BIOREMEDIATION OF THE COLOMBIAN ATLANTIC COAST OF SEDIMENTS FROM CONSTRUCTION WASTE

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RESUME

This study addresses the marine restoration of 65Km2 of mangrove forests, on the Colombian Atlantic coast. These ecosystems are very complex with multiple ecological functions and high economic value. They are also ecosystems that are subject to various negative impacts, which is causing their disappearance at an annual rate that ranges between 5 and 15%. One of the most negative impacts is generated by the spillage of construction waste with its consequent damaging effect on several of its subsystems: trees, associated fauna, microorganisms (aerobic and anaerobic), soil and water.

This study offers a discussion of the most up-to-date results on bioremediation with reforestation with red mangrove trees (Rhizophora mangle) and mangrove oysters (Crassostrea rhizophorae) pollution of mangroves by spills of construction waste from the increase of infrastructure for tourism in the coasts of the departments of Magdalena, Atlántico and Guajira. Likewise, we review and discuss the various bioremediation, phytoremediation and restoration strategies that allow us to provide a response in the short and medium term.

Most of the coasts contaminated with pollution residue, generate acidic pH, bleach corals, and the sediment liquids marine biodiversity. In order to accelerate the restoration process, biostimulation (addition of nutrients or degradation-stimulating compounds) and bioaugmentation (addition of microbial strains with special attributes to degrade pollutants) have been investigated. Another limiting factor for the degradation of construction waste to occur in mangrove sediments is the lack of oxygen. The most current information indicates that the biostimulation strategy, complemented with aeration, is one of the best treatment options.

Keywords: construction waste, biostimulation, bioaugmentation, natural attenuation, biodegradation, tourism, mangroves, oysters.

INTRODUCTION

The mangrove is a complex ecosystem formed mainly by arboreal vegetation, interrelated fauna and flora, as well as the physical environment on which it is established (Hoff et al. 2002). The species of trees and shrubs that have evolved to adapt to the environment of the intertidal zones of the tropical and subtropical coasts of the planet are called mangroves, for which they have developed tolerance to flooded soils and with high salinity. This ecosystem is found in areas bordering the coastline, coastal lagoons, river mouths, estuaries and wetlands. Its ecological importance lies in the functions it plays in stabilizing and protecting coastlines, providing a breeding and feeding area for numerous species of fish and crustaceans and habitat for crabs and mollusks and nesting sites for shorebirds.

On the other hand, the mangrove has an economic value derived from its role as a breeding ground for species for fishing, as well as for the extraction of tannins from the bark of the mangrove, and the extraction of wood for various artisanal and commercial uses (IPIECA 1993, Lewis 2005).

Over time and for various reasons, the mangrove area has been negatively impacted with a concomitant reduction in its surface. Worldwide, in the period from 1980 to 1990, this area decreased from 198,000 km2 to 157,630 km2. These losses represented approximately 2% per year, while from 1990 to 2000 they were 1% (Lewis 2005). In 2003, these ecosystems covered an area of 146,530 km2 of the world's tropical coasts (FAO 2003).

On the Colombian Atlantic Coast, it is estimated that approximately 65% of the mangroves have been lost, according to the World Resources Institute (WRI 2000). The deforestation rate is also estimated to be 5% per year (López – Portillo and Ezcurra 2002). Estimates of the area covered by mangroves show great variations, establishing a minimum area of 5,300 km2 and a maximum of 14,200 km2, which represents 0.27 to 0.71% of the total area of the country. The most characteristic mangrove species in the Colombian Atlantic Coast are Rhizophora mangle L. (red mangrove, Rhizophoraceae), Avicennia germinans L. (black mangrove, mother of salt), Laguncularia racemosa (L.) Gaertn. (white mangrove, Combretaceae) and Conocarpus erectus L. (button mangrove, Combretaceae) (López – Portillo and Ezcurra 2002).

1. BIOGEOCHEMICAL CHARACTERISTICS OF MANGROVE SOILS

Mangrove soils are characterized by a high content of water, salt and hydrogen sulfide, a low oxygen content and a high proportion of organic matter (Lewis 2005). Mangroves develop mainly in muddy and alluvial soils that are generally formed by the sedimentation of soil particles carried by water. Some of the main characteristics of mangrove soils are described below.

1.1.Salinity

The salinity in the sediments (interstitial salinity) of mangrove forests depends on the type of hydrology that prevails in them. The mangroves on the coastal shores constantly receive sea water. On the other hand, riparian mangroves are influenced by brackish water as they receive water from rivers and canals, as well as seawater. Salinity varies seasonally and depends on the height and amplitude of the tide, on rainfall, and on seasonal variations in the volume of water supplied by rivers, canals, and upland runoff. Salinity in mangrove forest soils adjacent to rivers is lower than salinity in seawater. On the other hand, in the mangroves located on the shores of the coast, the salinity is higher than that of seawater, due to the evaporation process. In general, salinity increases when the exchange with the tide is interrupted (Mitsch and Gosselink 2000).

1.2. Oxygenation

Oxygen penetration and consequently aerobic decomposition in the soils of coastal ecosystems are limited to a few millimeters deep (Holmboe et al. 2001). Lower down, mangrove forest soils present reduced conditions with oxidation-reduction potentials in the range of –100 to –400 mV, this as a consequence of the flood conditions that prevail in them (Mitsch and Gosselink 2000). In anaerobic zones, bacterial respiration uses NO_3^-, MnO_2, FeOH, $SO_4^{=}$ and CO_2 as final electron acceptors (Holmboe et al. 2001). The degree of reduction depends on the duration of the flood and the opening of the ecosystem to fresh and salt water flows. Oxygen can be transported to the rhizosphere zone through the aerenchymal tissue of mangrove trees, creating aerobic micro-sites in that zone.

