Assessment of the effectiveness of the regulatory regime in controlling the effects of oil pollution on Kenya's coastal and marine environment

By

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Table of Contents

List of tables v

List of figures vi

Executive summary vii

Acknowledgements viii

1.0. Chapter 1: Introduction 1

1.1. Purpose 6

1.2. Methodology 6

2.0. Chapter 2: An overview of Kenya's coastal and marine resources and their susceptibility to oil pollution 8

2.1. Mangrove communities 8

2.2. Coral reefs 14

2.3. Seagrass communities 18

2.4. Fisheries 23

2.5. Sandy substrates 27

2.6. The water column 29

3.0. Chapter 3: Kenya's policies and legislation dealing with the management of oil pollution in the coastal marine environment (an overview) 35

3.1. National Legislation 36

3.2. International Legislation 40

3.3. The Environmental Management and Co-ordination Act, 1999 43

3.4. The National Oil Spill Response Committee/Contingency Plan 45

4.0. Chapter 4: An overview of indicators of effectiveness of oil pollution management 48

4.1. Beached bird surveys 51

4.2. Tar ball monitoring........55

4.3. Aerial surveillance of oil spills........58

4.4. Satellite remote sensing........61

5.0. Chapter 5: A review of some effective regulatory regimes for oil pollution control........68

5.1. New Zealand........69

5.2. The North Sea coastal states........71

5.3. Canada........74

6.0. Chapter 6: The potential risk of oil spills and institutional capacity........79

6.1. Potential risk of oil spills........80

6.1.1. Ocean-going commercial ships........80

6.1.2. Coastal trade vessels........81

6.1.3. Petroleum refineries, power producers, and fishing boats........81

6.2. Institutional capacity........82

7.0. Chapter 7: Recommendations........86

References........90

List of tables

Table 1: Summary of oiled birds (percent of total encountered) recorded by beach survey programs in Atlantic and Pacific Oceans........53

Table 2: Total annual number of oil slicks by country observed during hours of aerial surveillance (number of observations per flight hours)........60

Table 3. Best developed indices of the effectiveness of oil pollution management, respective data collection methods and possible applicability to Kenya........66

List of figures

Figure 1: Map of Kenya showing the Exclusive Economic Zone (limit of territorial Sea not shown) and the approximate location of shipping lanes..........3

Figure 2: A flow diagram showing the sources of mandate, the ministry and the agency responsible for implementing the mandate for the control of vessel source oil pollution..........39

Figure 3: A flow diagram showing how Kenya Ports Authority (mandated by the KPA Act, Cap 391) interacts with some of the responsible agencies in the event of catastrophic oil spills in Kenya's coastal marine waters..........46

Figure 4: A flow diagram showing how marine pollution control agency would respond to an illegal oil spill following evidence from space-borne satellite data..........63

Executive Summary

The Kenya coast is part of the major tanker route that stretches along the East African coast, running from the Middle East round the Horn of Africa to other parts of the world. The major Kenyan port, Mombasa, handles a substantial number of ocean-going ships (including oil tankers) and other smaller vessels, in addition to having other installations that deal with oil such as refineries and power generators. This raises the prospect of chronic oil pollution in the coastal and marine environment, which could have negative impacts on the natural resources thereof.

The paper presents an assessment of the effectiveness of the regulatory regime in the management of marine oil pollution. It provides an overview of the natural resources, and their susceptibility to oil pollution. They include mangrove communities, coral reefs, seagrass communities, fisheries, sandy substrates and the water column, with special emphasis on the surface microlayer of the latter given its high propensity to accumulate the hydrophobic petroleum hydrocarbon pollutants. An overview of Kenya's policies/legislation, together with regional and international agreements, treaties and protocols relating to marine oil pollution management is also presented.

Some of the most commonly used indicators of effectiveness in oil pollution management are reviewed. They include surveys of beached oiled birds, monitoring of stranded tar balls along beaches, aerial surveillance and space-borne remote sensing. A comparative analysis of some effective environmental regimes is provided, having specific regard to marine oil pollution. The problem of chronic oil pollution in Kenya's coastal and marine environment is assessed from the viewpoint of the potential sources in terms of its magnitude. The capacity of Kenyan institutions with regard to marine oil pollution management is also analysed.

From the reviews and analyses, it emerges that although Kenya could be having the right policies and regulations in place to deal with the management of marine oil pollution, the complete set of instruments by which policy effectiveness can be evaluated seems to be lacking. Instruments can be defined strictly as the resources that can be used by or through the government to attain policy objectives. These include staff, administrative structures, financial means, training and awareness raising. In light of this, the paper ultimately distils some key recommendations that Kenya could adopt in an effort to enhance the effectiveness of the existing marine oil pollution regulatory regime. Thus the use of tar ball monitoring (quantitative measurement of stranded beach tar) as a relatively cost-effective indicator of the state of the coastal marine environment with regard to petroleum pollution is recommended. Other recommendations are conducting educational campaigns to vessel operators on the meaning and consequences of oil spills, and the prospect of using satellite data subject to the availability of some of the instruments such as financial means and training.

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CHAPTER 1

1.0. Introduction

Kenya is located on the east coast of Africa between Somalia and Tanzania, with an Indian Ocean coastline that extends for approximately 600 km from the Somalia border in the north at latitude 1° 41' S, to the Tanzanian border in the south at latitude 4° 40' S. The coastal and marine environment is rich in natural resources (both living and non-living) that contribute significantly to the national economy in general, and to the socio-economic well-being of coastal communities in particular. These resources include, among others, highly productive and biologically diverse ecosystems such as mangrove forests, seagrasses, fringing coral reefs, fisheries, and clean sandy beaches. Tourism, for instance, is Kenya's second most important industry in terms of foreign exchange earnings, and almost 60% of this industry is dependent on coastal tourism. Despite their immense contributions, these coastal and marine resources experience varying degrees of threats from anthropogenic activities, oil pollution being one such threat (GoK, 2002; Munga and Mwaguni, 2003).

The East African coast is a major shipping route for large petroleum and oil tankers transporting these products from the Middle East to Europe and other parts of the world through the Mozambique channel. Approximately 350 million tonnes of crude oil are transported along this route per year (ITOPF, 2003). Major shipping routes run close to the coral reefs in some parts of the coast, for instance near the port of Djibouti and port

Sudan. Oil transport poses a major pollution threat to coastal and marine ecosystems, as long-shore currents and winds in the Western Indian Ocean are instrumental in the horizontal distribution and spread of pollutants, particularly in bringing oil slicks from the open sea beyond the EEZ limit to the coastal waters (UNEP, 2003). For instance, an aerial survey revealed a long stretch of several kilometres of oil at sea along the coast of Comoros (UNEP, 1982).

The importance of shipping and port activities to the Kenyan economy cannot be over-emphasized, as the port of Mombasa, linking the country with eastern and central African countries and the rest of the world, handles over 9 million metric tonnes of goods annually (GoK, 2002). Tankers that call into the port of Mombasa deliver approximately 2 million tonnes annually of crude oil to the Kenya Petroleum Refinery, and 109,000 tonnes of other oil products. There is therefore the elevated risk of high-impact oil spills, on top of the frequent tanker operations. In addition, the Kenya coast faces onto the busy shipping lanes that run from the Middle East round the Horn of Africa en route to other parts of the world, and this raises the prospect of large oil spills reaching the coast (Obura, 2001). The Kenyan coast and the shipping lanes, the latter situated approximately 250 nautical miles offshore (Obura, 2001), are shown in figure 1.

