result of proposing and discussing each particular link (Table 6.2).

Table 6.2 Explanation of the links present in loop models representing three regimes observed in a wetland environment. Regime A (“Initially non-degraded mangroves”; represented by models 1 and 2), Regime B (“Degraded flooded mangroves”; model 3) and Regime C (“Recovered canopy in conditions of increased flooding”; model 1). Variables: infrastructure (i) and mangrove cover (m)



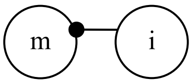
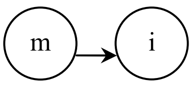
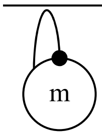
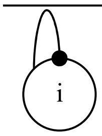
Link	Explanation
 Present in models 1 and 2 (Regimes A, C) and model 3 (Regime B).	Negative influence of infrastructure (i) on mangrove cover (m): A change in the level of infrastructure causes a change of opposite sign to the change rate of mangrove cover. How it can happen: if the barrier is modified (decrease in i) and flooding water depth is diminished, then with shallower water more seedlings can establish per year (increased change rate of m). It is also possible that an increase in infrastructure causes fewer seedlings to establish per year.
 Present in models 1 and 2 (Regimes A, C) and model 3 (Regime B).	Positive influence of mangrove cover (m) on infrastructure (i): A change in the level of mangrove cover causes a change of the same sign to the change rate of infrastructure. How it can happen: Successful mangrove restoration or naturally recolonized areas (increase in m) indicate that mangroves are healthier, and that they could stand more negative impacts. It can lead to decisions for building more infrastructure per year (increased change rate of i). It is also possible that a decrease in mangrove cover (m) raises awareness leading to less infrastructure built per year.

Table 6.2 continued

Link	Explanation
 Present in models 1 and 2 (Regimes A, C).	Negative influence of mangrove cover (m) on mangrove cover (m): A change in the level of mangrove cover causes a change of opposite sign to the change rate of mangrove cover. How it can happen: The aerial structure of established plants has diminished, for instance through gap opening in the forest canopy (decrease in m), then more saplings can establish per year in that more open understory (increased change rate of m). It is also possible that an increase in mangrove cover (m) causes less saplings to establish per year in the understory. The situations described above do not happen when the mangrove cover is low. It is the case in Regime B (Degraded flooded mangrove), thus a self-link to mangrove cover is not present in the loop model of that regime.
 Present in model 1 (Regimes A, C), and model 3 (Regime B).	Negative influence of infrastructure (i) on infrastructure (i): A change in the level of infrastructure causes a change of opposite sign to the change rate of infrastructure. How it can happen: Recently built infrastructure (increase in i) generates debate on its environmental impacts. Then, environmental regulations avoid placing new infrastructure, which results in less infrastructure built per year (decreased change rate of i).

A major challenge in resilience assessment and management, and thus in environmental management, is that environmental information is usually scarce (IPCC 2001; MEA 2005b; Petschel-Held et al. 1999). Indication that loop analysis is a suitable method for helping to deal with such scarce information is due to the way loop analysis allowed handling the scarce information available for the regimes observed in the wetland environment. When addressing the regime "Initially non-degraded mangroves" (Regime A), the tool stepwise reconstruction of loop models was restricted to rely on information of only one event influencing the variables of interest, i.e., the construction of a freshwater barrier in 1986, a direct influence on the variable infrastructure. That tool allows using the information available, no matter how scarce it is (Bodini 1998;

Puccia and Levins 1985). However, for the above-mentioned Regime A, the tool could not indicate which of the two alternative models yielded by the tool was actually representing the regime, i.e., model 1 or model 2. The scarcity of information can be more acute, for instance if for that known event influencing the variables, the response of only one variable instead of the two variables would had been recorded. With less information entering the stepwise reconstruction of loop models, a higher number of loop models can result as candidates for representing Regime A. Having more abundant information is possible, for instance by knowing the response of the variables to other events different from the construction of the barrier. Such more abundant information can reduce the number of candidate models representing a regime. It requires a more detailed environmental history of the territory.

In the absence of more information on Regime A, deciding which of the candidate models (1 and 2) actually represents the regime had to rely on a critical discussion of the links present in the models. The only difference between the two models involves a negative self-feedback on infrastructure, which is present in Model 2 but not in Model 1. Such negative self-feedback means that the placement of infrastructure generates control over new placement of infrastructure. Such control, and thus that negative feedback, was present in the form of legal regulations and awareness of the need for proper use of natural resources in the studied wetland environment. Therefore, Model 1 was presumably representing Regime A. However, the negative self-feedback on infrastructure could not have been very markedly strong, i.e., the initiation of the construction of the freshwater barrier (increase in infrastructure) did not prevent the construction of the barrier in the entire planned territory being completed. This suggests that the territory may actually have been represented by the two alternative models, i.e., models 1 and 2. Interestingly, finding out which alternative model represents a specific regime is not always indispensable in loop analysis (Lane 1998; Levins 1998); both models 1 and 2 would have helped to realize that the variable mangrove cover would decrease in the event of new infrastructure in form of a barrier, as was finally observed in reality.

When managing the persistence (resilience) of regimes with loop models as working reference (Berkes and Folke 1998; Berkes et al. 2003; Carpenter et al. 2001; Holling 1986; Kinzig et al. 2006; Walker et al. 2006b), the links between the variables in the loop models allow addressing the underlying processes or dynamics of resilience manifestation. This should help avoiding the misunderstanding of punctual values of the variables and correlations between them. It means that the existence of a desired regime, in which the variables have desired values (e.g., high crop yields in agriculture), does not prevent those variables from achieving other values typical of non-desired regimes (e.g., with low crop yields), even if the links between the variables have not changed.

For example, when addressing the regimes in the wetland environment in this study, the issue of surprising, contradictory or unexpected results in environmental management appeared, i.e., cases in which the outcomes of management differed from its goals and expectations (Abel et al. 2006; Holling 1986; Kinzig et al. 2006; Levins 1998; Puccia and Levins 1985; Walker et al. 2006b). Such surprises are paradoxical, because management actions implemented to increase the value of a variable lead to a decrease in the value of that variable.

In the observed regimes, loop analysis supported addressing such paradoxical behavior for the variable mangrove cover in the regime “Degraded flooded mangroves” (Regime B). Since loop analysis addresses the interactions of the variables in the loop model(s) representing the regime under study, the method helps to understand that the outcomes of environmental management, including the so-called surprising results, are influenced by several variables and links. Such approach stresses that the rest of the environmental system, and thus the environmental system as a whole, remains relevant for changes in specific variables and supports holistic approaches in environmental management (e.g., cross-sectorial approach).

Both methodologically and conceptually, the way in which such paradox or surprising results were addressed with loop analysis qualifies the method for exploring scenarios, a common and challenging demand to both short- and long-term environmental management (e.g., in Environmental Audit, Environmental Impact Assessment, and Planning). For instance, in the wetland environment taken as an empirical example in this study, applying loop analysis helped widening the reasoning and leads to postulate that in order to avoid a surprising result in environmental management, the reform of the Cuban environmental legislation must be effectively enforced. This reform started about 1995, and has included a new framework law on the environment, and new regulations both for Environmental Impact Assessment and Environmental Audit (GORC 1997; IGT 1999).

The impossibility of knowing the direction of change of one variable (i.e., mangrove cover) without adding extra information about the relative magnitude of two links included in the loop model representing the regime is an example of what has been named an ambiguous result in loop analysis (Puccia and Levins 1985). Such ambiguous result might be seen as a limitation of the method. However, since the method detects which links should be addressed to know the resulting change in the variable under study, such ambiguity can also be perceived as a tool to identify research priorities when addressing the resilience of a regime. In the example addressed, the priority is a more quantitative assessment of the relative magnitude of the corresponding links.

6.4.2 Non-linearity, stability and complexity when addressing resilience with the help of loop analysis

This chapter has explicitely illustrated how loop analysis deals with non-linearity, stability, and complexity, the scientifically most challenging topics when trying to find a proper methodological framework for assessing and managing resilience, and in general when doing environmental management (Levins 1998; MEA 2005b; Petschel-Held et al. 1999; Holling 1973, 1986, 2003; Walker et al. 2006b).

As already described in the previous section, the use of qualitative approaches for addressing regime shifts relates to non-linearity and the multiple-equilibrium approach for addressing environmental systems (Carpenter et al. 2001; Holling 1986, 2003; Kinzig et al. 2006).

With respect to the manifestations of nonlinearity involved in the shifts between regimes observed in a wetland environment, there are two alternative explanations for the first manifestation of nonlinearity (i.e., existence of more than one regime or stable equilibrium for the same territory under study). The first alternative explanation is illustrated by the path from Regime A to Regime C (Figure 6.1a): Each regime involved in the shift corresponds to different stable equilibriums, which can be addressed by the same loop model (loop model 1), which means that the relations between the variables are qualitatively similar in Regimes A and C. This first explanation would be actual mathematical non-linearity if the two equilibriums belonged to the same system of differential equations. But, although loop analysis can address situations involving more than one equilibrium as in the shift from Regime A to regime C, loop analysis cannot prove the system of differential equations represented by a loop model as non-linear. This is because a loop model specifies a system of differential equations only partially, through its signs, and not with full equations. Therefore, two different equilibriums, even if addressed by the same loop model as in the above discussed path of Regime A to C, could still belong to different systems of differential equations. When this is the case, the corresponding shift between regimes would not be actual mathematical non-linearity.

The second alternative for explaining the shifts between observed regimes in the wetland environment shows that a shift can certainly not be a manifestation of non-linearity (Figure 6.6a). This is illustrated by the shift from Regime A to Regime B, and the shift from Regime B to Regime C: Each regime involved in these shifts corresponds to different stable equilibriums addressed by different loop models (1 and 3), which represent different systems of qualitative differential equations. Therefore, these shifts are not manifestations of non-linearity of the mathematical system (loop model) describing the territory, but a change of the mathematical system (loop model) for describing the territory.