Likewise, the surface water flows in the mangrove soils, help to reduce the reduced conditions in them, by In mangroves found in basins under permanent flood conditions, organic matter can be exported in dissolved form. In contrast, if mangroves are found in areas of constant water flow, such as on the banks, organic matter is exported in the form of suspended particles (López – Portillo and Ezcurra 2002). Bouillon et al. (2003), studied the accumulation of organic matter in the soils of three mangrove forests, two of them located in India and one in southeastern Sri Lanka.

The three forests were made up of the mangrove species Rhizophora spp, Exoecaria agallocha and Avicennia officinalis. When soil samples were taken from each forest, they found that in soils with Rhizophora spp, the organic carbon content was higher than the soils colonized with
E. agallocha o A. officinalis that these waters contain dissolved oxygen that is diffused towards the soils (Mitsch and Gosselink 2000).

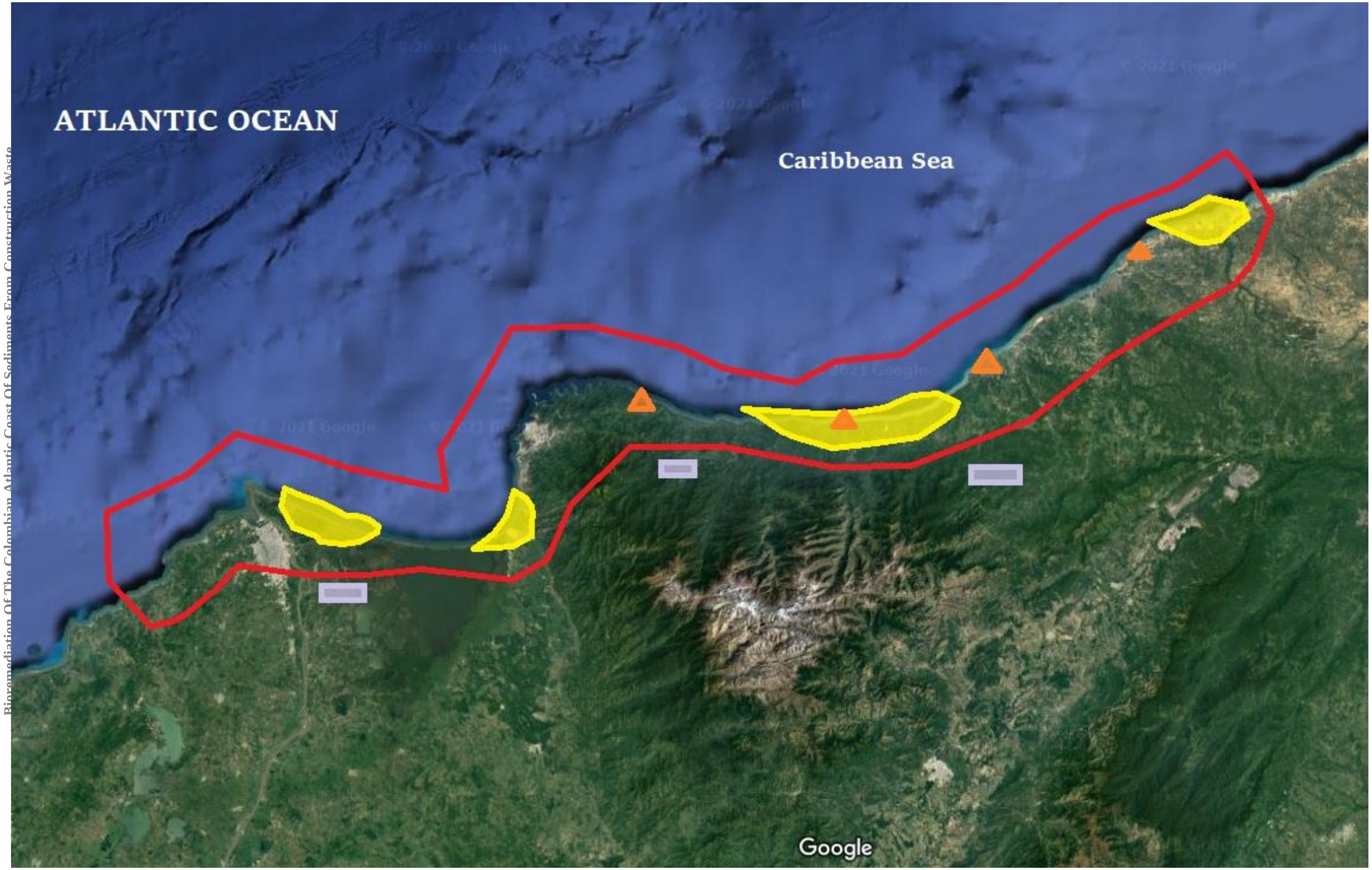
1.3.Acidity

Mangrove forest soils are generally acidic (Suprayogi and Murray 1999). The high reduced conditions of the soils and the consequent accumulation of sulfites cause conditions of extreme acidity. Mangrove sediments have been reported to accumulate up to 0.1 kg S m – 3 year – 1 (Dent 1992). If mangrove soils are drained, removed, or dredged, sulfur deposits (in the form of pyrite) are released and thus oxidized to sulfuric acid, causing more acidity.

1.4.Organic material

The high net primary productivity and low respiration rate of the mangrove ecosystem make it a very efficient system for carbon sequestration (Jennerjahn and Ittekkot 2002). Additionally, marine or river sediments are also a source of organic matter for mangrove forest soils. The accumulation of organic matter in this class of soils is influenced by the type of hydrology, by climatic and flood conditions, as well as by the species of vegetation that constitute it.