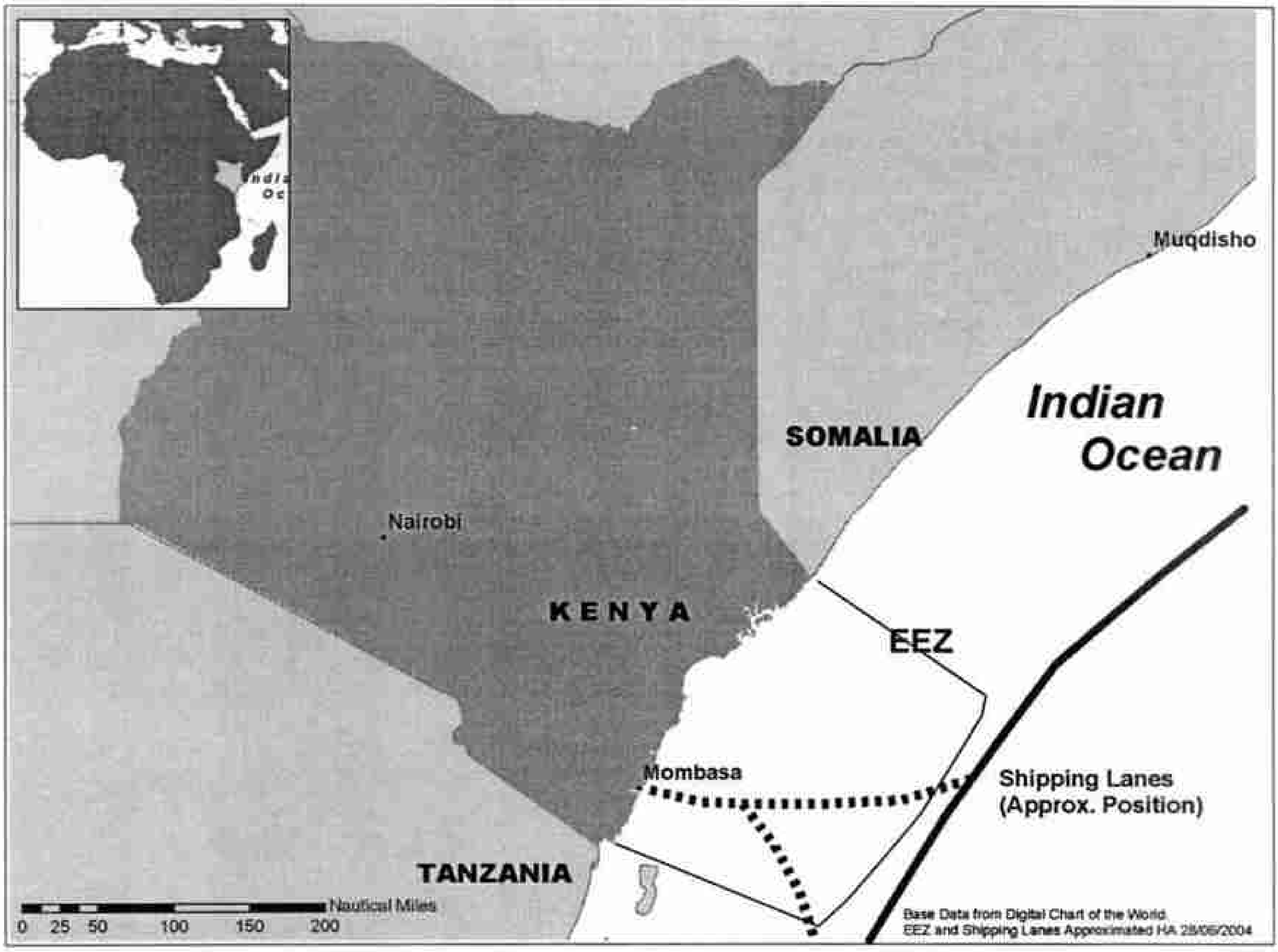

Figure 1: Map of Kenya showing the Exclusive Economic Zone (limit of territorial sea not shown) and the approximate location of the shipping lanes.

Oil pollution is a major threat to the coastal and marine environment, as it produces a variety of deleterious effects across a wide range of habitats and species. The effects are evidenced on resources such as fisheries, benthic communities, wetlands, including mangrove ecosystems, marine wildlife, including seabirds, and sandy inter-tidal habitats and beaches. Most of these resources have played pivotal socio-economic roles in many countries/regions of the world (Fingas, 1993; Peterson, 2001; Born et al., 2003; Yamamoto et al., 2003). Oil possesses a multi-pronged lethality, given that it is a complex mixture of many hydrocarbon compounds with varying physical and chemical properties such as water-solubility, toxicity, and environmental persistence. The toxicological effects of petroleum hydrocarbons on coastal and marine environments are therefore of direct, indirect, acute (short-term) and chronic (long-term) nature. The volatile low molecular weight, water-soluble fractions such as the smaller ring (1- and 2-ring) aromatic hydrocarbons are always associated with acute types of toxicity, e.g., narcosis. However, the high molecular weight components (3-, 4-, and 5-ring polycyclic aromatic hydrocarbons, PAHs) are toxic and very persistent in the environment. PAHs have been shown to produce toxic effects in a variety of marine fauna exposed to very low levels in field studies. Due to their hydrophobic nature (water-insolubility), PAHs tend to associate with particulate material and finally accumulate in the sediment. They have a high propensity to bio-accumulate in the marine food web. They therefore pose serious, long term direct and indirect effects to a wide variety of marine biota, including valuable flora such as mangroves and seagrasses (da Silva et al., 1997; Trust et al., 2000; Peterson, 2001; UNEP, 2003; Tolosa et al., 2004).

Despite Kenya's exposure to the negative effects of accidental and chronic oil pollution of it's coastal and marine environment, there has been a general lack of capacity to deal with the threat. The legal regime and institutional arrangements that can address the issue have been in existence, but formulated with a sectoral perspective. The fragmented nature of the regulatory regime has been found to have some deficiencies and shortcomings with the potential of jeopardizing its effectiveness (Munga and Mwaguni, 2003). Saharuddin (2001) showed that in Malaysia, organizational structures governing the ocean for implementing national policies were well in place, but in a fragmented and uncoordinated fashion. As a consequence, sectoral and intersectoral management problems were created, such as overlapping of jurisdiction and duplication of efforts. Kenya's other acknowledged constraint has been the minimal local operational capacity to deal with oil spills and other disasters in the territorial waters and her 200 nautical mile EEZ, as the available capacity is probably restricted to the immediate vicinity of Mombasa port (Obura, 2001; GoK, 2002).

The enactment of the Environmental Management and Co-ordination Act (GOK, 1999b) can be viewed as a solution to the ineffectiveness in environmental policies ascribable to fragmentation. However, a mere enactment of an Act does not guarantee the ultimate success of an environmental policy. For a complete assessment of the effectiveness of a regulatory regime, there are other factors that need to be considered. They include indicators of environmental health, or any other deterrent measures that are in place to ensure that environmental regulations are not being violated. A more holistic assessment of the effectiveness of a regime, say, for the control of illegal discharges of oil into the marine environment, must include these factors. Thus in situations where there are no good indicators of the state of pollution of the environment, and no measures which can deter polluters from undertaking illegal activities, the regulatory regimes are bound to be rendered ineffective.

1.1. Purpose

The purpose of the project was to assess the effectiveness of the policies and regulations Kenya has in place for the control of chronic marine oil pollution, in terms of the presence or absence of suitable indicators and deterrent measures. Appropriate recommendations are then presented based on successful practices elsewhere.

1.2. Methodology

The project was conducted mainly through a review of the literature and practice relevant to the objectives of the research. These included an overview of Kenya's policies and legislation on the management of oil pollution in the coastal marine environment. A literature review of indicators of the state of the marine environment, and hence the effectiveness of management measures, was also conducted. A few relatively effective regulatory regimes were then examined from published and grey literature, with a view to making viable recommendations for Kenya. All literature reviews were conducted from the Dalhousie University Killam library, using the sciencedirect data base. Key words used in the literature searches included oil pollution+the resource in question, e.g., oil pollution+mangroves, etc.

I also used pertinent relevant grey literature documents held by my colleagues and advisors. These included copies of the relevant Acts of the Laws of Kenya, such as the Continental Shelf Act, the Merchant Shipping Act, the Fisheries Act, the Maritime Zones

Act, Environmental Management and Co-ordination Act of 1999, and a draft of the National Marine Oil Spill Contingency Plan, among others posted to me by colleagues from Kenya. I undertook a 4 week internship at the Canadian Wildlife Service branch of Environment Canada, located at Dartmouth, Nova Scotia, one of the agencies empowered to regulate and enforce the control of oil pollution in the Canadian coastal marine environment. In the process I was exposed to the use of space-based remote sensing in the tracking of discharges of oil into the sea, and gained some insights to the operations and effectiveness of the method through discussions with Mr. Joe Pomeroy. I also held very fruitful discussions with Dr. Peter Wells about indicators of the state of the coastal marine environment with respect to petroleum pollution, which contributed to chapters 4 and 5.