Regarding the above two explanations for the first manifestation of non-linearity, further conceptual clarification for addressing regime resilience with loop analysis can be found in Svirezhev (2000). That author provides support for the present study to address each regime with stable loop models (local stability), although the actual problem of environmental management demands to address several regimes and thus several stable loop models (e.g., as evidenced in the addressed wetland environment). Svirezhev (2000) mathematically explains that the progress for applying with a general approach the theory of the Lyapunov stability concept (i.e., in the present study, the actual problem of environmental management addressing several regimes) was predetermined by two fundamental theorems of Lyapunov. One of these theorems explains how such applications with general approach can be partitioned into particular cases of local stability and domain of stability (i.e., a regime in the present study).

The second manifestation of nonlinearity involved in the shifts between the regimes in the wetland environment addresses the non-linear nature of the relation between two specific variables in a given regime. While addressing change by taking stable equilibrium as reference, the scope of loop analysis is not restricted to cases where relations among variables are linear (Figure 6.6b). However, loop analysis uses a tool considered appropriate in mathematics, which postulates that those relations are usually linearizable (Levins 1974, 1998), i.e., the linear approximation is considered a proper representation of the actual relations, be they non-linear or linear. Most important, if the relations addressed by loop analysis around a particular equilibrium were actually linear (i.e., originally linear, not only after considering them linearizable), non-linearity would be still present. This is because those linear relations addressed in loop analysis focus on the rate of change of variables, and linear equations for the rate of change of variables do not impose linear relations between the variables themselves as illustrated.

Despite the usefulness of the mathematical concept of stability to address change as used in this study, formal (mathematical) clarification is needed regarding the actual mathematical scope of loop analysis. Empirical evidences indicate that such scope goes well beyond the stable loop models (i.e., stable equilibria) used in this study. In this regard, works to be re-examined should include Lane and Collins (1985), where loop analysis (i.e., the tool analysis of parameter change) provides predictions on variable change that fitted experimental data in 201 out of 211 cases (95.3%) in a land-based mesocosm experiment for plankton communities. Also Lane (1986a) should be re-analized. In this last case, the tool analysis of parameter change allowed obtaining 165 correct predictions out of 173 (94.8 %) with data from the plankton community of Delaware Bay.

Elaborating such clarification about the actual mathematical scope of loop should also examine Svirezhev's (2000) comments on the difference between stability, and ecostability. The author's mathematical interpretation of ecostability using an ecostabiliy domain is similar to the concept of domain of attraction in the resilience literature, and thus the approach taken in this study. The methodological approach of Svirezhev (2000) to ecostability can also help to go beyond that done in the present study and assess how resilient a regime is.

Another important issue is Svirezhev's (2000) observation that Lyapunov stability is a sufficient, but not a necessary, condition of ecostability. However, he recognizes that considerably complex problems (e.g., with more than two variables) would typically transform the ecostability problem to that of classic Lyapunov stability, coincidently the one in which loop analysis is explicitely based.

Also important would be to re-examine the method of time-averaging of Puccia and Levins (1985). These authors discuss under which conditions the results of this more general method would be similar to those of loop analysis. Also important is the work of Flake (1980), who argues that Levins (1974, 1975) did not mathematically restrict loop analysis to qualitatively stable systems and proposed an extension of the method for tracking the behavior of stable systems.

The above-stated need to formally (mathematically) clarify the actual mathematical scope of loop analysis will necessarily involve addressing the trajectories of the environmetal system between regimes (i.e., between equilibria). Important sources can be found in the qualitative modeling approach proposed by Petschel-Held et al. (1999).

Although Svirezhev’s (2000) work on stability concepts in ecology contains the basis for explaining the existing very close relation between adressing regime resilience with loop analysis and the views that predominate in the resilience literature, care should be taken. This is because that author’s work, despite its generality supported by mathematics, focuses on ecological communities. Such focus might render part of his contributions too narrow for the wider empirical demands of resilience assessment and management, and environmental management in general.

If mathematical loop analysis is finally proven to be practically useful for resilience assessment and management, a major challenge will be to make this tool easy to use by people without a mathematical background, as it is mostly the case in environmental management.

6.5 Conclusions

Stable loop models developed in loop analysis, a qualitative mathematical modeling method, allow representing resilient regimes. The stability of the loop model makes it possible to know how the relations between the components (variables) of a regime influence the components to have persistent values or to tend to persistent values, i.e., how the relations influence the regime in its resilience.

In environmental management, promoting the persistence (resilience) of a desired regime, and the change of non-desired ones, can be supported by analyzing what the influences (links) between the components of the regime are, or should be, so that the regime remains, or becomes, resilient and thus representable by a stable loop model.

The loop models for three regimes in a wetland environment show that these can be obtained from scarce empirical data. Loop analysis can also allow alternative interpretations of the environmental system of interest. Addressing manifestations of non-linearity in regime shifts was shown possible, too, as well as exploring scenarios and addressing examples of management actions with surprising, contradictory or unexpected results. Loop analysis can also help identifying research priorities.

Opportunities should be taken to complement qualitative approaches, like loop analysis, with other quantitative methods such as modeling and statistics, which do not always have holistic approaches, but which do reveal critical links between components of an environmental system. Quantitative methods can also complement the qualitative nature of the information to be included, and obtained, in loop analysis.

7 RESILIENCE AND RESTORATION OF MANGROVES

7.1 Introduction

The Millenium Ecosystem Assessment released in 2005 warned that “estimates of the loss of mangroves from countries with available multiyear data (representing 54% of total mangrove area at present) show that 35% of mangrove forests have disappeared in the last two decades” (MEA 2005a). That decrease in mangrove extension illustrates how inappropriate some uses of mangroves have been, and also raises a pressing question about what has happened with the mangroves not accounted for in the assessed 54%.

Although common and usually indispensable human activities like agriculture and urbanization degrade mangrove covers, mangroves have been documented as having a high capacity to recover, i.e., to be resilient (Ellison 2000; Lewis 2005; Lugo 1998; McLeod and Salm 2006). This chapter deals with the resilience of mangroves as the capacity or tendency of the mangrove plants and vegetation, and so of mangroves as ecosystems and land covers, to recover. That capacity makes non-returnable changes of mangroves into another land-cover a rare event, even when the components of mangrove ecosystems have markedly changed, as it is commonly the case when natural and human impacts influence mangrove areas.

The resilience of mangroves typically manifests itself as regeneration of the mangrove vegetation, either naturally or by restoration or planting (Capote-Fuentes and Lewis 2005; Field 1996). Restoration refers to any process that aims to assist the return of an ecosystem to a pre-existing condition (whether or not this was pristine); restoration is commonly undertaken once an ecosystem has been degraded, damaged, or destroyed (Lewis 2005; SER 2002). Although more knowledge in mangrove ecology is needed for absolute assessment of the need for and outcomes of mangrove restoration, e.g., regarding the functional status of mangroves (McKee and Faulkner 2000), knowledge currently available is more than enough to slow down the ecological mechanisms of current mangrove degradation (Ellison 2000).

Among the most important sources of impacts preventing a mangrove cover to persist are human activities that change the ground altitude (i.e., the topography) and/or the inundation of mangroves, and the properties of mangrove substrates. Common examples are man-made structures interacting with the direction of water flow, and with the depth, duration and frequency of inundation in mangroves (e.g., dikes, dams, roads). Those structures can disrupt the dispersal of the water-dispersed mangrove seedlings, and also can substantially impact abiotic factors like the composition, salinity and oxidation status of soils relevant to mangrove plants and mangrove biodiversity in general (Cohen and Lara 2003; Ensminger 1997; Field 1996; Lewis 2005; Perdomo et al. 1998).

Besides the challenge of transforming the knowledge on mangrove ecology into technologies for mangrove restoration, the major challenges for successful mangrove restoration include restoration costs and management of the organizations involved in mangrove restoration (National Research Council 1994). These aspects belong to the institutional framework of mangrove restoration. That framework includes not only formal rules like regulations (e.g., laws and juridical normatives) but also informal rules supporting customary practices. Most important, the institutional framework of mangrove restoration can be interpreted as encompassing more than just formal and informal rules, and thus includes the dynamic of the ideas behind the rules. Those ideas relate to knowledge, beliefs, power, property regimes, traditions, culture and all non-biological aspects of how the persons interact among themselves and with the other components of ecosystems (Capote and Capote-Fuentes 2004; Folke et al. 1998; Le 2001).

The objective of this chapter is to illustrate how close the findings of research on resilience of mangrove vegetation are to the practice of mangrove restoration. The findings should facilitate better collaboration between practitioners of mangrove research and mangrove restoration. Five studies cases in Cuba, Mexico and United States of America are addressed.

7.2 Methodology

Based on the study cases, and literature directly relevant to mangrove ecology and restoration, a diagram is presented for generalizing on trajectories of change between a mangrove cover and non-mangrove covers (Capote-Fuentes and Lewis 2005; Ellison 2000; Lewis 2005; Lugo 1998; McLeod and Salm 2006; MEA 2005a; Menéndez 2000). The trajectories represent both degradation of the mangrove cover, mainly due to anthropogenic impacts, and manifestation of resilience of the mangrove cover as a tendency to recover via natural regeneration and restoration.

7.2.1 Study cases

The chapter addresses five studies cases in Cuba (three cases), Mexico (one case) and USA (one case) (Figure 7.1).