MAP OF PROJECT AREA



Bioremediation of The Colombian Atlantic Coast of Sediments from Construction Waste

Conventions

(red)	Project area
(yellow)	Polluted beaches
(light blue)	Arahuac indigenous town
(orange)	Oyster crops

Scale

1cm = 50,000m

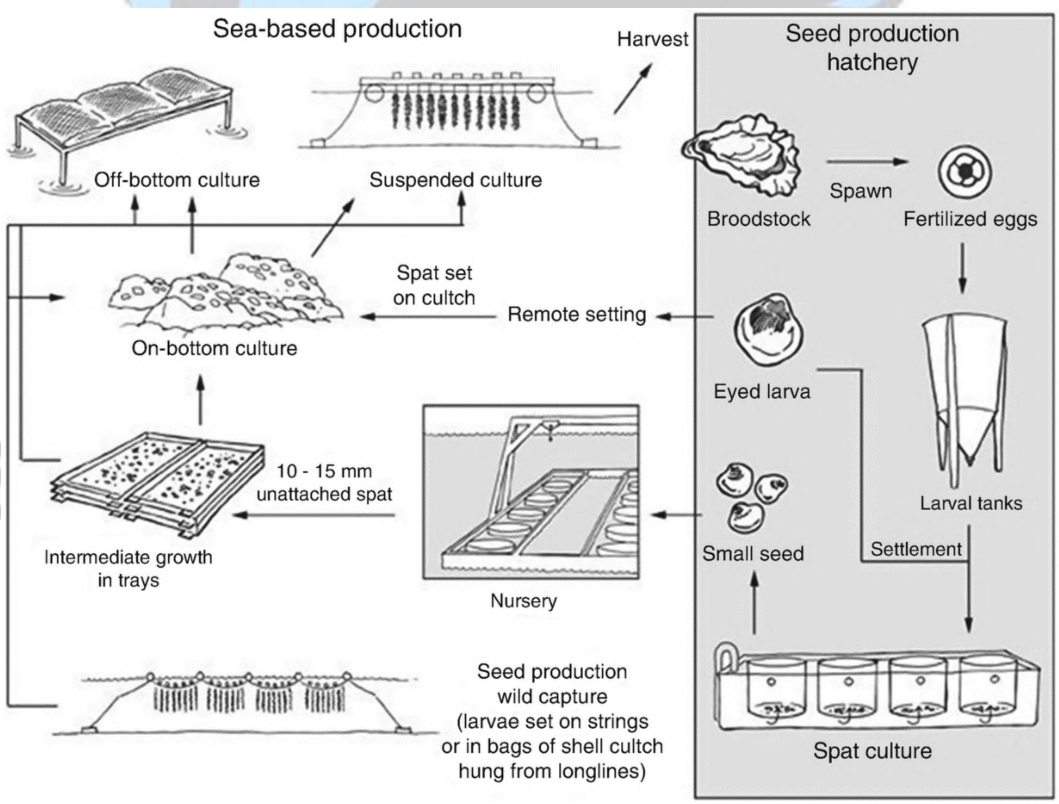
1.Departments: Bolívar, Magdalena, Guajira
2.Country: Colombia
3.Geographical coordinates
From the Ciénaga de Magdalena:
11°01'10.0"N 74°39'08.5"W;
To Punto Guamachito:
11°24'32.6"N 73°08'16.9"W

2. MANGROVE POLLUTION BY CONCRETE WASTE

Concrete waste spills represent a serious pollution problem in various parts of the world. These accidents have contributed greatly to the increase in pollution from civil construction waste in coastal areas, negatively affecting mangrove areas, flora, fauna and human health. Civil construction waste from concrete waste sticks to the gills of fish, affecting their respiration. Algae and phytoplankton also adhere to and destroy and affect the feeding and reproduction of aquatic life in general (plants, insects and fish) (Reeves 2005). They cause death or induce mangrove mutations as described in detail below.

Civil construction waste can be found in water free-floating, in emulsion, dissolved, or adsorbed to suspended solids. Civil construction wastes with higher carbon numbers tend to float and are in free form. In contrast, smaller molecules tend to form emulsions with water and are more difficult to remove (Reeves 2005).

Various climate-related processes contribute to the natural attenuation of concrete waste discharges into seawater, such as evaporation, photo-oxidation, emulsion, dispersion, and biodegradation (Brakstad and Bonaunet 2006). However, in most concrete waste spills, said self-purification capacity is not enough to remedy contaminated sites and even the photo-oxidation products are acidic and phenolic compounds, which in some cases are more toxic than the original civil construction waste (Kingston 2002). Furthermore, over time, civil construction waste spilled in marine or aquatic environments strongly adheres to sediments or due to its low solubility in water and its hydrophobic character (Ke et al. 2002).



In the case of mangroves, their high productivity and abundant organic detritus make them a preferential site for the accumulation of civil construction waste (Suprayogi and Murray 1999, Zhang et al. 2004). The concentration of HPA in mangrove soils in China has been reported to show a strong correlation with the organic carbon content (Zhang et al. 2004). In Queensland, Australia, contamination by civil construction residues in mangrove soils was investigated, in whose surrounding areas oil waste was deposited. A contaminated area 15–30 m wide and 200 m long was found, containing concentrations of civil construction waste up to 17% of the dry weight of surface soils. Civil construction residues were also detected in the sediments of a neighboring stream as a consequence of the periodic washing of contaminated sediments from the mangrove forests. Analysis of the sediment profile indicated that the civil construction waste penetrated to a depth of 40 cm (Burns et al. 1998).