CHAPTER 2

2.0. An overview of Kenya's coastal and marine resources and their susceptibility to oil pollution

The coastal and marine environment of Kenya is endowed with habitats highly productive and rich in biodiversity. They form the resource base upon which the overall national economy, and the coastal economy in particular, is highly dependent. These resources, which are of enormous ecological and socio-economic importance, include the highly productive and biologically diverse mangrove communities, fringing coral reefs, seagrass communities, fisheries and sandy intertidal habitats. The resources also form the basis for the coastal tourism industry, a significant contributor to the national economy that is primarily dependent on the infrastructure developed along sandy beaches (GoK, 2002; Munga and Mwaguni, 2003). These resources exhibit varying degrees of vulnerability and sensitivity to the negative effects of oil pollution, with the attendant potential to jeopardize the benefits derived from them. In this chapter, I describe each resource and assess its vulnerability to oil pollution from any source.

2.1 Mangrove Communities

Mangrove communities are typically tropical marine tidal forests. They occupy a narrow ecological range, constrained mostly between mean sea level and highest tidal levels between 1 and 2 m. elevation, or considerably less in some regions. The constituent trees and shrubs are a taxonomically diverse group of approximately 70 species in the world (Duke and Watkinson, 2002). They are, however, genetically diverse being derived from about 20 plant families, and they have developed common morphological and physiological adaptations in a convergent shared evolution (Duke and Watkinson, 2002). Despite restrictions such as occupation of narrow ecological range, mangroves are well adapted to their tight niche. They are, however, at risk when energy, circulation patterns or nutrient conditions change rapidly during coastal development projects and severe storms, and when additional stresses such as oil pollution are applied (Hatcher et al., 1989; Duke and Watkinson, 2002).

Mangrove forests along the Kenyan coast cover an area of approximately 53,000 hectares, with nearly 70% being found in Lamu district (Lamu-Kiunga area) in the northern part of the coast (UNEP, 1998). Eight mangrove species are found in Kenya, of the nine found along the East African coast, the dominant ones being *Rhizophora mucronata* and *Avicennia marina* (Munga and Mwaguni, 2003). The mangrove forests are highly productive and biologically diverse, providing sanctuary to a variety of terrestrial fauna which include a number of bird species, monkeys, wild pigs, hippos, buffalos, crocodiles and other reptiles (Obura, 2001; Munga and Mwaguni, 2003). Mangrove ecosystems are ecologically important coastal habitats throughout much of the tropical and subtropical world (Zuberer and Silver, 1987). They provide nursery areas and adult habitat for many harvested species of reef fish, molluscs and crustaceans such as prawns, and other fish. Other ecological functions of mangroves are nearshore nutrient trapping, concentration and recycling, thereby acting as long-term nutrient and energy reservoir; shoreline stabilization and protection, thus exerting a controlling influence against coastal erosion. Mangroves act as traps for land-derived sediment and thus reduce siltation, consequently preventing the smothering of adjacent coral reefs and seagrass communities. Human uses of mangals include recreational activities and scientific research though their unusual flora and fauna, direct uses such as provision of fuel, building materials, and a source of food and traditional medicine (Boto et al., 1989; Hatcher et al., 1989; Semesi and Howell, 1992; Hegazy, 1998; Duke and Watkinson, 2002).

Mangrove habitats exist at the interface between land and sea and as such, they are highly susceptible to exposure to oil pollution. Because they still water and trap sediments they are one of the principal places where spilled oil and associated impacts converge. There is a fairly robust literature on the effects of oil on mangrove ecosystems. The diversity and abundance of the biological communities associated with mangroves are evident with the first visit to an otherwise healthy mangrove stand after an oil spill. Oil is a complex mixture of many kinds of chemical compounds with different physical and chemical characteristics, and as such it elicits a variety of responses from mangrove communities. Mangroves suffer both lethal and sub-lethal effects from oil exposure. Oil-impacted mangroves may suffer yellowed leaves, defoliation, and eventual tree death, the latter having a long-term negative effect on mangrove communities independent of any persistent toxic effects of hydrocarbons on the organisms themselves (Garrity et al., 1994). More subtle responses include branching of pneumatophores, germination failure, decreased canopy cover, increased rate of mutation, and increased sensitivity to other stresses (Dutrieux et al.,1990; Klekowski et al., 1994).

Duke and Watkinson (2002) found a positive correlation between the concentrations of environmentally persistent fractions of oil- (polycyclic aromatic hydrocarbons-PAHs) – in sediments, and the frequency of chlorophyll-deficient mutations in *Avicennia marina* mangroves in Australia. The occurrence of the apparently lethal genetic mutation was manifest as "albino" propagules and seedlings lacking all green colouration in cotyledons and leaves, a characteristic which was expected to reduce the regenerative capacity of the local mangrove habitat. The lack of green colouration in mutant propagules implies that these individuals lack chlorophyll pigment and so are unable to photosynthesize. This was considered lethal since newly established albino seedlings appeared unable to survive more than two months. Thus the findings identified potentially serious environmental (coastal) management issues, since high mutant densities may be indicative of longer term genetic deterioration of mangrove habitats in oil-polluted wetland environments. Petroleum hydrocarbons are known to persist in mangrove sediments for decades. In a study combining chemical and biological assessment methods in a coastal mangrove ecosystem in Panama, it was found that a time period of upto 20 years or longer is required for deep mud coastal habitats to recover from the toxic impact of catastrophic oil spills (Burns et al., 1993; Burns et al., 1994).

Klekowski et al. (1994) observed a strong correlation between the frequency of chlorophyll-deficient mutations and the concentration of PAHs in the underlying sediment for mangrove populations in Puerto Rico. There was also a correlation with both acute and chronic petroleum pollution. In a mangrove swamp in Indonesia, Dutrieux et al. (1990) showed experimentally that the plants grown on oil-polluted substrate were at first submitted to a mortality phase which could be attributed to the intense toxicity of the oil. The surviving plants exhibited decreased growth rate due to the influence of the pollutant. In an experiment on the effects of oil on mangrove seedlings grown under different environmental conditions, Proffitt et al. (1995) showed that synergistic interactions between oil effects and environmental conditions may contribute to the wide range of effects reported in studies of oil contamination and mangroves. For instance, they observed that seedling mortality was greatest and growth rate lowest under hot, bright outdoor conditions. Kenya's coastal and marine environment is characterized by warm tropical conditions, with temperatures varying between 25°C and 31°C throughout the year (Obura, 2001). The potential negative effects of oil on mangrove populations in Kenya may be exacerbated by these warm conditions.

In an attempt to minimize the eventual hazardous effects of spilled oil, the major emergency response is more often than not to use chemical dispersants. The response technique reduces oil contact with mangroves and the resultant toxicity as well. Tradeoffs include potential increased toxicity to adjacent communities such as corals and seagrasses. However, field studies conducted by Burns et al. (1999) indicated that dispersant use in oil contaminated mangroves resulted in no significant difference in the amount of oil absorbed by the sediments, the penetration to depth, or the weathering patterns of the oil over time. Duke et al. (2000) showed that dispersant use had a beneficial effect in reducing the mortality of mature mangroves in a natural setting. Foliage recovery of surviving trees was also shown to be enhanced. The vulnerability of a mangrove habitat is basically the vulnerability of the trees. This finding therefore has important implications for spill responders since it offers a convenient justification for a strategy that has wider benefits, given the many functions of mangrove ecosystems.

The residence times of oil in mangrove sediments, which stretch to decades, prolongs ecosystem recovery. Another oil spill response technique commonly applied in an attempt to ameliorate the toxic effects of persistent or sediment-bound oil fractions is bioremediation. The process involves, for instance, forced aeration with supplemental application of nutrients to facilitate a faster rate of biodegradation of oil in sediments. Burns et al. (2000) conducted a chemical investigation of the use of the bioremediation protocol in oiled mangrove and salt marsh sediments. The results indicated that there was no difference in the amount of oil absorbed by the sediments, the penetration of oil to depth, or the weathering patterns of the oil over time. The biological assessment of the effectiveness of using the bioremediation protocol in mangroves conducted by Duke et al. (2000) provided mixed results. No apparent reduction in mortality of mangrove trees occurred where bioremediation was applied. However, one year after oiling, canopy leaf densities were found to be greater than controls in bioremediated plots, and less than controls in oil-only plots. The incorporation of such results into oil spill response management strategies for mangroves is a logical proposition.