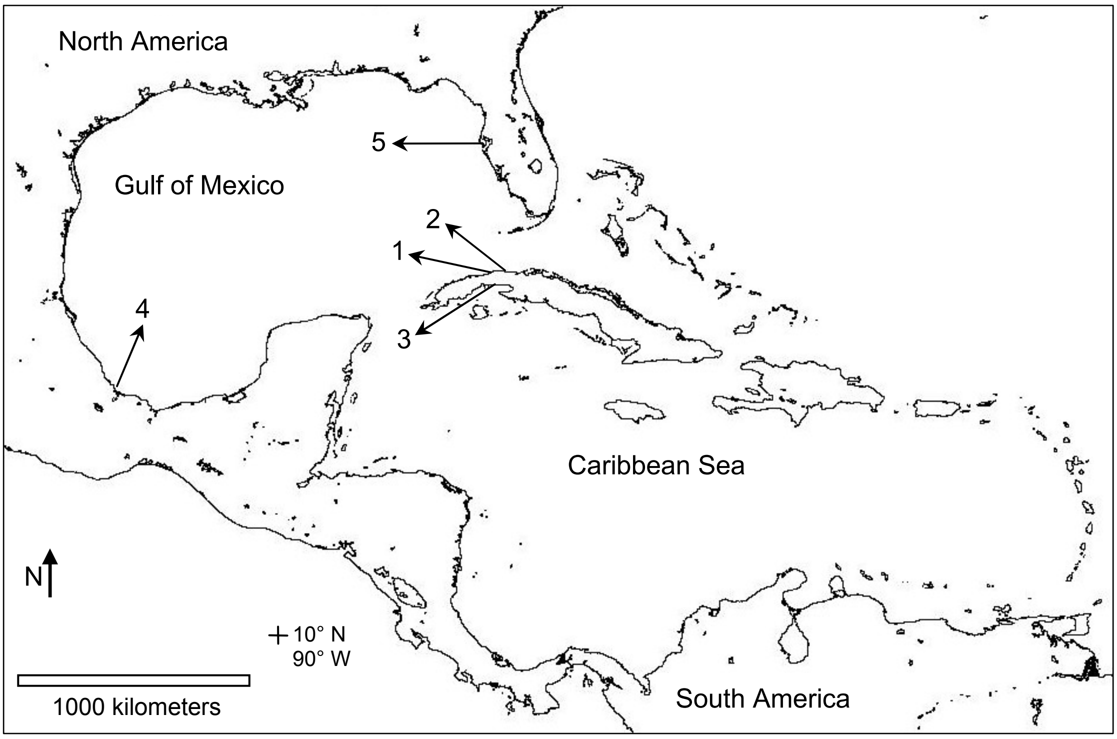


Figure 7.1. Location of five study cases on resilience and restoration of mangroves in Cuba (1, 2, 3), Mexico (4) and USA (5)

The information presented for each study case always regards changes in mangrove vegetation, ground altitude and/or inundation. Furthermore, the institutional framework of mangrove restoration is addressed, specifically the presence of formal institutions directly relevant for the recovery of the mangrove areas (i.e., protected area and/or restoration project formally established). Primary data is only available for one study case in Cuba; secondary sources provide the information for all other study cases. Therefore, the study cases do not all address exactly the same aspects.

Study case 1

Sudy case 1 focuses on mangroves located in Cuba (Figure 7.1) in a locality named Santa Ana in the municipality of Playa, Havana City (Capote-Fuentes 2003a). These mangroves occupy about 2 km^2 on the mouth of the Santa Ana river. This mangrove area is typical of the north coast of Cuba, where mangroves can develop in small areas of a predominantly rocky coast, mainly in river mouths or where the low altitude of the ground allows tidal exchange and sediment accumulation suitable for mangrove development. In the site addressed in this study case the most frequent mangrove species are *L. racemosa* and *A. germinans*, followed by *R. mangle*.

The data was surveyed by monitoring vegetation cover in a gap of about 1 ha (Capote-Fuentes 2003a). In this gap, natural regeneration of vegetation had begun after the plants had previously been removed by logging; an artificial deposit of sand existed from 1983-1986. Monitoring was done in two transects crossing the gap from seaward to landward of the mangroves. Each transect had ten 1 m x 1 m plots. The vegetation cover was surveyed (visual estimation) in February 1999, February 2000, May 2001 and May 2002. The data are presented in a graph with the temporal trend of the vegetation cover during the 39 month period.

Study case 2

Study case 2 addresses remnant patches of mangroves located in Cuba (Figure 7.1) in a locality named Sibarimar in the municipality of Habana del Este, Havana City (Garcell and Capote-Fuentes in prep.; Roig and Capote-Fuentes in prep.). Local mangroves relate to a catena naturally found in some sectors of Cuba’s north coast: sandy beach, sandy coast vegetation, coastal thickets and mangroves associated with coastal lagoons. Major transformation of the former mangrove cover occurred in the 1950s through urbanization, which had started in 1917. Urbanization has been related to tourism, since the local beaches are considered the best sandy beaches of the Havana City province.

Most of the mangrove patches lie in two protected areas formally established in 1999. In 2005, the two largest patches in these protected areas occupied 0.7 km^2 and 0.1 km^2; in their pristine status prior to urbanization they covered 1.6 km^2 and 0.5 km^2. The most frequent mangrove species is *L. racemosa*, followed by *A. germinans* and *R. mangle*.

The data were surveyed in 7 transects (Garcell and Capote-Fuentes in prep.; Roig and Capote-Fuentes in prep.). Each transect started in a mangrove patch, crossed the border between that patch and the surrounding land cover, and ended in that land cover. Each transect included two 10 m x 10 m plots. One plot was located at one end of the transect in the mangrove patch, the other plot was located in the land cover adjacent to the mangrove patch. A plot was representative of the vegetation composition and structure of the mangrove patch or the adjacent land cover. In each plot, the floristic composition was recorded, and the vegetation cover (total, and per species) was visually estimated. Over the entire transect, the ground altitude was surveyed every 2 m applying the principle of communicating vessels with a plastic hose containing water (Flores-Verdugo 2003). The values of ground altitude do not refer to sea level or any other absolute reference; each ground altitude value expresses the difference between one point and the previous point where the ground altitude was measured.

The differences between mangrove patches and adjacent land-covers were determined and refer to percentage of mangrove cover, number of mangrove species and ground altitude. For each a mangrove patch and its adjacent land cover addressed in a transect, the mean ground altitude was calculated (Garcell and Capote-Fuentes in prep.; Roig and Capote-Fuentes in prep.).

Study case 3

Study case 3 deals with mangroves located in Cuba on the south coast of Havana province (Figure 7.1; sector II Figure 2.1 and Figure 3.1). These mangroves are located in a locality named Majana in the municipality of Artemisa, Havana province. Across wetlands, including mangroves, a barrier to the freshwater flowing towards the sea was built along the coast in 1989. Besides enhancing the quality and quantity of groundwater landwards, the barrier caused mangrove dieback leading to gaps in the mangrove cover and conversion of mangroves into flooded mangroves. The barrier was one the measures implemented to guarantee an adequate freshwater supply for an important agricultural area and settlements landward of the barrier. The barrier is a non-paved road, with gravel on top of clay layers (Chapter 2). The most frequent mangrove species are *A. germinans* and *R. mangle*, followed by *L. racemosa*.

The data comprise water level (dry and rainy seasons) landward and seaward of the barrier, and the predominance of mangrove species in landward, flooded and seaward mangroves (data collection see Chapter 4).

Study case 4

Study case 4 addresses mangroves located in Mexico (Figure 7.1) in a locality named La Mancha in the municipality of Actopan, Veracruz state (Capote-Fuentes 2003b). The coast of the central part of the Veracruz state is characterized by a mosaic of open water, mangroves, „tifales“, „popales“, beaches and dunes. The mangroves surround the coastal lagoon La Mancha; the most frequent mangrove species are *A. germinans* and *L. racemosa*, followed by *R. mangle* and *C. erectus*. Likely in 1996, a road mainly composed of compacted clay was built from the landward side of the mangroves to the lagoon. The road is a gap in the mangrove cover of about 100 m long and 15 to 50 m wide. To build the road, vegetation was removed and filling was placed to increase the ground altitude. A restoration project has attempted to promote the recovery of the mangrove cover. From 1997 to 2003, the project included removing filling, establishing creeks, creating a nursery for mangrove seedlings, and planting mangroves.

The restoration of the mangroves impacted by the road is part of a wider solution for the local environmental management: the formal creation of a protected area (INECOL 2003). The restoration project has included negotiations among stakeholders, mainly a private land-owner who had encouraged the construction of the road, the local community, which traditionally fished in the lagoon, and the Institute of Ecology A.C. (INECOL) (based in the Veracruz state). This institute has a long research and environmental advocacy in La Mancha, where it coordinates CICOLMA (Center for Research on Coastal Ecosystems La Mancha). Several organizations from the state and federal levels have also been involved in the negotiations.

The data were surveyed in a cross-section transect placed over the road approximately halfway between the lagoon and the outer boundary of the mangroves (Capote-Fuentes 2003b). The two ends of the transect were located in the mangrove areas bordering the road. Only mangrove plants were present in the transect; their height and species were recorded. Over the entire transect, the ground altitude was measured every 2 m after Flores-Verdugo (2003) (see study case 2). In about the middle of the transect, the depth of the allochthonous filling was investigated with a driller (Forestry Suppliers), which also allowed measuring the water level (water table) with a plastic rubber. The data are presented as a cross-section profile of the road built across the mangroves (Capote-Fuentes 2003b).

Study case 5

Study case 5 focuses on a mangrove restoration project that was conducted in the USA (Figure 7.1) in a locality named Cross Bayou Canal in the Pinellas County of the Tampa Bay area, Florida state (Capote-Fuentes and Lewis 2005; Lewis 2004; Lewis et al. 2005). In Florida, approximately 2000 km^2 of mangroves remain from an estimated historical cover of 2600 km^2. The decrease is largely associated with pressures from the human population (e.g., urbanization and mosquito control impoundments). Mangrove forests in Florida naturally occur in a catena including mangroves, low marshes, high marshes and salt flats (Lewis 2005). The mangroves in the Tampa Bay area are representative of the Florida state mangroves.

The mangrove restoration project belongs to a wider project including marsh restoration and shoreline stabilization. The mangrove restoration project was conducted in an area where human impacts (e.g., through dredging) had caused gaps in the mangrove cover and conversion of mangroves into ruderal vegetation. The area is about 4.3 ha, including about 1.9 ha of upland fill and 2.4 ha of mangroves or open water. The most frequent mangrove species were *A. germinans* and *L. racemosa*, followed by *R. mangle* and *C. erectus*. The project included the excavation of the uplands to ground altitudes appropriate for mangrove development. The excavations were done in the period May-July of 1999. Concerning mangroves, the primary goal of the restoration was to establish a typical Tampa Bay mangrove forest composed of *R. mangle*, *A. germinans*, and *L. racemosa*, with a typical transition zone including *C. erectus*. The restoration project did not plant mangroves.

The data were surveyed by monitoring vegetation cover (Lewis, 2004). The percentage of vegetation cover (total, and for the mangrove cover) was visually estimated following a stratified random sampling. The two blocks where the random sampling was conducted differ in ground altitude. In each block, sampling was conducted in ten 1 m x 1 m plots randomly placed. The surveys took place at month number 3, 6, 9, 12, 18, 24, 36, 48 and 60 after the above-mentioned excavations. The final survey took place in November 2004. The data are presented in a graph with the temporal trend of the vegetation cover during the 60 months addressed by the monitoring (Lewis 2004; Lewis et al. 2005).