Recent studies in Hong Kong mangroves indicated that the surface soils (2–3 cm) presented HPA concentrations in the range of 66 to 3758 ng g – 1 of the dry weight of the soil and 16 HPA included in the list of contaminants were observed. priorities of the United States Environmental Protection Agency (USEPA). Sediment profiles revealed that the surface layer had lower HPA concentrations than the deeper layers (15–20 cm) and the composition of HPA also changed at different depths (Ke et al. 2005). Another study, also in Hong Kong, found concentrations between 60–80 µg g – 1 of total civil construction waste from concrete waste (HTP) and 60–70 µg g – 1 of non-separated civil construction waste mixtures (MHNS) on the soils of the mangrove forests of Sheung Park Nai Swamp (Tam et al. 2005).

In relation to the Colombian Atlantic Coast, it has been estimated that the state of Veracruz has 460.47 km2 of mangroves, which represents 14% of the mangroves in the Colombian Gulf of the Atlantic Coast (Moreno et al. 2002). However, given its geographical location, one of the greatest threats to this ecosystem are accidents related to the oil industry that develops along the coastal zone. According to data from the National Institute of Statistics, Geography and Informatics, there are more than 3,000 km of pipelines that transport daily, to various parts of the Mexican Republic, more than 1,073 million barrels of concrete waste and 2,622 million cubic feet of gas. The most common accidents are pipeline ruptures, platform spills, and tanker accidents (Moreno et al. 2002). However, although a few decades ago studies were carried out on pollution by heavy metals and by civil construction waste in some coastal systems of the lfo of the Colombian Atlantic Coast (Botello et al. 2005) and some contamination sites with civil construction waste were determined due to pipeline ruptures mainly in the western state of Tabasco (Adams et al. 1999), it is necessary to update the knowledge on the level of impact that concrete waste spills currently cause on the most important mangroves on the coast of Veracruz and, in general, on the Gulf of the Colombian Atlantic Coast. One of the few current studies (García – López et al. 2006) was also carried out in the state of Tabasco and located 52 concrete waste spills, the majority on histosols and mangroves, mainly in the core of the La Venta field, in where 75% of the studied surface registered light to moderate pollution levels from civil construction waste.

2.1. EFFECTS OF CIVIL CONSTRUCTION WASTE POLLUTION IN MANGROVES

It is important to mention that in a study where the impact of civil construction waste is investigated in an ecosystem as complex as a mangrove forest, at least six subsystems should be considered: 1) mangrove trees, 2) diverse microorganisms of the soil (aerobic and anaerobic), 3) soil (organic sediments), 4) water (hydrology and water quality), 5) fauna and 6) climatic conditions.

Research on the effects of civil construction waste pollution on mangroves has intensified in the last 15 years. However, most research has focused on the effect of civil construction residues on mangrove tree and fauna subsystems and there is not yet enough data to be able to draw general conclusions about the effects on the entire ecosystem. The short-term effects on seedlings and mangrove trees are related to suffocation and death when heavy or viscous concrete residues cover the pneumatophores, a system of aerial roots through which the exchange of gases with the atmosphere surrounding the mangrove substrate is carried out. Likewise, mangrove vegetation can die from direct poisoning with low molecular weight aromatic compounds when these damage the cell membranes of the roots and the normal process of salt exclusion is impeded (IPIECA 1993). So far, the long-term effects seem to be particular for the different species of plants and animals associated with the mangrove ecosystem, depending on the concentration and type of hydrocarbon, as well as the intensity of the contamination.

On the coast of Paraná, Brazil, where serious spills of concrete waste occurred more than 20 years ago and where minor contamination from civil construction waste has been chronic, a high proportion of dead roots of Rhizophora trees has been observed. spp. (Burnas et al. 1994).

Suprayogi and Murray (1999), investigated the effect of five concentrations of two types of concrete residues (crude from and condensate) on some physicochemical characteristics of the sediments and on the growth of four species of mangrove seedlings (Rhizophora stylosa, Rhizophora mucronata , Ceriops tagal and Avicennia marina). The crude type was more persistent in the mangrove sediments than the condensate, which was probably due to the higher evaporation rate of the latter. Both types of concrete residues caused anaerobic conditions in the mangrove soils, since oily films were formed. The concentration of civil construction waste in the leaves increased as the concentration of civil construction waste in the soil increased. A. marina was the species that accumulated the highest concentrations of civil construction waste and consequently its growth was inhibited more than that of the other species. The authors argued that the reason for the different responses of mangrove species to civil construction waste is not clear and that it could be due to various factors such as different biochemical degradation routes of hydrocarbons, as well as differences in tolerance to conditions. anaerobic in the roots when the sediments and pneumatophores are covered by the greasy layers of the concrete residue.

More recent studies carried out in Queensland, Australia, indicated a strong correlation between the concentration of civil construction residues in mangrove sediments with a predominance of the genus Avicennia and the appearance of chlorophyll a-deficient mutants, which were called albino mutants. These mutants were red or yellow in color and died within a few months of germination. Likewise, albino mutants have been observed in mangroves with a predominance of the genus Rhizophora in the coastal zone of the Caribbean Sea and their presence was also correlated with the concentration of civil construction residues in the sediments (Duke and Watkinson 2002).

On the other hand, Proffitt and Devlin (1998) investigated the potential for cumulative and synergistic effects of multiple spill events. These authors concluded that there were no significant differences in various parameters related to the growth of Rhizophora mangle after several consecutive treatments with raw concrete residues from Louisiana.