It is noteworthy from the foregoing overview that the toxicity implications from oil pollution in a mangrove community are dependent on a wide variety of factors. The key variables determining the severity of effects are the amount of oil reaching the mangroves and the length of time (spilled) oil remains in the vicinity of the mangroves. For the management of oil pollution in mangrove communities, it seems obvious that prevention is the best tool for minimizing the environmental impacts, say, of a spill incident. Reducing the amount of oil reaching the mangroves not only reduces the short- and long-term toxicological effects, but also reduces impacts from clean-up strategies and the potential for chronic contamination. In a response management strategy, these considerations may imply increased protection for mangroves at risk from exposure to oil pollution, and possible use of response management measures that minimize that exposure. From the intense impacts on mangroves that have been observed, it is imperative that such strategies be accorded serious consideration.

2.2. Coral Reefs

Shallow coral reef ecosystems are predominantly confined (with few exceptions) between 30°N and 30°S of the Equator, and are arguably the oldest, and certainly the most diverse and complex ecosystems on earth. They play a crucial role in the ecology of coastal tropical waters, providing a broad range of benefits to humans, such as food, raw materials for construction, ornamental and decorative goods, medicinal and industrial chemicals, aesthetic and educational experience, among others (Elgershuizen and De Kruijf, 1976; Hatcher et al., 1989; Hodgson, 1999). However, coral reefs worldwide are under threat by an onslaught of a variety of natural and anthropogenic activities. These include global warming, overfishing and other ecosystem disturbances, sedimentation and nutrient enrichment, alien species introductions, physical disturbances e.g., from storms, and marine-based pollution such as oil spills and deliberate discharge of oily ballast water by ships (Reimer, 1975; Bryant et al., 1998).

Coral reefs, like most ecosystems, are inherently resilient to naturally occurring impacts and can recover over varying time frames. However, combined impacts from a variety of sources can jeopardize an ecosystem's ability to recover. For instance, an additional stress from chronic oil pollution or an oil spill and associated response operations in a compromised reef system, could have a greater impact than if it had occurred in a healthier, more pristine system (Hodgson, 1999).

Coral reefs along the Kenyan coast are of the fringing variety composed of reef slopes, coral flats, lagoons and reef platforms. Coral reefs cover an area of approximately 50,000 ha, and are characterized by fringing reefs extending from the Kenya/Tanzania border in the south to the Malindi area in the north (UNEP, 1998). The extent, size and diversity of coral reef ecosystems decrease northwards along the Kenyan coast, due to the un-conducive conditions arising from the influence of the Tana and Athi-Sabaki rivers and the Somali current system (Obura, 2001). Most of the information on the status of coral reefs along Kenya's coastline is based on the marine protected areas (MPAs), the system of which is considered one of the best in the Western Indian Ocean region. The MPAs act as reservoirs of biodiversity, which include a variety of faunal and floral species, and coral reefs. All the marine national parks and reserves contain coral reefs (Salm, 1996). The non-coral fauna include, among others, a variety of coral fish species and molluscs (Munga and Mwaguni, 2003).

Coral reefs have not been shown to be highly sensitive to pollution from oil slicks, but the use of dispersants and other chemical agents in oil spill response activities has been shown to be detrimental to the health of the reefs (Elgershuizen and De Kruijf, 1976; Salm, 1996). However, petroleum hydrocarbons pose a variable, but significant threat to coral reef communities. The major cause of concern is chronic oil pollution related to the routine tanker cleaning activities along oil transportation routes, production and port activities, rather than massive spills (Hatcher et al., 1989). An estimated 700 million tons of oil is transported annually from the Middle East to other parts of the world through the route traversing the Western Indian Ocean region (ITOPF, 2003). A further 6.5 million tons is shipped to countries in the region, and it has been estimated that 40% of the total oil spilled into the world oceans discharges into the Indian Ocean basin (Salm, 1996). The coral reefs along the Kenyan coast are therefore, by implication, at risk from the effects of chronic oil pollution.

Bak (1987) in studies of a Caribbean coral reef showed that sublethal effects of oil pollution on coral reefs may be manifested over very long periods of time. Quantitative surveys of reef structure, coral cover, and number of juvenile corals along 15 km of the coast showed significant variations in reef characteristics in relation to the location of a refinery and the persistent local current direction. The results of chronic oil pollution (discharges, small spills, clean-ups, etc.) were clearly discernible after a period of 60 years. Guzman and Holst (1993) evaluated sublethal effects of oil on corals 39 months after a major oil spill in Panama. There was increased number of injuries and associated reduction in colony size, and decreased size of gonads (eggs) on oiled reefs 5 years after the spill. Consequently, there was a reduction in the number of reproductively viable colonies and gametes in coral populations, and ultimately reduction in population survival. Chronic oil pollution in a tropical (Red Sea) reef was shown to cause higher rates of colony mortality, smaller numbers of breeding colonies, a decrease in the number of ovaria per polyp and a decrease in the average reproduction index (Dicks, 1984; Negri and Heyward, 2000).

Coral nubbins (the tips of coral branches) have been demonstrated to be useful for both short and long term ecotoxicity testing. Nubbins from the branching Red Sea coral *Stylophora pistillata* were used for evaluating the impacts of water soluble fractions from a crude oil, an oil dispersant and dispersed oil. Results of short and long term ecotoxicological tests revealed that the survivorship of nubbins exposed to different concentrations of crude oil was not significantly different from the controls. However, the oil dispersant was found to be highly toxic, for all nubbins exposed to 100, 10 and 1% concentration of the stock solutions died within 1 day (Shafir et al., 2003).

The effects of petroleum products, including water accommodated fractions of crude oil, production formation water and dispersant, on fertilization and larval metamorphosis of scleractinian coral Acropora millepora was investigated by Negri and Heyward (2000). Larval metamorphosis was found to be more sensitive than fertilization to crude oil. Although crude oil and dispersant inhibited larval metamorphosis and fertilization individually, the toxicity was magnified when larvae were exposed to combinations of both. This suggests some contribution to the toxicity by the dispersed hydrocarbons (i.e., toxicity may be additive). Epstein et al. (2000) also showed that coral larvae exposed to water-soluble fractions of oil, dispersed oil water accommodated fractions, and dispersants exhibited a number of sublethal effects owing to the varying toxicities. There was increased toxicity of dispersed oil as compared to untreated oil. The dispersants were found to be harmful to early life stages of corals, exhibiting high toxicity and reduced settlement rates at low concentrations. However, the oil water soluble fraction treatments were less toxic as measured by the rates of settlement and by the absence of death or morphological and behavioral alterations. The dispersed oil revealed synergistic detrimental impacts expressed as the highest mortality figures, no settlements and significant alterations in behaviour and morphology in the larvae.

The prime objective of application of dispersants is to prevent spilled oil from arriving ashore. However, this is likely to facilitate dissolution of petroleum hydrocarbons into the water column. This in turn enhances the overall toxicity and exposure of benthic organisms, particularly in the subtidal zone, resulting in cumulative, detrimental effects such as morphological abnormalities and reduced settlement rates. Management options that make use of chemical dispersants within or near coral reef habitats, or when sea conditions may drift dispersed oil into coral reefs, should therefore be avoided.

2.3. Seagrass Communities

Seagrasses are widely distributed in both tropical and temperate coastal ecosystems, forming marine meadows in coastal waters from the arctic to the tropics. They are a functional group of about 60 species of underwater marine flowering plants, creating one of the most productive aquatic ecosystems on earth (Gullstrom et al., 2002). Tropical seagrass ecosystems, like the mangals and coral reefs, exhibit high productivity and biodiversity. Seagrass beds play vital ecological and socio-economic roles. Like mangroves, tropical seagrasses provide important dwelling sites for juveniles of a variety of commercially important fish and invertebrate species such as shrimps, which move into other habitats as adults. Other species of fish that use the mangrove areas as nursery grounds use the seagrass habitat as their grow-out areas. In tropical Australia, seagrass meadows have been shown to serve as the primary nursery grounds for several species of juvenile paranoid prawns which are the basis of the major fishery in that region (Hatcher et al., 1989). Seagrasses also provide food and shelter for many animals, for instance, the endangered dugongs and turtles graze the grasses, and they are important habitat for seahorses.

Seagrasses perform important physical functions such as filtering coastal waters, dissipating wave energy and anchoring sediments. With dense root masses, they may create and stabilize near-vertical sediment walls, greatly reduce the height of storm surges, and in some instances seem hardly affected by hurricanes which severely damage adjacent mangroves and coral reefs. In addition to this stabilizing effect, seagrasses impart a chemical environment to the sediments (the rhizosphere) which allows them to support a much greater biomass of aerobic micro- and macro-fauna than areas of unconsolidated sediment. Seagrasses also impart an influence on the hydrodymanics in such a way as to encourage the settlement of the larvae of benthic animals. In some instances, they contribute significantly to local sedimentation by trapping excessive silt loads, thereby preventing coral reefs from being smothered (Hatcher et al., 1989; Munga and Mwaguni, 2003).