7.3 Results

Relatively well preserved mangrove areas, including pristine ones, naturally experience impacts on the mangrove cover; for instance when lightning opens gaps in the mangrove canopy (Figure 7.2). The consequences of these impacts will typically not lead to the degradation of the entire mangrove area. Further impacts, for example those associated with urbanization, can progressively transform mangroves into non-mangrove covers with remaining mangrove patches. Further impacts can transform mangroves into non-mangrove covers without any mangrove species (Figure 7.2).

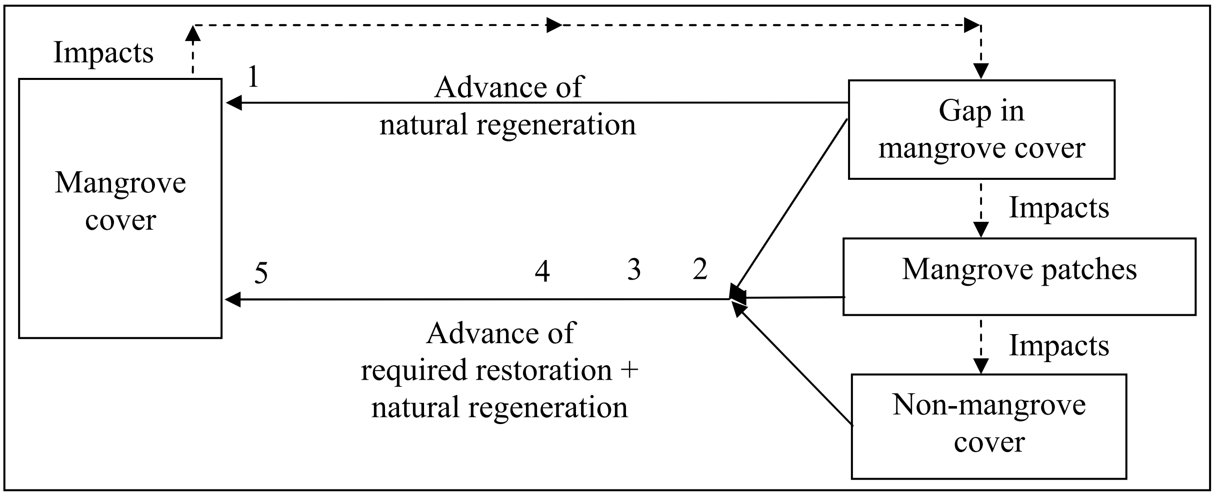


Figure 7.2. Trajectories of change between a mangrove cover and non-mangrove covers: the role of natural regeneration and restoration in the manifestation of resilience of the mangrove cover. Numbers represent five study cases in: Cuba (1, 2 and 3); Mexico (4); and USA (5). Positions of numbers on the trajectories indicate relative advancement of the recovery of the mangrove cover.

When impacts on the mangrove cover are spatially concentrated, such as the above-mentioned gaps in a mangrove cover, the recovery of the mangrove cover can entirely rely on natural regeneration processes (Figure 7.2). However, if the mangrove cover has been degraded further into a non-mangrove cover with mangrove patches, or non-mangrove covers without any mangrove species (Figure 7.2), then it is usually human impacts that have modified the ground altitude (i.e., topography) and/or the inundation of the mangrove area. In this case, natural regeneration remains an important process in the recovery of the mangrove cover, but this will usually need to be promoted or enhanced by restoration measures (Figure 7.2).

The five studies cases (Table 7.1) are examples for the trajectories of change (Figure 7.2).

Table 7.1 Mangrove vegetation, ground altitude and/or inundation, and institutional framework of mangrove restoration in five study cases

Study case	Mangrove vegetation	Ground altitude and/or inundation	Institutional framework of mangrove restoration	Main references
1. Cuba	Full recovery of the mangrove cover via natural regeneration in a forest gap. Logging had previously removed the plants and an artificial deposit of sand existed.	No permanent changes in ground altitude.	Area does not have any formal site-specific conservation status (e.g., it is not a protected area).	Capote-Fuentes 2003a.
2. Cuba	Since mangroves are patches left by urbanization, these patches differ in the percentage of mangrove cover with respect to adjacent land covers. The number of mangrove species can remain the same in a mangrove patch and its adjacent land cover.	A mangrove patch can have higher or lower ground altitude than its adjacent land cover.	Most of the patches studied are included in two protected areas formally established in 1999.	Garcell and Capote-Fuentes (in prep.), Roig and Capote-Fuentes (in prep.).
3. Cuba	Partial recovery of the mangrove cover via natural regeneration in flooded mangroves located landward of a freshwater barrier.	The barrier constitutes an increase in ground altitude and thus, in both the dry and the rainy season, the water level tends to be higher in the mangroves located landward of the barrier.	The area does not have any formal site-specific protection status (e.g., it is not a protected area). The awareness of the need for mangrove restoration was stronly supported by an Environmental Audit (IGT, 1999), a new tool for environmental management in Cuba, as result of the modernization of the Cuban environmental legislation in the 1990s (GORC, 1997). According to the new regulations, an Environmental Impact Assessment should have been conducted prior to the construction of the barrier.	Present thesis.

Tabelle 7.1 continued

4. Mexico	Partial recovery of the mangrove cover on fillings (road) via natural regeneration and partial mangrove restoration.	The road constitutes an increase in ground altitude. Part of the filling used to build the road remains on top of the mangrove soil and prevents the natural vertical movement of the water level (e.g., as it would naturally move under the influence of sea tides).	The restoration of the mangroves impacted by the road is part of a wider solution for the local environmental management: the formal creation of a protected area.	Capote-Fuentes (2003b), INECOL (2003).
5. USA	Full recovery of the mangrove cover after natural regeneration had been promoted by a mangrove restoration project. Human impacts (e.g., through dredging) had caused gaps in the mangrove cover and conversion of mangroves into ruderal vegetation. The project did not include planting of mangroves.	The restoration project included the clearing of all upland vegetation and removal of accumulated trash from the uplands, excavation of the uplands to appropriate elevations for mangrove development, and the construction of tidal creeks to provide additional tidal inundation and fish access to the restored site.	The mangrove restoration project has included formal procedures like entering a consent decree after damages to natural resources, designing and proposing the restoration project, obtaining the permission to implement the project, and certifying the achievement of agreed success criteria.	Lewis (2004), Lewis et al. (2005).

In Cuba (study case 1), natural regeneration of mangrove vegetation led to full recovery of the mangrove cover in a gap of about 1 ha from which plants had been removed by logging (Figure 7.3a). In the USA (study case 5), natural regeneration led to full recovery of the mangrove cover after restoration measures regarding ground altitude and thus inundation (Figure 7.3b).

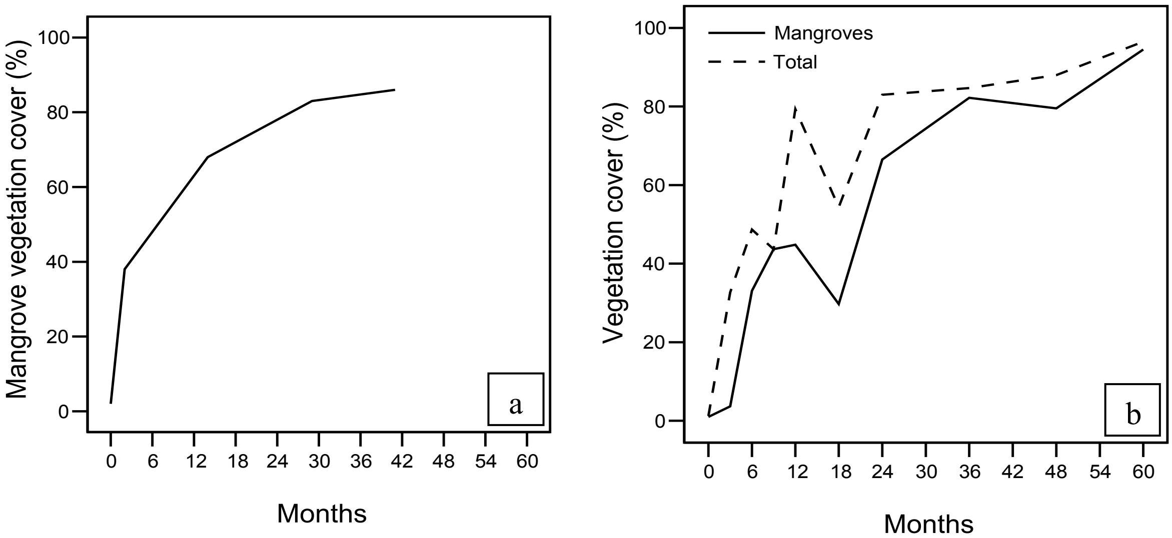


Figure 7.3. Temporal trend of vegetation cover towards complete recovery of the mangrove cover. a: via natural regeneration in a gap where restoration was not needed since only plants had been affected by logging in an area without formal status of protected area (study case 1, Cuba). b: via natural regeneration after a formal restoration project had rectified mangrove ground altitude and thus inundation (study case 5, USA). Source: modified from Capote-Fuentes and Lewis (2005) with data from Capote-Fuentes (2003a), Lewis (2004) and Lewis et al. (2005)

In Mexico (study case 4), natural regeneration led to partial recovery of the mangrove cover after the allochthonous filling had been partially removed (Figure 7.4).