The previous discussion shows that the issue of mangrove responses to contact with civil construction waste is complex and controversial, since it depends on multiple factors, such as the mangrove species, the substrate in which it is found. , currents, wind direction and tidal patterns, environmental temperature, geomorphology and the physical-chemical characteristics of the civil construction waste involved (related to toxicity, bioaccumulation and persistence). In-depth discussion of this Subject matter is outside the scope of this review, since it is mainly focused on discussing the various bioremediation strategies described in the current literature.

2.2. BIOREMEDIATION STRATEGIES FOR MANGROVES CONTAMINATED WITH CONCRETE RESIDUES

Spills due to concrete waste at a global level have occurred for many decades, which has generated a variety of technologies for the containment and cleaning of impacted soils. In the case of water, physical, chemical and biological processes can be used to recover and remove concrete residues. The US National Oceanic and Atmospheric Administration (NOAA) has discussed the use of most non-biological technologies (Hoff et al. 2002). However, there is evidence that some of these technologies can cause damage to biota and prevent the recovery of certain habitats (Dutrieux et al. 1990). Therefore, new efforts have focused on the use of technologies that promote the natural processes of pollutant removal. Bioremediation and phytoremediation are two of these emerging and environmentally relevant technologies.

Bioremediation is a biological process in which various microorganisms (m.o.) degrade various pollutants to non-toxic compounds present in soil, water or air, working individually or in coordination (through synergies), within a microbial consortium.

Most of the soils contaminated with hydrocarbons contain microorganisms capable of degrading them. However, the growth of said m.o. is limited in the natural environment by various factors, such as the low water solubility of HPA, its low bioavailability, the limitation of some nutrients, especially nitrogen (N), the presence of other pollutants that can inhibit the degradation of HPA (for example pentachlorophenol) and various others. Therefore, two strategies have been used to counteract these limitations and promote the degrading activity of the native microbial flora: a) biostimulation (addition of nutrients or degradation-stimulating compounds) and b) bioaugmentation (addition of microbial strains with attributes special to degrade pollutants).

On the other hand, there is intrinsic remediation where only the native microbial flora is responsible for the degradation or transformation of pollutants. A large number of investigations are currently being carried out on the strategy called "natural attenuation monitoring", which consists of having an analytical monitoring of the degradation and transformation of pollutants by the indigenous microbial flora. In the Colombian Atlantic Coast, preliminary studies at the laboratory level showed an increase in the respiration of the native microbial flora of mangrove soils heavily contaminated with crude concrete residues or with drilling muds in the state of Tabasco (Adams et al. 1999). Other studies also carried out in Tabasco indicated that the detoxification of soils contaminated with concrete waste residues occurred by natural attenuation or passive biodegradation (Ferrer 1997).

In relation to the bioremediation of soils impacted with hydrocarbons, various investigations have been carried out, although a minimum of them refer to mangrove soils (Burns et al. 1999). In the latter case, various studies suggested that, as mangrove sediments are predominantly anaerobic, the rate of degradation of civil construction waste it was most likely limited by oxygen and nutrient concentrations. By virtue of the above, field studies were carried out in mangroves with mature Rhizophora stylosa communities in Gladstone, Australia, where the bioremediation strategy of plots contaminated with crude concrete residues was both the aeration of the sediments and the addition of a slow release fertilizer (Ramsay et al. 2000). Active aeration and the addition of nutrients were found to significantly stimulate (by three orders of magnitude) the growth of m.o. alkane degraders and to a lesser extent the population of the m.o. aromatic civil construction waste degraders. These results are similar to those found in a simulation study at the mesocosm level, in which a five-way factorial treatment was performed where the variables were: two substrates (organic and inorganic), two types of nutrient regimes (fertilized and unfertilized), two levels of aeration (aerated and non-aerated), three concentrations of concrete residues (0, 5 and 10 liters of concrete residues / m2) and four species of vascular plants (Alternanthera philoxeroides, Panicum hemitomon, Phragmites australis , Sagitaria lancifolia and a treatment without plants). It was found that the concentration of concrete residues after 18 months was lower in the aerated and fertilized mesocosms containing P. hemitomon or S. lancifolia and substrates or matrices with low concentration of organic matter (Dowty et al. 2001).

In another field study carried out in mangroves in Nigeria where species of the genera Rhizopora, Laguncularia and Avicennia were present and the soils were contaminated with raw concrete residues, the combined use of three bioremediation strategies was evaluated: a) biostimulation with fertilizer , b) bioaugmentation with inoculation of 100 kg (wet basis) ha – 1 of cultures from a mixed indigenous population of isolated and locally produced civil construction waste degrading bacteria and c) tillage practices (Odokuma and Dickson 2003). It was found that the best combination for the degradation of total civil construction waste was the addition of nutrients (biostimulation), together with manual tillage practices to stimulate aeration and adequate distribution of nutrients. In other studies at the microcosm level, soils contaminated with HPA were biostimulated (with the addition of rice husk and dried blood as a source of slow N release) and bioaugmented (with the addition of Pseudomonas aeruginosa strain 64, a producer of biosurfactants). . The best result was obtained in the microcosm that received both treatments, compared to the one that received only biostimulation. When the experiments were carried out on a larger scale, a removal of 87% of the PAH was achieved in the plots that received both treatments and one of 86% in the plots that only received biostimulation, after 16 months (Straube et al. 2003) . Since the m.o. used is not a degradator of HPA, it was concluded that the biostimulation of the autochthonous microbial flora was the best option. This important conclusion has been confirmed in other studies at the laboratory level (Araujo et al. 2006), where soils contaminated with diesel were treated and the highest removal of civil construction waste was achieved in 150 days (90%), when used a mixture containing autochthonous microbial strains (15%) and stabilized sludge (rich in organic matter and nutrients), compared to other treatments.