The Kenyan coast has an extensive coverage of seagrasses, occurring mostly in back-reef lagoons found between the beaches or cliffs and sheltered the adjacent fringing reefs. At several places along the coast, the lagoons grade into, semi-enclosed bays where mangroves, seagrass meadows and coral reefs occur as adjacent and interlinked ecosystems. The seagrass beds are thus found both intertidally and subtidally, sometimes down to about 40m. Where the supply of terrigenous sediments is limited, seagrass vegetation is also common in the creeks and channels that run through the mangroves, possibly functioning as traps and reducing the extent of the fluxes of particulate matter and nutrients between the mangroves and the ocean (Ochieng' and Erftemeijer, 2003; Gullstrom et al., 2002). Twelve seagrass species have been shown to be present in Kenya, widely distributed along the entire coastline in extensive beds that cover the largest proportion of shallow reef slopes. They often occur in mixed communities consisting of two to several of the twelve species. The most dominant species is *Thalassodendron ciliatum*, which attaches itself to both hard and soft substrates forming pure stands with high biomass (Obura, 2001; Ochieng' and Erftemeijer, 2003).

The seagrass beds in Kenya, as indeed elsewhere, have been found to possess a number of ecologically important functions. They provide habitat to a diverse array of associated marine plant and animal species. Studies on seagrass associations have revealed a good number of macroalgae and species of algal epiphytes, species of benthic invertebrates such as gastropods and bivalves, species of sea cucumbers and sea urchins. Various shrimp, lobster and crab species, and a good number of fish species including coral reef-associated herbivorous fishes, have also been identified in association with Kenya's seagrass beds. This confirms that due to the high primary productivity and the complex habitat structure of seagrass beds, they are indeed important for local and regional biodiversity conservation (McClanahan et al., 1994; Obura, 2001; Gullstrom et al. 2002; Ochieng' and Erftemeijer, 2003).

Seagrass ecosystems, like mangals and coral reefs, are subject to many threats, both natural and anthropogenic. A number of human activities have major impacts on seagrass ecosystems. Seagrasses have relatively high light requirements and, as such, are vulnerable to decreases in light penetration of coastal waters. Dredging and filling activities have been shown to be more destructive to tropical seagrass habitats than any other human activity (Hatcher et al., 1989). Direct harm to seagrass beds may occur through other activities such as boating, coastal zone development, and destructive fishing practices. Human induced climate change may impact seagrasses as sea level rises and severe storms become common-place. Living on the coastal fringe, seagrasses are under threat from oil pollution from a diverse range of sources. Not only oil can affect seagrasses, but also the chemical dispersants often used in the treatment of oil spills. The synergistic effects of oil pollution and other threats are therefore likely to result in disastrous effects on seagrasses (Hatcher and Larkum, 1982; Thorhaug et al., 1986; Hatcher et al., 1989).

Oil pollution is one of the potential threats to the seagrasses in Kenya owing to the large fleet of oil tankers that traverse the coastal waters daily, and the chronic release of oils in harbours, and in case of accidental spillages. For instance, since the accidental spillage of about 5000 tons of fuel oil in Mombasa in 1988 adjacent to mangrove and seagrass ecosystems, there has been no recovery of either the mangroves or the seagrasses that died following the spill (Obura, 2001; Ochieng' and Erfftemeijer, 2003). Photosynthesis is an important process for the thriving of seagrasses, and hence their high light requirements. It has been observed that in most cases, oil has minimal direct effect on seagrasses in form of acute toxicity since they are mostly submerged plant communities (Durako et al., 1993; Dean et al., 1998; Price, 1998). However, use of dispersants in treatment of oil spills, and enhanced irradiation from sunlight as is common in tropical environments, have been found to worsen the negative effects of oil on seagrasses. Dispersants increase the bioavailable fraction of oil in the water column and alter the interaction of these compounds with biological membranes, while sunlight is an important consideration in assessing the impact of petrochemicals since photomodification has been shown to increase toxic impacts (Catriona et al., 2003). The solvents used in dispersants have been shown to allow the toxic surfactant to penetrate the waxy protective coating of the seagrass blade, thereby impacting on the cellular membranes and chloroplasts and so decrease the photosynthetic oxygen production (Durako et al., 1993; Ralph and Burchett, 1998; Catriona et al, 2003).

The warm and sunny conditions prevalent in tropical marine environments have the potential to exacerbate the toxicity of oil to seagrass ecosystems, especially in situations where chronic exposure to oil pollution is likely to occur as is the case with Kenya. The use of dispersants has a synergistic effect on the overall toxicity of petrochemicals, and this adds to the physiological stress caused by exposure of seagrasses to these pollutants.

2.4 Fisheries

The fisheries sector in Kenya contributes about 10% to the national GDP. It is a source of employment, recreation, and foreign exchange earnings, in addition to being a source of animal protein to the population (GoK, 2002). Marine fisheries in Kenya account for only about 5% of the volume of fish landings, the large majority being from freshwater fisheries (inland lakes and rivers), predominantly in Lake Victoria. Despite the minimal contribution to the overall fisheries catch, a good proportion of the human population in coastal communities depends on marine fisheries for food security, employment and socio-economic stability. Into the bargain, the price of marine fish has been shown to be about four times that of freshwater fish on weight-to-weight basis, a factor attributed to coastal tourism (beach hotel market). The exploitation of pelagic fishes such as tuna and swordfish, even though unreported in national statistics and not fully exploited, is showing a good potential for marine fisheries in Kenya (Obura, 2001; GoK, 2002).

Coastal and marine fishery resources worldwide, like coral reefs and mangroves, are under threat from a variety of both natural and anthropogenic factors. The threats are ascribable to, among other factors, overfishing, destructive fishing methods that lead to serious ecological effects, aquaculture practices, and pollution of the coastal and marine environment from a number of sources, for instance oil pollution. Global marine fisheries landings are declining by about 500,000 metric tons per year from a peak of 80 to 85 million tons in the 1980s, a factor attributed to overfishing and habitat destruction (Pauly et al., 2003). Industrialized fishing has led to declines of large predators in coastal regions throughout the global ocean, with potentially serious ecological consequences (Myers and Worm, 2003). Global production of farmed fish (aquaculture), especially of carnivorous fish species, reduce wild fish supplies through large inputs of the latter for feed. In addition, some aquaculture systems reduce wild fish supplies through habitat modification, wild seedstock collection, and other ecological impacts such as exotic species introduction, interference with food web interactions, and effluent discharge with the potential for the spread of pathogens (Naylor et al., 2000). Pollution contributes to coastal and marine habitat degradation in general. Being one of the most important parts of the marine food chains, the ultimate effects of all sorts of coastal and marine pollution are seen in fish. Pollution from different sources and subsequent impacts on commercial use of ecosystems has been reported from many parts of the world including the major fishing areas (Islam and Tanaka, 2004).

Oil pollution and its impacts on coastal and marine ecosystems, including their fisheries, has been receiving increasing attention over time with the increase in fossil fuel use, maritime commerce, oil tanker operations and marine tanker collisions and accidents resulting in oil spills. There is enough evidence that oil pollution has serious, adverse effects on aquatic ecosystems and their organisms extending from the primary producer level through secondary, tertiary and up to top levels (Islam and Tanaka, 2004). Exposure to petroleum and its components can potentially damage fishery resources in numerous ways, including reducing the reproductive rates of fish stocks. Several studies have documented oil-related declines in reproductive parameters in a number of fish species. The types of effects observed include alterations in levels of reproductive hormones, inhibited gonadal development, and reduced egg and larval viability. Thomas and Bundiantara (1995) showed that chronic exposure of female Atlantic croaker to water-soluble fractions (WSF) of diesel fuel oil and naphthalene resulted in the blocking of sexual maturation in some fish and impaired ovarian recrudescence in others. These effects were observed after exposing the fish to pollutant levels as low as 2.5 or 5% of WSF, and 0.5 or 1.0 ppm of naphthalene (the polycyclic aromatic hydrocarbons – PAHs).