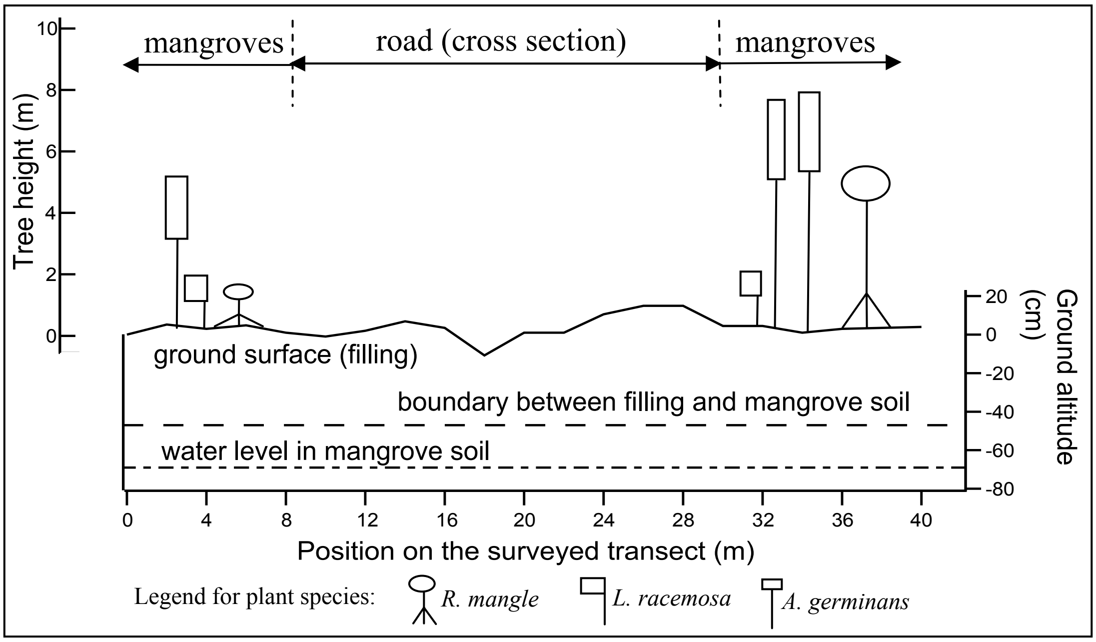


Figure 7.4. Cross-section of partially restored mangroves on a road built perpendicular to a lagoon shore: the scattered vegetation on the road indicates that the recovery of the mangrove cover has been only partial. Mangrove restoration, part of a wider program that created a formal protected area, has not completely removed filling and rectified ground altitude and inundation (study case 4, Mexico). Source: modified from Capote-Fuentes (2003b)

In another Cuban locality (study case 3), natural regeneration of mangroves led to partial recovery of the mangrove cover in mangroves flooded through a man-made freshwater barrier, which caused massive mangrove mortality (Figure 7.5).

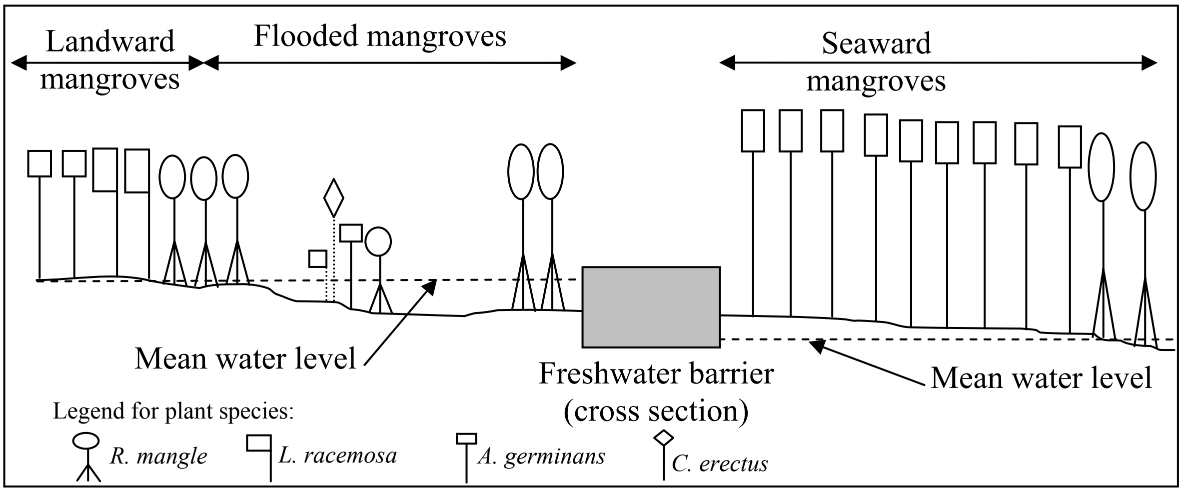


Figure 7.5. Partial recovery of the mangrove cover (via natural regeneration) in flooded mangroves landward of freshwater barrier built parallel to the coast in an area without formal status of protected area (study case 3, Cuba). Mean water level in landward mangroves: dry season -4 cm, rainy season 3 cm, N=3; in flooded mangroves: dry season 38 cm, rainy season 49 cm, N=10; in seaward mangroves: dry season -6 cm, rainy season 3 cm, N=19

In the third Cuban locality (study case 2), where urbanization has left mangrove patches, differences in ground altitude between mangrove patches and the adjacent land covers are concomitant with high differences in mangrove cover (60 to 93%) (Table 7.2). Despite the changes in mangrove cover, mangrove species can persist in the land cover adjacent to the mangrove patch (Table 7.2).

Table 7.2 Remnant mangrove patches included in formally protected areas (study case 2, Cuba). Seven transects, each from a mangrove patch to adjacent land cover. The column "Difference in ground altitude" refers to the difference of the adjacent land-cover with respect to the mangrove patch (Source: modified from Garcell and Capote-Fuentes in prep.; and Roig and Capote-Fuentes in prep.)

Transect		Difference in ground altitude (cm)	Mangrove cover (%)	Number of mangrove species
1	mangrove	-81.2	75	2
	adjacent land-cover		0	0
2	mangrove	-47.4	75	1
	adjacent land-cover		0	0
3	mangrove	11.1	70	1
	adjacent land-cover		1	1
4	mangrove	-51.2	100	2
	adjacent land-cover		25	2
5	mangrove	71.3	95	1
	adjacent land-cover		2	1
6	mangrove	73.8	70	1
	adjacent land-cover		10	1
7	mangrove	131.1	90	1
	adjacent land-cover		0	0

7.4 Discussion

Research on resilience of mangrove vegetation indicates that changes in mangrove ground altitude and/or inundation and in the composition of mangrove substrates (soils) are important impacts leading to the degradation of a mangrove cover. Therefore, those impacts should have high priority when considering what to assess, monitor and manage in order to promote recovery of degraded mangroves. For mangrove restoration, avoiding the conditions that impede natural regeneration of the mangroves will be the fundamental ecological task.

When the ground altitude of mangrove areas is increased or reduced, mangrove species will usually diminish their presence in the vegetation cover. Sites with increased ground altitudes tend to be less frequently inundated and thus less easily reached by mangrove seedlings, which are naturally buoyant. On the other hand, sites with lower ground altitude can be easily reached by mangrove seedlings, but the resulting water might be too deep, thus prevent seedling anchoring and sapling development (Ellison 2000; Lewis 2005). Also, changes in ground altitude and/or inundation will commonly cause changes in soil salinity, a relevant abiotic factor influencing the dominance of mangrove plants. Soil salinity respectively increases or decreases in places that are less or more frequently inundated (Field 1996).

Linear structures constructed with the aim of blocking a water flow, e.g., dams and roads, initially change the ground altitude and substrate composition only where the structure is located. However, even linear structures that have not aimed at changing the inundation pattern of areas much larger than the area occupied by the linear structure itself (e.g., roads and pipelines), frequently lead to undesired changes in inundation. The greatest impacts on the mangrove cover can be expected when the structure is perpendicular to the water flow. The resulting blockage of the water flow will typically drastically change the frequency, duration and depth of flooding on one or both sides of the structure (Cohen and Lara 2003; Ensminger 1997; Perdomo et al. 1998).

Even when the mangrove ground altitude and/or inundation and the composition of mangrove substrates (soils) have been impacted, restoration is usually ecologically possible. After such impacts have been corrected to levels naturally occurring in mangroves, the trend of the natural regeneration of the mangroves towards full recovery of the mangrove cover can show patterns similar to the natural regeneration observed in mangroves where restoration was not needed. Only when mangrove seedlings cannot naturally reach the restored sites or are not naturally available, should restoration measures involve spreading seedlings or planting (Capote-Fuentes and Lewis 2005; Lewis 2005).

Restoration can enhance the process of natural regeneration and thus the recovery of the mangrove cover in degraded mangroves. Typical measures include removing allochthonous fillings, ameliorating the impacted flow of water so that extreme water levels do not occur, and appropriate increase or decrease of ground altitudes. However, such restoration measures can again lead to negative impacts. For example, the removal of fillings from above mangrove soils will decrease the ground altitude, increase flooding, and make the mangrove soil underneath the fillings more available for the establishment of mangrove seedlings. This should in principle enhance mangrove regeneration via the water dispersal of mangrove seedlings. However, the fillings may have caused compression and subsidence of the underlying soils. In this case, removing the fillings might increase the water level such that the establishment of mangrove seedlings and thus mangrove recovery is made difficult.

Other relevant challenges to restoration are changes in soil properties, e.g., the decrease in soil redox potential, which is typical for extended periods of soil water logging (McKee and Faulkner 2000). Also of importance is the fact that if there is less flooding (depth, frequency or duration) in wetland soils that have been subjected to natural or human-induced long-term flooding, then extreme acidification can prevent the development of any natural or cultural vegetation (Boto 1984; Siciliano and Germida 2005; Thomas et al. 2004). In other cases not addressed in this chapter, pollution of soils (e.g., as a consequence of fertilizer use), together with changed ground altitude and/or inundation, may be also an important challenge when attempting successful mangrove restoration in mangrove areas abandoned by forms of mangrove-unfriendly shrimp farming (McLeod and Salm 2006).

The institutional framework of each territory makes different local strategies for recovery of an impacted mangrove cover necessary (Capote-Fuentes and Lewis 2005; Le 2001; Lewis 2005; McLeod and Salm 2006). In highly formalized institutional settings, restoration projects will require close observation of legal regulations and procedures. In contrast, in less formalized institutional settings, e.g., in areas with no site-specific conservation status, achieving full recovery of mangroves via natural regeneration can be just a matter of hoping for the absence of new impacts impeding natural regeneration. In protected areas of recent establishment, the involvement of local stakeholders (e.g., research centers) with acknowledged committment to nature conservation can be crucial for addressing the land-use conflicts that had led to mangrove degradation and made mangrove restoration necessary.

The greatest institutional challenge is when other environmental management priorities (e.g., water use in agriculture) cannot smoothly accommodate mangrove protection or restoration. Declaring protected areas can be only part of the solution. For instance, in highly fragmented mangroves (e.g., by urbanization), mangrove patches located outside protected areas remain critical for the ecological integrity of the mangroves included in protected areas and the entire territory.