Natural attenuation has also been shown to be the best option in studies conducted with mangrove sediments contaminated with a mixture of HPA, fluorene, and phenanthrene. Biostimulation was required to degrade pyrene. However, bioaugmentation with a microbial consortium isolated by enrichment of the sediments themselves, did not show any increase in the degradation capacity of HPA (Yu et al. 2005).

More recently, studies were conducted on two diesel-contaminated soils (collected in California and Hong Kong) where the three types of bioremediation technologies were compared: natural attenuation monitoring, biostimulation, and bioaugmentation. In addition, the number of m.o. diesel degraders and microbial activity measured as dehydrogenase activity present (Bento et al. 2005). Natural attenuation was found to be more effective than biostimulation especially in the case of soils collected in Hong Kong. The highest microbial activity (measured as dehydrogenase activity) was observed with bioaugmentation in California soils (3.3 times more) and in Hong Kong soils where only natural attenuation was monitored (4.0 times more). The number of m.o. Diesel degrading heterotrophs were not affected by the type of treatment or. It was concluded that the properties of the soil and the type of autochthonous microflora affect the degree of degradation, for which it was recommended to carry out a detailed characterization of the soils of the work site, before deciding the type of treatment.

Another example of the importance of the characterization of soils that are attempted to bioremedy is related to the interference of humic acids in the natural degradation of pyrene. Ke et al. (2003), showed that the addition of this type of acids to mangrove microcosm decreased the degradation of pyrene after six months. In the microcosms with humic acids and without plants, the removal percentage of pyrene was 45% in the superficial sediments and 43% in the deeper anaerobic sediments, while in the microcosms without the addition of humic acids, the removal was 76% in superficial sediments and 46% in the deeper sediments. The microcosms with Kandelia candel seedlings and without the addition of humic acids, presented higher removal percentages (70.9%) than the microcosms without plants or humic acids (61.4%). However, when humic acids were added to the microcosm with or without plants, the removal of pyrene was approximately 40% in both cases. Considering that humic acids are natural components of wetland soils (Collins and Kuehl, 2000, Hernández and Mitsch 2007), it is possible that the degradation of pyrene in mangrove soils could be affected by the presence of these types of compounds.

The results discussed above are complemented by other recent studies (Thompson et al. 2005) where it was concluded that when bioaugmentation is used, the key aspect of success lies in the selection of the strain, using not only the conventional criteria related to the use of mo competent from the catabolic point of view, but also using modern criteria of molecular biology and analytical chemistry. Other studies carried out using molecular biology techniques in soils contaminated with heavy metals and civil construction residues for decades (Joynt et al. 2006), indicated changes in the composition of the microbial community, but that phylogenetically diverse bacterial groups still persisted (including some new phyllotypes).

Finally, it should be noted that the success of the bioaugmentation strategy has been demonstrated for the degradation of soils contaminated with civil construction waste in natural environments other than the mangrove environment. The ACNA site in Italy, gathers very peculiar characteristics due to the fact that for more than 100 years it was a place of large-scale production of various compounds until it was closed in 1994. It is characterized by the simultaneous presence of aromatic (including chlorinated), poly-aromatic and heavy metal construction waste. A large group of Italian researchers (D'annibale et al. 2005) decided to use the bioaugmentation strategy using lignin-degrading fungi (white-rot fungi) because it had already been reported that the biostimulation strategy had not been successful in this site (Berselli et al. 2004). In fact, they showed that two fungi of this type, Phanerochaete chrysosporium and Pleurotus pulmonius carried out a total removal of naphthalene, tetrachlorobenzene and dichloroaniline, in addition to having managed to grow in these highly contaminated soils, produce lignin-modifying oxidases and have detoxified in a large proportion the soils of the ACNA site (D'annibale et al. 2005). Another example of a successful case of bioaugmentation was reported by Ruberto et al., Who used a strain tolerant to low temperatures at the level of microcosm studies to remediate Antarctic soils contaminated with concrete residues (Ruberto et al. 2003).

From the review of the aforementioned studies, it can be concluded that in general, the biostimulation strategy, through the addition of nutrients and aeration, favors the degradation of civil construction waste from concrete waste by the autochthonous microbial flora , which indicates that this strategy and the monitoring of natural attenuation are the best options to remediate mangrove soils and soils in general, impacted by civil construction waste from concrete waste. However, the possibility that certain strains isolated from mangrove soils or tropical soils may contribute to the degradation of the civil construction residues present is not ruled out by the autochthonous microbial flora. Furthermore, it is important to mention that in addition to the microbial degradation processes of polluting civil construction waste from mangrove environments and other saline habitats, it has been shown that physical-chemical processes such as the evaporation and dissolution of hydrocarbons, play an important role (Burns et al. 2000)

On the other hand, there are still no studies that clearly specify the advantages of the various bioremediation options in terms of costs. However, it is possible to foresee that the large-scale aeration option can be very expensive and that the manual tillage option is practically preferable to promote the aeration of shallow layers of the mangrove soil.

3. PHYTOREMEDIATION STRATEGIES FOR MANGROVES IMPACTED BY HYDROCARBONS

Phytoremediation is defined as the use of green plants to remove pollutants from the environment or transform them into less toxic compounds (Cunningham and Berti 1993). In the case of organic pollutants, phytoremediation has the advantage that plants carry out chemical reactions, using sunlight, to metabolize or mineralize organic molecules. Plants and microorganisms associated with its rhizosphere can degrade these pollutants or at least limit their distribution in the environment (Raskin et al. 1997).