Field studies following the Exxon Valdez oil spill (EVOS) have indicated the oil exposure in a number of marine teleost species, and injury to pink salmon due to chronic exposure to low concentrations of pollutants. Fish (female dolly warden, yellowfin sole, and Pollock) collected from oil-impacted areas had high concentrations of metabolites of naphthalene and phenanthrene in bile, PAHs associated with some degree of carcinogenicity (Sol et al., 2000). Impacts to pink salmon (Oncorhynchus gorbuscha) following long-term exposure to low concentrations of weathered crude oil included damage to eggs and larvae, depression of growth rate among migrating fry, and population reduction via size-dependent mortality. Elevated egg mortality in oiled streams continued for at least 4 years after the spill, and the studies further verified that embryos are sensitive to long-term exposure to weathered oil in the low parts per billion (ppb) range (Rice et al., 2001).

Oil pollution following the 1991 Gulf war had devastating effects on the Saudi Arabian prawn stocks and the fishery they support. Prawn landings fell markedly, with a very low proportion of sexually mature adults. Post-spill cohort abundance index (tons/boat/year) for adults and juveniles was about 1.5 units relative to a pre-spill value of 195-205 units for 1989-1990 respectively. The spawning biomass index fell to about 1.8% of the pre-war level, a decline said to be capable of causing a recruitment collapse. Landings fell from nearly 4000 tons in 1989 to about 25 tons in the 1st half of 1992, occasioning the suspension of the prawn fishery (Mathews et al., 1993). Elevated concentrations of PAHs were detected in the edible tissue, and their metabolites in bile, of fish sampled from the most oil impacted areas following the Gulf war oil spill (Al-Yakoob et al., 1993; Krahn et al., 1993). Studies on the impacts of oil on the fisheries of enclosed regions of inlets of the Arabian Sea exposed to chronic oil pollution due to tanker operations revealed both lethal and sublethal effects. A wide variety of fish and shrimp fry exhibited high mortalities, attributed to the toxicity of the low boiling saturated hydrocarbon fractions of crude oil. The worst affected ones were the larvae that had deformed bodies and abnormal flexures of the tail and so were unable to swim normally (Ramamurthy, 1991).

The impact of oil pollution on coastal and marine fisheries is a major concern, for it is likely to exacerbate and pose significant ecological risk in the near future, especially in developing countries where aquatic pollution is bound to worsen as they try to industrialize. Most of the world's important fisheries have been damaged to varying extents, and situations are even more critical in those fisheries that are already overexploited or otherwise vulnerable and, therefore, deserve immediate attention. Knowledge of the impacts of oil pollution to fisheries is important not only for a better understanding of the ecosystem responses to the pollutants, but more so for formulation of effective and sustainable management of the resources for broader interest of mankind.

2.5. Sandy substrates

Sandy intertidal, subtidal habitats and beaches along the coasts are important for biodiversity conservation for they support large meiofauna and other benthic communities. Organisms such as nematodes, harpacticoid copepods, oligochaetes, and other important natural resources are all common to these habitats (Ansari and Ingole, 2002). In addition to their contribution to biodiversity, sandy beaches critical for tourism development due to their exceptional aesthetic value (Eid and Fawzi, 1991). Another important natural function of exposed sandy beaches is the filtration of large volumes of sea water, which are flushed through the interstitial pore system by tidal and wave actions. In the process, organic material is mineralized by the interstitial fauna and nutrients returned to the sea (McLachlan and Harty, 1981).

Along the Kenyan coast, sandy subtidal habitats dominate a good portion of the shoreline, influenced by the two major rivers Tana and Sabaki, from Malindi to Lamu in the north. The rivers supply a lot of sediment to the coastline in this area, some of which also feeds a system of terrestrial dunes that reach to about 50 m above sea level. The soft bottom communities support significant shrimp and bottom-fish populations which are the target of an active trawling industry (Obura, 2001). Most of the infrastructure on which coastal tourism, which accounts for nearly 60% of this important industry, is based is developed along the coastline with clean sandy beaches (GoK, 2002).

Despite the important functions associated with the sandy beaches and intertidal/subtidal habitats, they are quite vulnerable to oil pollution in our coastal environments. Routine discharges from tankers carrying oil can contribute to chronic levels of oil pollution, occasionally severe enough to form tar balls and other residues on beaches and other areas. Significant spills can also become entrained in sediments, poisoning benthic life and, by extension, other marine wildlife that forage for food in sandy substrates. Littering of sandy beaches with tar balls is aesthetically distasteful to tourism (UNEP, 1982; Da Silva et al., 1997; Tolosa et al., 2004). Tar ball pollution had been observed on tourist beaches in Kenya (UNEP, 1982). Although the situation has not reached an alarming stage as yet, there is need for effective preventive measures given the predisposition of the Kenyan coast to chronic oil pollution from both maritime and land-based sources. Hennig and Fricke (1980) showed that although beaches recover from effects of accidental oil spills after some time, they tend to deteriorate due to the arrival of unaccountable oil batches responsible for chronic oil pollution on the coast.

Studies on the ecological effects of oil pollution on meiofauna communities of sandy beaches tend to suggest that, in the case of spill incidents, the effects are strong but in most instances of a short term nature (Ansari and Ingole, 2002). A spill of heavy marine diesel oil almost totally destroyed the meiofauna of littoral sandy beaches in Picnic Bay, Hong Kong within four days of pollution. However, a few nematodes reappeared within one month and harpacticoids within 8 months (Wormald, 1976). In two South African beaches, oil deposited in sediment was found to depress the numbers of harpacticoid copepods and nematodes, but both beaches recovered after six months (Fricke et al., 1981). The synergistic effect of dipersant use on oil toxicity to meiofauna communities was shown in studies of the meiofauna of the supralittoral zone of a sandy beach. The meiofauna was reduced 1 month after dosing but numbers had returned to normal by 5 months except in the site dosed with fresh oil mixed with dispersant (McLachlan and Harty, 1982). The long-term effects of oil on meiofauna communities after spills have not been proved conclusively (Ansari and Ingole, 2002). The case could be different in situations where the oil pollution is chronic and the effects are cumulative, given the tendency of some toxic oil fractions to persist in sediment and bioaccumulate in organisms.

2.6. The water column

The sea-surface microlayer (SML), i.e., the uppermost 1-1000 μm layer of the water column, represents the interface between the ocean and the atmosphere, where the transfer of material is controlled by complex physicochemical processes. Environmental processes are controlled by the SML, and it is known to play a key role in the global distribution of anthropogenic pollutants (Wurl and Obbard, 2004). Chemically, the SML tends to have a higher content of organic compounds such as lipids, fatty acids and proteins. Due to its unique chemical composition, this interface can serve as both a sink and a source of a range of hydrophobic organic pollutants including petroleum hydrocarbons and polycyclic aromatic hydrocarbons (PAHs) (Hardy et al., 1987; Wurl and Obbard, 2004). These pollutants can be enriched in the SML by upto several orders of magnitude relative to the concentrations in the subsurface water (Cross et al., 1987). An enrichment factor of 500 has been shown to occur in some instances (Wurl and Obbard, 2004), while Kucklick and Bidleman (1994) showed that PAHs concentrated in the microlayer 18 times relative to the subsurface water samples in Winyah bay, South Carolina.

The SML is a unique ecosystem, serving as an important micro-habitat for a vast diversity of surface-dwelling organisms (zooneuston), and the developmental stages (eggs and larvae) of many fish and invertebrates (Hardy et al., 1987). Exposure of the biota in the SML to residual pollutants has important implications with respect to the ecology of the wider marine environment. The accumulation of organic pollutants such as petroleum hydrocarbons and PAHs in the SML leads to ecotoxicological impacts to the neustonic community, including mortality, developmental abnormalities, depression of growth rates and prolonged hatch time of fish eggs (Wurl and Obbard, 2004). Hardy et al. (1987) conducted studies to determine the extent to which the sea surface of Puget Sound was toxic to the early life history stages of fish. Their results showed chromosomal aberrations in developing sole embryos, reduced hatching success of sole larvae, and reduced growth in trout cell cultures. Cross et al. (1987) also showed that there was significant induction of developmental abnormalities and chromosome aberrations in fish embryos due to toxicity of sea-surface microlayer samples from Los Angeles and Long Beach harbours in California.