7.5 Conclusions

Restoration of mangrove covers should be more frequently undertaken because the ecological knowledge needed is basically available. However, more research is needed on how to direct post-restoration trends towards a desired or planned floristic composition and vegetation structure of the mangroves. Other important research issues regard the assessment of the functional status of restored mangroves and the ecosystem services mangroves provide. The consequences of restoration for all non-floristic components of mangrove biodiversity also demand more research.

8 GENERAL DISCUSSION

The acknowledged importance of ecosystem services of mangroves for human well-being (e.g., food and water consumption) (MEA 2005a), together with the abundant existing body of knowledge on mangrove ecology (McLeod and Salm 2006), makes the extensive worldwide mangrove degradation of 1980-2005 difficult to understand (i.e., 35% of documented mangrove forests have disappeared).

Transforming mangroves, and in general transforming ecosystems, can be indispensable for adequating a territory to reasonable human needs, but the existing resources for mangrove restoration (e.g., human resources, knowledge, technological resources) evidences that there has been lack of success in coordinating the existing worldwide commitment and willingness for mangrove conservation (i.e., protection and use including restoration).

Mangrove conservation, which is based in mangrove resilience, is first of all a practical topic of social relevance (Capote-Fuentes and Lewis 2005; Dahdouh-Guebas and Koedam 2006; Ellison 2000; Lewis 2005). Therefore, findings from science on mangrove resilience must contribute to mangrove conservation. This study shows that the concept of resilience can be of practical value for mangrove conservation. When studying degraded mangroves, focus should be not only on how mangroves can recover (succession), but also on what the reasons for the degradation were, how their recovery can be enhanced (e.g., restoration), and how new degradation can be prevented (protection).

For mangrove ecology, this study has methodological value regarding the way the temporal and spatial demands of mangrove resilience were approached. In fields related to mangrove resilience (e.g., historical vegetation degradation), other authors had already proposed combinations of methods similar to those used in this study, but this study is pioneering in that it actually applies, in a single study, a combination of remote sensing (with satellite images), field ecological surveys (e.g., ground altitude, water level, soil redox potential), analysis of the interacting role of restoration and natural regeneration of the mangrove vegetation, and the institutional framework of mangrove restoration.

With respect to more basic ecology, testing a recent theoretical framework (Berger and Hildenbrandt 2003) on one of the longest (since 1933) unsolved problems in ecology, i.e., the biomass-density trajectories of self-thinning (Reynolds and Ford 2005; Shaw 2006), with data surveyed in a Cuban research station, opens a potential for re-directing Cuban vegetation science and ecology towards a more general approach. The future input of Cuban ecology to world level science will be based on the disparate historical, scientific partners in Cuban ecology. The existing generation of senior Cuban scientists has built up expertise and influenced young Cuban scientists at the intersection of two complementary styles, i.e., a western style (mainly from USA and western Europe), and an eastern European style (not only Russian). A major overarching topic for such input from Cuban ecology, which manifests in the temporal and spatial aspects of resilience research, is a still pending consensus about the spatial (i.e., territorial) and complex features of ecosystems. This goes together with an improved complementation of the ecological concept of ecosystem and the geographical concept of landscape. As manifested in resilience research by the influence of Holling and his colleagues (Holling 1997), the roles of scientists and leadership in science can be a critical ingredient for the outcomes of science. Unfortunately, the potential input of the Cuban ecology to world level science suffered an irreparable loss when one of the Cuban ecologists best positioned to attempt a book on ecology of world level interest, Ricardo A. Herrera Peraza, died (2006). His works on structure, functioning, and succession of terrestrial ecosystems (particularly tropical forests), land-use and land-cover change, among others (e.g., mycorrhizal taxonomy and ecology), most of which are still unpublished (Herrera-Peraza et al. 1988, 1997), had made him ready to locate Cuban ecology at the frontier of world level science, e.g., regarding the relation of ecological succession and the so-called tentative fourth law of thermodynamics (Jørgensen 2000).

The resilience concept, and thus the scientists who have most supported the use of this concept (e.g., Holling, Walker, Carpenter, Folke), will remain facing the strong rise of other concepts like vulnerability and adaptive capacity. Concepts and the organizations that support such concepts, mutually enhance their strength. The concepts of vulnerability and adaptive capacity are linked to strong organizations, including the United Nations University Institute for Environment and Human Security (UNU-EHS)

(Thywissen 2006). However, the above-mentioned resilience promoters have proved to be skilled in managing the formal framework for promoting the use of the resilience concept, e.g., by launching the journal Conservation Ecology (Holling 1997). A decisive event will be the conference Resilience 2008, announced as the “first high-level science symposium ever on the concept of resilience” (Resilience 2008). The organizers of the conference are led by the Resilience Alliance (Resilience Alliance 2007), an international research organization established in 1999 under the leadership of Holling, who undoubtedly brought to that alliance the unusual style he had manifested while coordinating the book Adaptive Environmental Assessment and Management (Holling 1978), and directing the International Institute for Applied Systems Analysis (IIASA) (1981-1984). The conference, to be held in Stockholm (Sweden), could be a benchmark in that Europe could become the main source of the development of ecosystem ecology. Non-Europeans will be keynote speakers (i.e., Elinor Ostrom, Crawford S. Holling and Steve Carpenter). The two most influential scientists of the American school of ecosystem ecology after the 1950s (i.e., E.P. and H.T. Odum), with whom the American scientists of the resilience research community share theoretical foundations, died in 2002.

The main scientific contribution of this study is the proposal of qualitative mathematical modeling, specifically loop analysis (Levins 1998), in a methodological approach for improving the assessment and management of resilience of environmental systems. Although it has been presented through empirical information, and can already be applied to practical situations in environmental management, the approach obviously needs more mathematical work. That work should focus on how to complement the representation of resilient regimes with stable loop models (mathematical stable equilibria) with the mathematical exploration of trajectories between stable equilibria, e.g., with the approach of Petschel-Held et al. (1999). The framework in this study prepares the way for approaching resilience in the absolute sense (e.g., explicitly including socioeconomic information) and not only focusing on the ecological aspects of resilience as was the emphasis in this study.

9 GENERAL CONCLUSIONS

On the south coast of Havana province (Cuba), the mangrove cover has manifested resilience (i.e., has partially recovered) from the flooding induced by a freshwater barrier built during 1985-1991. However, achieving the important goals of the barrier of maintaining or increasing the availability of freshwater for agriculture and human consumption has led to a decrease in forest wetland plantations and the increase in flooded mangroves. The barrier has also promoted new mangrove plantations via increased access to wetlands.

Maintaining the spillways of the barrier can reduce the acute flooding and thus support restoration and management to enhance the recovery of the local mangroves. It should include new spillways, revising the altitude of some of the existing spillways, and removing plants that can block the water flow through the spillways.

If wetlands and mangroves transformed by the barrier are to be restored or actively managed, a comprehensive view should be taken. Decreasing the current water level towards non-permanent flooding can cause extreme soil acidification and thus prevent the establishment of natural and cultural vegetation. Also, in the flooded mangroves, the local population has developed intensive fishing, whose importance has not yet been assessed.

Loop analysis, a qualitative mathematical modeling method, allows assessing the resilience of an environmental system. The assessment is based on the feasibility of representing the system with stable loop models. Such a representation can also support promoting the resilience of desired statuses of the environmental system, and the change of non-desired ones. Robust results can be expected if the holistic approach of qualitative methods like loop analysis is complemented with quantitative methods.

Restoration of mangrove covers should be more frequently undertaken, because the ecological knowledge needed is basically available, as this study exemplifies with study cases from Cuba, Mexico and United States of America.

A research station established in Havana province in the 1980s, relocated during the present study, should be reactivated. In this study, local data also allowed testing a recent (2003) theoretical framework on one of the longest (1933) unsolved problems in ecology, i.e., the biomass-density trajectories of self-thinning.

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11 APPENDICES

Appendix 1 Plots established for survey. Sector number and position: location of the plot with respect to the freshwater barrier as in Figure 2.1 (south=seaward, north=landward). Cover (1985) – cover (2005): characteristic land cover (in 1985 and 2005): Mang-Mang (mangrove-mangrove), Mang-FloodMang (mangrove-flooded mangrove), Herb-MangPlantat (herbaceous swamp-mangrove plantation), Herb-Mang (herbaceous swamp-mangrove), Herb-Herb (herbaceous swamp-mangrove plantation), WetForest-Herb (wetland forest-herbaceous swamp). Coordinates refer to projection UTM zone #17 north; Datum WGS-84 as in the ENVI software (Research Systems Inc. 2001)

Plot No.	Sector number and position	Cover (1985)-cover (2005)	Located in transect (yes/ no)	Coordinate east (m)	Coordinate north (m)
1	II south	Herb-MangPlantat	no	318653.2	2512153.4
2	II south	Mang-Mang	yes	319493.2	2512213.4
3	II south	Mang-Mang	yes	320033.2	2512183.4
4	II south	Mang-Mang	yes	321743.2	2512423.4
5	II south	Herb-MangPlantat	no	318226	2511311
6	II south	Herb-MangPlantat	no	318366	2511408
7	II south	Herb-MangPlantat	no	324443.2	2511763.4
8	II south	Herb-MangPlantat	no	324383.2	2511763.4
9	II south	Herb-MangPlantat	no	329783.2	2510233.4
10	II south	Herb-MangPlantat	no	329843.2	2510293.4
11	II north	Mang-FloodMang	yes	319343.2	2512483.4
12	II north	Mang-FloodMang	yes	In between plots 11 and 13; geo-referencing impossible.	
13	II north	Mang-FloodMang	yes	319403.2	2512393.4
14	II north	Mang-FloodMang	yes	319373.2	2512363.4
15	II north	Mang-FloodMang	yes	319463.2	2512303.4
16	II north	Herb-Herb	no	318653.2	2512183.4
17	II north	Herb-Herb	no	324443.2	2511823.4
18	II north	Herb-Herb	no	329753.2	2510293.4
19	II north	Mang-FloodMang	yes	320543.2	2512453.4
20	II north	Mang-FloodMang	yes	320663.2	2512513.4
21	II north	Mang-FloodMang	yes	320363.2	2512813.4
22	II north	Mang-FloodMang	yes	320423.2	2512633.4
23	II north	Mang-FloodMang	yes	320123.2	2512483.4
24	III south	Herb-MangPlantat	no	330293.2	2510563.4
25	III north	Herb-MangPlantat	no	330353.2	2510623.4
26	IV south	Herb-Mang	no	336863.2	2510293.4
27	IV south	Mang-Mang	yes	341753.2	2509663.4
28	IV north	Mang-Mang	yes	341753.2	2509693.4
29	IV south	Mang-Mang	yes	344213.2	2509813.4