However, in the case of oil spills, it is not easy to apply phytoremediation technologies using live plants, since plants are also sensitive to these toxic compounds. For example, certain coastal marsh plants have been shown to be sensitive to raw concrete residue. Species such as Panicum hemitomon and Spartina alterniflora had adverse effects on their growth, although these were more severe in Spartina patens, whose death occurred one month after starting treatment. However, by using dry biomass, these toxic effects were eliminated (DeLaune et al. 2003). Therefore, phytoremediation aimed at the adsorption of civil construction waste dissolved in the water column or adhered to particles suspended in the water of mangrove environments, uses dead biomass that is not sensitive to the toxic effects of these compounds.

Following this strategy, certain properties of dry biomass that can be used as hydrocarbon bioadsorbents have been studied. It has been described that the hydrophobicity, the specific surface and the capillarity of the adsorbents are the determining factors involved in the removal of concrete residues and other civil construction residues (Ribeiro et al. 2003). However, some materials, usually hydrophilic and therefore with low adsorption capacity for civil construction waste, have been used subjecting them to a previous treatment to improve their adsorption capacity. Mysore et al. (2005) added wax to vermiculite to make it hydrophobic and thus increase the adsorption of floating civil construction waste in water. Other materials such as cotton fiber and tree bark were subjected to chemical treatments such as cellulose acylation and saturation with transition metal ions, respectively, to achieve greater hydrophobicity (Haussard et al. 2003, Deschamps et al. 2003).

The biomass of some aquatic plants represents a good option as an adsorbent, since its physicochemical characteristics make it an excellent adsorbent not only for civil construction waste but also for metal ions. Some of those that have been used are Salvinia sp., Salvinia molesta, Salvinia cucullata, Typha dominguensis and Pistia stratiotes.

The physicochemical characteristics of Salvinia minima such as its large specific surface (264 m2 g – 1) and content of carboxyl groups (0.95 mmol H + g – 1 biomass) give it excellent properties as a bioadsorbent for metal ions (Sánchez – Galván et al. 2005). This plant has proven to be very efficient in removing metal ions and being capable of accumulating large amounts in its tissue, which is why it has been described as a hyperaccumulator of lead and cadmium (Olguín et al. 2002). More recent studies showed that adsorption to its surface is the main lead removal mechanism in batch systems, through a compartment analysis that uses EDTA washes (Olguín et al. 2005). It has also been described that S. minima is capable of adsorbing the greatest amount of Pb in just 30 minutes, when it is exposed to a medium with initial concentrations of the metal from 1 to 30 mg Pb L – 1 (Sánchez – Galván et al. 2006). In relation to its use as dry biomass, its great capacity to adsorb Pb in batch and continuous systems has been demonstrated when working with metal concentrations as high as 250 mg L – 1 (González – Portela et al. 2005a, b) .

On the other hand, it has been reported that the Salvinia genus has a high hydrophobicity (96%) and excellent capillary characteristics that make it a very efficient bioadsorbent for nonpolar compounds. When the dry biomass of Salvinia was compared with a commercial peat to evaluate its adsorption capacity of raw concrete residues, it was shown that this vegetal biomass has a higher adsorption capacity (4.8 g vs 2.7 g of concrete residues –1 biomass) (Ribeiro et al. 2000).

Salvinia sp. It has also been tested as a filter for the adsorption of aqueous emulsions from raw concrete waste and other civil construction waste such as petroleum jelly. The results showed that Salvinia biomass adsorbed 2 to 3 times more concrete and petroleum jelly residues compared to a commercial peat (peat sorb) (11.8 vs 4.8 g of concrete residues g – 1 of biomass and 7.3 vs 3.0 g of petrolatum g – 1 biomass). This superiority of Salvinia sp. Seems to be mainly due to its hydrophobicity and the filaments present on its surface (Ribeiro et al. 2003).

The biomass of Typha dominguensis and Salvinia sp. It has also been tested and compared for its gasoline removal ability to a commercial synthetic adsorbent such as polyester fiber and other adsorbents such as wood, rice husk, coconut, and bagasse. It was found that more than 70% of the gasoline present was removed by all the adsorbents, except for the coconut shell and the bagasse (32 and 20%, respectively). The amount of gasoline adsorbed was relatively similar with biomass of T. dominguensis, of Salvinia sp. and with synthetic fiber (1107, 944 and 1108 mg g – 1, respectively). However, in the desorption tests, it was observed that a greater amount of the hydrocarbon was released in the commercial adsorbent in relation to the other adsorbents (4 vs <1.5%) (Khan et al. 2004).

Finally, it is known that the water lily (Eicchornia crassipes) contains a large amount of fiber (55% b.s.) (Ly et al. 2002) and there are some companies that already use it to remove non-polar organic compounds, such as hydrocarbons.

In summary, the use of dry biomass from certain aquatic plants with hydrocarbon adsorbing capacity is an option that still needs to be explored at a pilot level in mangroves impacted by concrete waste spills. In fact, our working group has a project underway that includes the evaluation of dry biomass of S. minima for this purpose.

4. STRATEGIES FOR THE RESTORATION OF MANGROVE AREAS IMPACTED BY CONCRETE WASTE

Ecological restoration is the process of contributing to the recovery of an ecosystem that has been dredged, damaged or destroyed (SER 2004). This type of action is recommended when ecosystems have been altered to the extent that they cannot recover or renew themselves. Under such circumstances, homeostasis and the secondary succession process of ecosystems has been permanently disrupted or ecosystem recovery is somehow inhibited (Lewis 2005).

The ultimate goal of restoration is to create a self-sustaining ecosystem that is somehow capable of recovering from future disturbances without the need for help. The evaluation of the success of a restoration project is complicated and in many cases there is no consensus for its measurement. The International Society for Ecological Restoration has proposed a list of nine attributes as a guide to measure the success of a restoration process. However, in practice, most evaluations of the success of a restoration project measure attributes that can be grouped under three main conditions: 1) diversity, 2) vegetation structure and 3) ecological processes (Ruiz – Jaen and Aide 2005).