In general, high concentrations of PAHs in the SML have been found at sampling locations associated with anthropogenic coastal activities, particularly shipping harbours. Cincinelli et al. (2001) showed that signals of pollution from chemicals of petrogenic origin were evident in SML samples collected in the harbour of Leghorn, Tyrrhenian Sea (Italy), and at various distances from the port. The SML was found to be enriched with n-alkanes, PAHs (constituents of crude and refined oil) and other organic pollutants with respect to subsurface waters. Kucklick and Bidleman (1994) showed that PAHs were more concentrated in the microlayer than in subsurface samples in Winyah bay, South Carolina, while Hardy et al. (1990) showed that concentrations of alkanes and aromatic hydrocarbons (including PAHs) were higher in the SML relative to bulk water samples in Chesapeake bay. Suspected sources of these surface contaminants included gasoline and diesel fuel combustion, coal combustion, and petroleum products releases.

Cross et al. (1987) showed that organic contaminant concentrations (including PAHs) were several orders of magnitude higher in microlayer samples from the highly industrialized Los Angeles and Long Beach harbours in California, compared to samples from a site 15km offshore. A decline of total PAHs concentration of 333 ng/l (EF = 198) in the harbour of Leghorn, Italy to 0.214 ng/l (EF = 3.4) at sampling station 33 km offshore was reported by Cincinelli et al. (2001). The total concentration of PAHs in the SML has been shown to exhibit a general increase with the size of the port and intensity of shipping traffic, as the discharge of waste water from shipping combined with limited water exchange in a harbour has been shown to be the main cause of PAH contamination (Wurl and Obbard, 2004). Sampling and analysis of the surface microlayer in effect provides a sensitive tool for source identification and monitoring of potentially harmful aquatic pollution (Hardy et al., 1990).

Seabirds are highly vulnerable to oil slicks/spills because oil forms a thin layer on the ocean surface (SML) where many birds spend their time. The hydrophobic nature of oil causes plumage to readily absorb the oil, clogging the barbs and barbules that make them waterproof. This decreases the birds' insulation, waterproofing and buoyancy, leading to death due to hypothermia (abnormally low body temperature) caused by impaired thermoregulation or starvation (Clark et al., 1997; Wiese and Ryan, 2003). The many toxic compounds in the oil, when ingested or inhaled, can lead to debilitating or fatal effects due to their impact on internal organs such as kidneys and lungs (Wiese and Ryan, 2003).

The toxicological and other negative effects of oil components such as PAHs extend to other aquatic organisms in the water column below the SML. Exposure of an estuarine planktonic copepod *Eurytemora affinis* to the water soluble fractions (WSF) of oil was shown to cause significant reduction in life span, total number of eggs produced, mean brood size, and rate of egg production (Berdugo et al., 1977). Short-term (6 h) exposure of Atlantic salmon (*Salmo salar*) to WSF of crude oil (composed mainly of low-boiling aromatics) was found to cause tainting (change in flavour). Flavours of experimental fish samples were significantly different from those of control samples, even at the lowest exposure concentration of 0.45 ppm WSF (Heras et al., 1992). High levels of PAHs have been determined in marine mammals in some oil-polluted regions. Hellou et al. (1990) found that muscle tissues of four species of seals and six species of whales from waters around Newfoundland and Labrador were contaminated with PAHs. Samples of muscle tissue from harbour porpoises (*Phocoena phocoena*) from UK waters were also shown to be contaminated with 2-4 ring PAHs (Law and Whinnett, 1992).

Overall, this overview indicates that there is a high potential for harm to the coastal and marine habitats of Kenya from oil pollution. Chronic oil pollution from vessel and harbour operations, and accidental spillages from vessel groundings, marine- and land-based activities (NMOSCP, 2003) potentially have both short- and long-term deleterious effects on these important natural resources. Given the climatic conditions of the region (sunny conditions and warm temperatures), the toxic effects of oil components such as PAHs are bound to be greatly enhanced. Under sunny conditions, sufficient ultraviolet (UV) radiation may penetrate the water column to elicit photoenhanced toxicity in aquatic organisms (Barron and Ka'aihue, 2001). The photoenhanced toxicity of spilled oil may pose greatest hazard to translucent planktonic organisms and larval stages that inhabit the photic zone of the water column and intertidal areas. Fish eggs and larvae that occur near the water surface are thus at risk due to potential exposure to both oil and UV.

In general, management options should handle response activities that influence the temporal and spatial extent of petroleum exposure to the coastal and marine resources in ways that minimize the potential deleterious effects.

CHAPTER 3

3.0. Kenya's policies/legislation on the management of oil pollution in the coastal marine environment (an overview)

Kenya has a variety of old legislation governing an array of environmental issues, dealing with the terrestrial environment in some detail but virtually ignoring marine ecosystems. Environmental legislation, most of which established several decades ago, has been spread amongst about 77 legal statutes, only some of which apply to the coastal and marine environment. Despite this volume of legislation relating to the management and conservation of the environment, there was no explicit reference to the word "environment" in the constitution of 1997 (Obura, 2001). Furthermore, the implementation of these laws has been hampered by lack of sufficient financial resources and institutional capacity. The sectoral nature of the laws has also made coordination and implementation difficult. To address this shortcoming, the Environmental Management and Coordination Act, 1999 (GoK, 1999b) has been enacted, and was passed in January 2000. This is an umbrella law established to achieve the harmonization of implementation of the environmental management laws. The act represents a significant step forward in dealing with environmental issues in general (Mutiso, 2001; Obura, 2001).

3.1. National Legislation

Regulations on pollution and its control are spread across several Acts, with different enforcing agencies. However, with regard to oil pollution, the Merchant Shipping Act (Cap 389 of the Laws of Kenya) is the principal statute that provides for the control of pollution of the sea by oil from ships. The Act, implemented by the ministry of Transport and Communications in conjunction with other ministries, can be considered as the mechanism for regulating the pollution of Kenya's territorial waters arising from ship-based sources. Thus section 309 relates to the prohibition of pollution by oil from ships. The provision is intended to deal mainly with oil pollution originating from beyond the territorial waters.

The Kenya Ports Authority (KPA), under the Kenya Ports Authority Act (Cap 391 of the Laws of Kenya), has among other mandates, the responsibility for controlling oil pollution within the port and in Kenya's territorial waters. The KPA extends the efforts to areas beyond the territorial waters under the Continental Shelf Act (Cap 312 of the Laws of Kenya). The Act vests in the Government rights in respect of the management and exploitation of the natural resources of the continental shelf, situated within Kenya's territorial waters and beyond. These rights include, among others, the exploitation of marine resources, and through them, the responsibilities for the protection of these resources. KPA therefore extends the effort of controlling oil pollution beyond the territorial waters through these rights (Munga and Mwaguni, 2003).

In fulfillment of the responsibility vested upon it by the Act, the KPA, together with some other stakeholders, has set up the National Oil Spill Response Committee (NOSRC). The committee has in turn developed a National Oil Spill Response Contingency Plan (NOSRCP) for dealing with oil spill incidents in Kenyan coastal waters and to oversee oil spill surveillance activities. Membership is drawn from private companies and parastatals involved in oil refining and distribution (the refinery and oil industries); government agencies dealing with maritime activities (the shipping industry), wildlife, and scientific institutions dealing with resource and environmental conservation. These include KPA, Esso Kenya Limited, Kenya Wildlife Services (KWS), Kenya Marine and Fisheries Research Institute (KMFRI) and the Fisheries Department. The committee has an anti-pollution boat equipped with fire fighting equipment from the KPA. Two aircraft from the KWS are used for oil spill surveillance duties and during emergencies. However, the aircraft are not well equipped with communication equipment and night watch facilities (Odido, 1997).

The Fisheries Act (Cap 378 of the Laws of Kenya) is another statute which has a provision for the protection of pollution of Kenyan waters. Although the Act does not refer directly to the prevention of the aquatic environment from oil pollution, this is implied from the phrasing of the relevant paragraph. Part X of the Act provides for the prevention of pollution and preservation and conservation of the fishery waters. The Act defines marine fishery waters as the waters of the maritime zones described in the Maritime Zones Act (Cap 371 of the Laws of Kenya). Thus section 59 declares the Kenya fishery waters a pollution prevention zone for the purposes of protecting the aquatic environment and ecology. According to section 60(2), offenders are to be reported to the Director of fisheries, giving details of the pollution incident. According to section 60(4), any person violating the regulations shall be guilty of an offence and liable to a fine not exceeding twenty thousand shillings or to imprisonment for a term not exceeding two years or both. This should therefore explain the inclusion of the fisheries department as a member of the National Oil Spill Response Committee.