Plot No.	Sector number and position	Cover (1985)-cover (2005)	Located in transect (yes/ no)	Coordinate east (m)	Coordinate north (m)
30	IV north	Mang-FloodMang	yes	344213.2	2509873.4
31	IV north	WetForest-Herb	no	348923.2	2510083.4
32	IV south	WetForest-Herb	no	348956	2509993
33	IV north	Mang-FloodMang	no	336833.2	2510323.4
34	II north	Mang-FloodMang	no	320153.2	2512843.4
35	II north	Mang-Mang	yes	320243.2	2513023.4
36	II north	Mang-Mang	yes	In between plots 35 and 37; geo-referencing impossible.	
37	II north	Mang-Mang	yes	320183.2	2512903.4
38	II south	Mang-Mang	yes	320543.2	2512003.4
39	II south	Mang-Mang	yes	320543.2	2512063.4
40	II south	Mang-Mang	yes	320543.2	2512153.4
41	II south	Mang-Mang	yes	320543.2	2512273.4
42	II south	Mang-Mang	yes	319373.2	2512213.4
43	II south	Mang-Mang	yes	319493.2	2511793.4
44	II south	Mang-Mang	yes	319463.2	2511853.4
45	II south	Mang-Mang	yes	319463.2	2511913.4
47	V north	Mang-FloodMang	yes	360683.2	2509183.4
48	V north	Mang-FloodMang	yes	360953.2	2509213.4
49	V north	Mang-Mang	yes	360893.2	2509393.4
50	V north	Mang-FloodMang	yes	360803.2	2509273.4
51	V north	Mang-FloodMang	yes	360803.2	2509273.4
52	V north	Mang-FloodMang	yes	360323.2	2509183.4
53	V south	Herb-MangPlantat	no	365753.2	2509333.4
54	V north	Herb-MangPlantat	no	365753.2	2509453.4
55	V south	Mang-Mang	yes	360953.2	2509123.4
56	V north	Herb-MangPlantat	no	350933.2	2509393.4
57	V south	WetForest-Herb	no	350933.2	2509273.4
58	V north	Herb-Herb	no	352883.2	2509333.4
59	V south	Herb-Mang	no	352883.2	2509273.4
60	V north	Herb-MangPlantat	no	353063.2	2509393.4
61	V south	Herb-MangPlantat	no	353153.2	2509393.4
62	V south	Herb-MangPlantat	no	354233.2	2509603.4
63	V north	Herb-MangPlantat	no	354233.2	2509663.4
64	V north	Mang-FloodMang	yes	364013.2	2510413.4
65	V north	Mang-FloodMang	no	364013.2	2510383.4
66	V north	Mang-FloodMang	yes	364163.2	2510323.4
67	V north	Mang-FloodMang	no	364193.2	2510143.4
68	V north	Mang-FloodMang	yes	364193.2	2510113.4
69	V north	Mang-FloodMang	yes	362873.2	2509903.4
70	V north	Mang-Mang	yes	362783.2	2509963.4

Appendix 2 Vegetation data for transect in sector II (Figure 4.5). Number of field plots in profiles: A (2 plots), B (1), C (6), D (3), E (1), F (5), G (4), H (2). For profiles with more than one field plot, the mean was calculated for each variable in the profile.

a. Vegetation cover (%) (non-mangroves refer to herbs and lianas)

Profile	Total	*R. mangle*	*A. germinans*	*L. racemosa*	*C. erectus*	Non-mangroves
A	77.5	0.0	20.0	52.5	0.0	27.3
B	90.0	90.0	0.0	2.0	0.0	1.0
C	38.0	24.2	4.2	9.2	0.2	5.8
D	0.7	0.2	0.0	0.3	0.0	0.2
E	75.0	75.0	0.0	0.0	0.0	0.0
F	77.0	14.0	64.0	0.4	0.0	1.4
G	72.5	0.4	78.5	0.0	0.0	0.1
H	92.5	92.5	0.0	0.0	0.0	0.0

b. Tree height (m)

Profile	*R. mangle* (layer 1)	*R. mangle* (layer 2)	*A. germinans*	*L. racemosa*	*C. erectus*
A	0.0	0.0	5.0	6.0	0.0
B	5.0	0.0	0.0	5.0	0.0
C	7.0	3.6	2.2	1.9	4.0
D	1.0	0.0	0.0	1.5	0.0
E	8.5	0.0	0.0	0.0	0.0
F	9.7	0.0	8.5	11.0	0.0
G	1.0	0.0	6.3	0.0	0.0
H	6.6	0.0	0.0	0.0	0.0

c. Tree diameter (DBH) (cm)

Profile	*R. mangle* (layer 1)	*R. mangle* (layer 2)	*A. germinans*	*L. racemosa*	*C. erectus*
A	0.0	0.0	5.0	5.0	0.0
B	4.4	0.0	0.0	3.0	0.0
C	9.0	6.0	2.0	2.0	2.0
D	1.0	0.0	0.0	2.0	0.0
E	18.0	0.0	0.0	0.0	0.0
F	11.5	0.0	14.4	21.0	0.0
G	1.0	0.0	9.6	0.0	0.0
H	11.8	0.0	0.0	0.0	0.0

Appendix 3 Vegetation data corresponding to the profiles represented in Figure 4.6, i.e., a transect in sector V. Number of field plots providing data for a single profile: A (1 plot), B (1), C (1), D (3), D1 (2), E (1), F (1), G (1), H (1). For profiles with more that one field plot, the mean was calculated for each variable to be represented in the profile.

a. Vegetation cover (%) (non-mangroves refer to herbs and lianas)

Profile	Total	*R. mangle*	*A. germinans*	*L. racemosa*	*C. erectus*	Non-mangroves
A	90.0	0.0	60.0	40.0	0.0	0.0
B	5.0	0.0	0.0	5.0	0.0	0.0
C	50.0	40.0	0.0	10.0	0.0	0.0
D	66.7	0.0	0.0	66.7	0.0	0.0
D1	27.5	3.5	2.5	25.0	0.0	0.0
E	80.0	25.0	20.0	25.0	5.0	0.0
F	0.0	0.0	0.0	0.0	0.0	0.0
G	75.0	60.0	20.0	30.0	0.0	2.0
H	85.0	0.0	60.0	30.0	0.0	2.0

b. Tree height (m)

Profile	*R. mangle*	*A. germinans*	*L. racemosa*	*C. erectus*
A	0.0	8.0	7.0	0.0
B	0.0	0.0	3.0	0.0
C	11.0	0.0	1.9	0.0
D	0.0	0.0	4.7	0.0
D1	1.8	2.0	3.5	0.0
E	5.5	4.8	4.3	6.0
F	0.0	0.0	0.0	0.0
G	6.5	7.0	5.8	0.0
H	0.0	8.8	6.8	0.0

c. Tree diameter (DBH) (cm)

Profile	*R. mangle*	*A. germinans*	*L. racemosa*	*C. erectus*
A	0.0	16.0	7.0	0.0
B	0.0	0.0	2.0	0.0
C	15.6	0.0	2.0	0.0
D	0.0	0.0	4.1	0.0
D1	1.5	2.0	2.0	0.0
E	5.8	5.2	4.2	4.2
F	0.0	0.0	0.0	0.0
G	7.8	6.9	4.9	0.0
H	0.0	13.9	7.2	0.0

Appendix 4 Vegetation data for two transects in sector IV (profiles 1A, 1B, 2A and 2B) (Figure 4.7). Different from sector II and sector V (whose respective data are in Appendix 2 and 3), each profile in sector IV is based in a single field plot.

a. Vegetation cover (%) (non-mangroves refer to herbs and lianas)

Profile	Total	*R. mangle*	*A. germinans*	*L. racemosa*	*C. erectus*
1A	80.0	80.0	0.0	0.5	2.0
1B	90.0	90.0	0.0	0.5	2.0
2A	80.0	35.0	45.0	2.0	0.0
2B	50.0	1.0	50.0	0.0	0.0

b. Tree height (m)

Profile	*R. mangle*	*A. germinans*	*L. racemosa*	*C. erectus*
1A	2.8	0.0	1.0	1.9
1B	2.4	0.0	1.0	2.5
2A	5.8	5.1	6.0	0.0
2B	1.0	6.1	0.0	0.0

c. Stem diameter (DBH) (cm)

Profile	*R. mangle*	*A. germinans*	*L. racemosa*	*C. erectus*
1A	3.5	0.0	1.0	4.9
1B	6.6	0.0	1.0	5.3
2A	5.8	9.2	5.7	0.0
2B	1.0	7.6	.0	0.0

Appendix 5 Plots in 2 transects outside the dammed area. Sector I = plots I1-I3; sector VI = plots VI1-VI3 (Figure 5.2). Coordinates refer to projection UTM zone #17 north; Datum WGS-84 as in the ENVI software (Research Systems Inc. 2001).