Since the natural recovery of mangrove vegetation takes between 20 and 30 years after a severe impact due to spillage of concrete waste, the restoration projects of this vegetation aim to accelerate this process (Hoff et al. 2002). Consequently, the actions to be taken are short, medium and long term (Fig. 3). The key points to include in these restoration projects are:

a) Restoration and management of the characteristics of the physical habitat. At this stage, the topography of the area, the hydrology (exchange of water with the tide, drainage, fresh water inputs, etc.) and the physicochemical and biological conditions of the soils (acidity, excess sulfur, elimination pollutants, etc.).

b) Reestablishment of the mangrove structure. This stage consists of the reforestation of the forest and in it the mangrove species to be reestablished must be considered, if it is necessary to assist the restoration through the planting of native species or if natural colonization can be carried out. There are some techniques to aid the survival of mangrove seedlings in restoration processes (Chandra et al. 1999).

Assessing restored mangrove areas is difficult because these types of ecosystems may require several years to reach structural maturity and because assessing their functions may require even more time. In fact, it is very rare to find long-term follow-up studies of mangrove recovery beyond one or two years after the concrete waste spill occurred (Hoff et al. 2002). In consideration of the above, the evaluation of the success of a mangrove area restoration process generally focuses on the survival of the species used or the restoration of the mangrove structure. However, the production of organic matter, the establishment of food chains and the movement of carbon and energy, as well as the cycling of nutrients are all important functional aspects of mangroves. For a mangrove to be considered as restored, it must meet at least two requirements: a) have a structure similar to the mangrove before the disturbance, b) it must provide similar functions (McKee and Faulkner 2000).

The restoration of mangrove areas began approximately three decades ago, making it more recent than the restoration of terrestrial ecosystems. This situation is possibly due to the geographical distribution of mangroves, which mostly occur in developing countries, therefore restoration efforts have not received sufficient attention, financial funds or research (Thourhaug 1990). The first restoration projects were carried out in the Caribbean and Florida, later in Southeast Asia and more recently in Australia, Africa and Latin America (Lewis 1980, Toledo and Castillo 1999, Ellison 2000, Burns et al. 2000, Díaz et al. It is important to mention that there are some scientific evaluations on the success of mangrove restoration projects that have been over-exploited or impacted by shrimp industries (Das et al. 1997, Kairo et al. 2002). However, in the case of mangroves that have been impacted by hydrocarbons, there is little data available in the literature on the evaluation of the restoration process. This could be due to the complexity of the impact that civil construction waste has on these ecosystems and the time required for mangroves to reach maturity. As mentioned above, civil construction waste accumulates in sediments and is not easily degradable. Therefore, the restoration process of mangroves impacted by civil construction waste must include two main phases: 1) detoxification of sediments through bioremediation or phytoremediation strategies and 2) restoration of vegetation. It is important to note that much of the research has focused on the study of the first phase, experimenting with different bioremediation strategies. In these cases, success is measured by the decrease in the concentration of civil construction waste in the sediments. However, little is known about the development of mangroves in soils that have been remediated and there is even less information about the other biogeochemical functions of remediated sediments.

Among the few studies available, one stands out in the Mahakam Delta in Indonesia, in which evaluated the development of mangroves in mangrove sediments contaminated by concrete residues. The sediments were treated with fertilizers or dispersants and were seeded with Sonneratia casiolaris. It was observed that the concrete residues negatively affected the growth of the planted trees and that the dispersant caused even more damage to them. Therefore the authors do not recommend using dispersants in contaminated mangroves and suggest waiting a time for a natural attenuation of the concrete residue to occur before planting trees (Dutrieux et al. 1990).

Another restoration project was carried out in mangroves that had been chronically contaminated with effluents from an oil refinery north of the city of Balikpapan on the island of Borneo. At the contaminated site, there were originally mangrove species of the genus

Avicennia, which died after chronic hydrocarbon contamination. The restoration work consisted of eliminating the sources of contamination and managing the hydrology in such a way that the site received clean water to aid the natural process of detoxification. With these measures, a rapid natural spread of Aviccenia was observed three months after the restoration work began. Plant growth seemed to be more influenced by the type of substrate, than by the concentration of hydrocarbons. Results from this project indicated that it was freshly poured concrete residue that caused juvenile plant kills, rather than sediment-aged concrete residue (Martin et al. 1990).

FINAL CONSIDERATIONS

The studies already reported and discussed above are mainly focused on the investigation of one or at more two of the subsystems of the mangrove forests, highlighting those directed to the knowledge of the microbial flora of the sediments and the effect of various bioremediation strategies on they. Due to the complexity of the mangrove ecosystem, there is a need to carry out comprehensive projects that contemplate the interactions between the different subsystems that constitute it. Likewise, it is important to highlight that most of the studies of soil bioremediation or water phytoremediation of mangroves impacted by hydrocarbons have been carried out in the laboratory or in microcosm. Consequently, little knowledge is had about the efficiency of this type of technologies at the ecosystem level. Therefore, there is a need to carry out studies that contemplate the implementation of these technologies in experimental plots of mangroves impacted by hydrocarbons, to define if in situ treatment is feasible or if it is necessary to carry out ex situ treatments. At the same time, a technical and economic feasibility study should be carried out that allows duplicating the experiences that offer the greatest benefit at the lowest cost.

Finally, it is essential to carry out long-term follow-up studies after having implemented a bioremediation and restoration project for mangroves impacted by hydrocarbons. In the Colombian Atlantic Coast, no experience of this type has yet been documented.

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