Plot No.	Sector number	Coordinate east (m)	Coordinate north (m)
I1	I	317033.2	2509453.4
I2	I	316913.2	2509483.4
I3	I	316073.2	2509333.4
VI1	VI	370043.2	2509543.4
VI2	VI	370103.2	2509633.4
VI3	VI	370013.2	2509873.4

Appendix 6 Vegetation data (percentage). Transect sector I: profiles I1, I2 and I3. Transect sector VI: profiles VI1, VI2 and VI3 (Figure 5.2). (For other variables see Appendix 7)

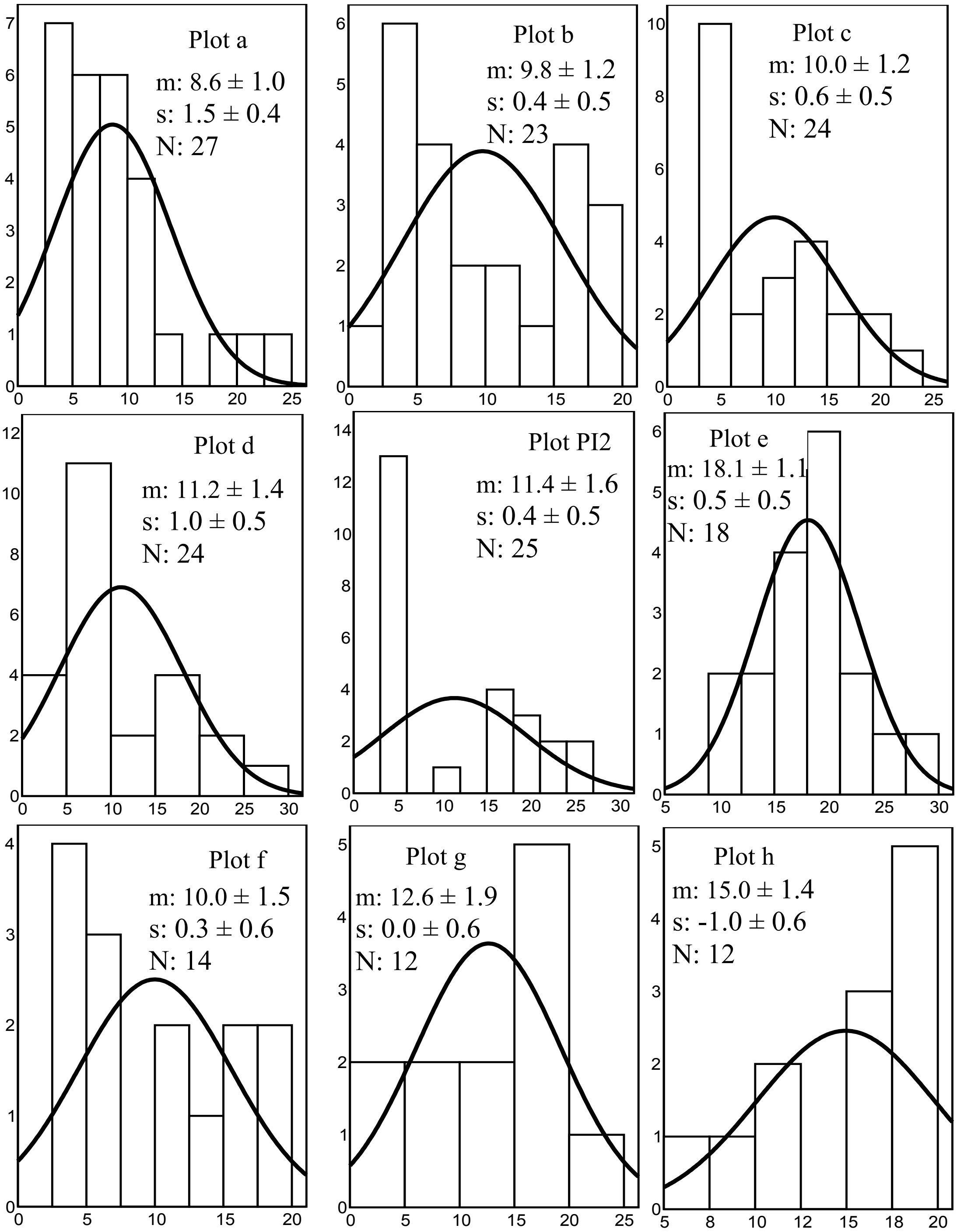
Profile	Cover	*R. mangle*	*A. germinans*	*L. racemosa*	*C. erectus*	Total
I1	General	45	45	1	0	95
	Saplings	0	0	1	0	
	Seedlings	5	10	0	0	
I2	General	40	50	2	0	80
	Saplings	2	50	1	0	
	Seedlings	1	1	0	0	
I3	General	5	10	60	0	75
	Saplings	1	0	1	0	
	Seedlings	0	1	2	0	
VI1	General	20	80	5	0	80
	Saplings	20	20	10	0	
	Seedlings	5	2	0	0	
VI2	General	50	5	40	0	70
	Saplings	0	1	0	0	
	Seedlings	2	1	1	0	
VI3	General	50	0	0	40	90
	Saplings	0	0	0	0	
	Seedlings	1	0	0	1	

Appendix 7 Vegetation data (height, stem diameter, tree density). Transect sector I: profiles I1, I2 and I3. Transect sector VI: profiles VI1, VI2 and VI3 (Figure 5.2) (For other variables see Appendix 6)

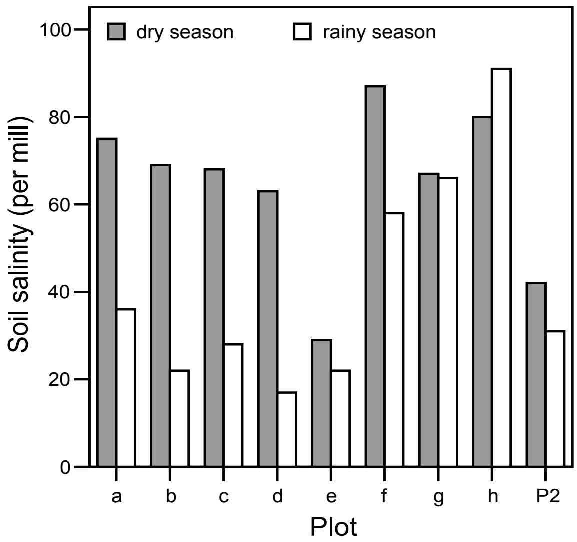
Plot	Species	Condition*	Height (m)	Stem diameter (DBH (cm))	Tree density (trees per 100 m^2)
I1	*A. germinans*	alive	12.3	25.7	6
	R. mangle	alive	7.6	8.5	26
	unidentifiable	stand. dead	5.0	4.5	3
I2	*A. germinans*	stand. dead	5.3	9.2	3
		alive	7.5	11.7	24
	R. mangle	alive	3.5	3.5	1
	unidentifiable	stand. dead	truncated	5.1	1
I3	*A. germinans*	alive	12.0	22.6	1
	L. racemosa	stand. dead	6.2	5.5	5
		alive	9.4	17.2	16
	R. mangle	alive	6.7	5.3	3
VI1	*A. germinans*	stand. dead	truncated	6.0	2
		alive	7.4	10.3	24
	L. racemosa	stand. dead	truncated	4.3	14
		alive	4.8	5.0	7
	R. mangle	alive	5.8	6.4	8
VI2	*A. germinans*	alive	8.0	9.4	2
	C. erectus	stand. dead	8.0	5.2	2
		alive	12.0	10.9	1
	L. racemosa	stand. dead	7.2	8.1	14
		alive	10.5	11.7	7
	R. mangle	alive	8.1	8.3	18
	unidentifiable	stand. dead	truncated	6.0	2
VI3	*C. erectus*	stand. dead	4.5	4.7	8
		alive	8.4	10.5	23
	L. racemosa	stand. dead	truncated	5.0	8
		alive	7.5	8.3	9
	R. mangle	stand. dead	truncated	3.6	1
		alive	7.3	7.7	23

* stand. dead = standing dead mangrove stem

Appendix 8 Histograms of tree stem diameters in plots surveyed in 2005. Order of histograms follows order in which the plots appear on the biomass trajectory (Figure 5.6). x–axis = classes of stem diameter (cm); y-axis = frequency; m = mean stem diameter ± standard error of mean; s = skewness of distribution ± standard error of skewness; N = number of stems. The normal distribution curve is included.



Appendix 9 Soil salinity in plots in mangroves in 2005 (a-h: sector II; P2: sector I)

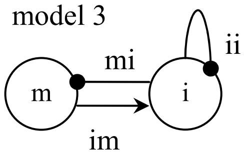


Appendix 10 Stability of a loop model in loop analysis: example assessing the stability of model 3. For the regimes observed in a wetland environment (Chapter 6), model 3 represents Regime C (Recovered canopy in conditions of increased flooding, Figure 6.5).

The two stability criteria for a loop model relate to Lyapunov stability applied to the zero solution of a system of differential equations (Puccia and Levins 1985). In loop analysis, those criteria are written as mathematical inequalities whose terms are the links of the loop model. It is based on the mathematical background of loop analysis presented in Figure 6.3: the equivalence of a loop model, a system of qualitative differential equations near a stable equilibrium point, and a square matrix.

Model 3 is stable because it fulfills the two criteria for stability as shown here:

model 3



mi: effect of i on m

im: effect of m on i

ii: self-effect of i on i

The self-effect of m on m was considered absent by the analysis which yielded this loop model (Table 6.2), so that link is not represented in the model.

The first criterion for stability addresses the levels of feedbacks present in the loop model. A feedback is a way of grouping the links represented in a loop model. The number of levels to be addressed by this first criterion equals the number of variables in the loop model. Thus, the first criterion for model 3 has to address two levels F_1 and F_2. For a loop model to fulfill the first criterion, feedbacks at all levels must be negative. So, $F_1<0$ and $F_2<0$ shall stand for model 3 to be stable. It is tested by applying the corresponding mathematical inequalities provided by loop analysis:

$F_1 = (-1)^{1+1} + (-1)^{1+1}(-ii) = -(ii) < 0; F_1<0$

$F_2 = (-1)^{1+1}(im)(-mi) + (-1)^{2+1}(mm)(-ii) = -(im)(mi)+(mm)(ii) < 0; F_2<0$

The above calculations show that both $F_1<0$ and $F_2<0$ stand, and so model 3 fulfills the first criterion for stability.

The second criterion for stability relates the negative feedback of long groups of links to the negative feedback of short groups of links. Verifying this criterion is demanded for loop models with more than 2 variables. In a 2-variable loop model as model 3 is, the fulfillment of the second criterion follows straight from the fulfillment of the first criterion as shown below.

For model 3 to fulfill the second criterion, applying the corresponding mathematical inequalities provided by loop analysis must yield $-F_1> 0$. In criterion 1 it was found $F_1< 0$; from which it follows straight that $-F_1>0$.

The above analysis indicates that model 3 fulfills the second criterion for stability.

Similar results ($F_1<0$, $F_2<0$; and $-F_1> 0$) can be obtained for models 2 and 3 (Figure 6.5), so they are also stable loop models.

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