The Water Act (Cap 372 of the Laws of Kenya), implemented by the Ministry of Land Reclamation, Regional and Water Development, also has a provision for the prohibition of water pollution in general. The Act prohibits pollution of water resources through, say, the discharge of industrial and municipal effluents into rivers, lakes and the ocean. Although the Act makes no specific reference to any class of pollutant, by implication oil pollution can be assumed to be considered in the general provision.

The statutes that have provisions for the protection of the aquatic environment from pollution in general appear to be quite a numerous. However, only the Merchant Shipping Act and the Kenya Ports Authority Act happen to make direct reference to the control of oil pollution in Kenyan waters. The agency responsible for the implementation of the two Acts is the Kenya Ports Authority, and, in case of oil pollution emergencies, in conjunction with the other identified members of the National Oil Spill Response Committee.

Figure 2 depicts the sources of mandate, and their interrelationship with the implementing agency, at least as far as ship-sourced oil pollution control is concerned. There is no equivalent set of mandates for land-sourced oil pollution in Kenya's marine environment.

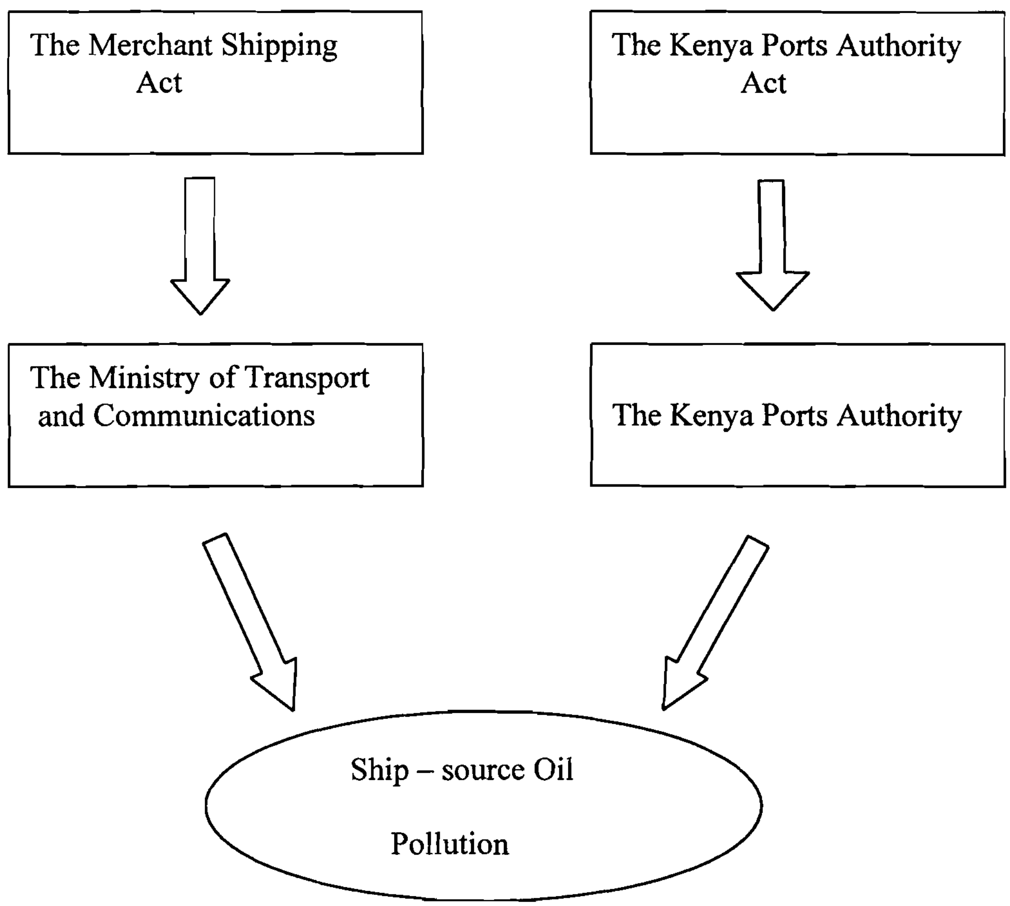

Figure 2: A flow diagram showing the sources of mandate, the ministry and the agency responsible for implementing the mandate, as far as ship-source oil pollution is concerned.

Thus as far as the statutes relating to the prohibition of pollution of Kenya's territorial waters and within the EEZ by discharge of oil from ships are concerned, the principal legislation is the Merchant Shipping Act. The Kenya Ports Authority, being responsible for the operations of the country's ports, is also mandated with the responsibility of containing oil pollution within the ports and in the outer territorial sea. The Authority, however, extends the effort beyond the territorial waters. The Ministry of Transport and Communication is the umbrella body with the general responsibility for administering the Acts, and Kenya Ports Authority falls under this Ministry. It therefore follows logically that Kenya Ports Authority is the principal government agency given the specific responsibility for implementing the mandate, and hence the structure depicted in the flow diagram above. The other legislation such as the Fisheries Act and the Water Act are general in character since they cover a wide variety of pollutants. As such they are not specifically geared to respond to the issues raised by oil pollution of the marine environment, explaining their exclusion from the flow diagram.

3.2. International Legislation

Kenya is a signatory to a number of important international environmental treaties and conventions. Although they mostly relate to pollution in general, some are specifically concerned with oil pollution of the coastal and marine environment. The country is a contracting party to the Nairobi Convention, an international convention covering Eastern Africa and the western Indian Ocean. This is an umbrella agreement that deals with the protection and management of the coastal and marine environment, and is implemented through the Regional Coordinating Unit (RCU) in the Seychelles, and the Department for Environmental Conventions (DEC) at UNEP headquarters in Nairobi (UNEP, 1999 cited in Obura, 2001). The convention lists, among the pollution sources that require control, pollution from ships, and also has a protocol concerning cooperation in combating marine pollution in cases of emergency in the East African region. With increasing unity among the member states and through a number of conferences, the Nairobi Convention stands as a strong platform for governments to commit resources to the protection of the coastal and marine environment.

The environmental provisions of the United Nations Convention on the Law of the Sea (UNCLOS) cover vessel source pollution, among others (Kenya ratified the Convention on 2nd March, 1989). The provisions apply to all ocean surfaces, not only the high seas, but also to areas under the jurisdiction of coastal states. They also seek to combat marine pollution by various preventive measures, including the duty to notify states of any imminent danger of pollution, and to develop contingency plans for responding to incidents. The provisions further seek to monitor the risks of pollution, assess potential effects of planned activities that may cause substantial pollution or significant changes in the marine environment and communicate such assessments. Thus articles 194(3)(b), 211, and 217-221 of UNCLOS outline the general framework of international norms concerning pollution of the marine environment by various forms of oil, including accidental pollution. The first article summarizes the problem, providing that the measures taken to enforce the Convention should include those designed to minimize pollution from vessels to the fullest possible extent, including accident prevention and dealing with emergencies (Kiss and Shelton, 1991; UNCLOS, 1982).

Article 211 is entirely concerned with pollution from vessels. It reaffirms the legislative objective and obligation of states to prevent, reduce, and control pollution of the marine environment. Paragraph one emphasizes the role of international rules and standards which states shall establish, "acting through the competent international organization or general diplomatic conference." States should also promote the adoption, as appropriate, of routing systems designed to minimize the threat of accidents which might cause pollution. The remaining parts of the article concern the laws and regulations that states must adopt regarding pollution, applicable to vessels registered by them or flying their flag. Among other provisions, the coastal states can adopt laws and regulations to prevent, reduce and control pollution of the marine environment by foreign vessels within their territorial sea and their exclusive economic zone. However, for the exclusive economic zone these norms shall conform to and give effect to generally accepted international rules and standards. In instances where the international rules and standards are inadequate to meet the special circumstances in an exclusive economic zone or part of it, the coastal state which exercises jurisdiction over the zone may take special mandatory measures to prevent pollution from vessels (Kiss and Shelton, 1991; UNCLOS, 1982).

The various provisions of UNCLOS relating to jurisdiction over vessel violations of environmental norms have been supplemented by a global convention and by several conventions concerning regional seas. The general convention, the International Convention for the Prevention of Pollution by Ships (MARPOL) has as its objective "the complete elimination of intentional pollution of the marine environment by oil and other harmful substances and the minimization of accidental discharge of such substances."

The principal obligation of states parties to MARPOL Convention, contained in article 1, is to give effect to its provisions and its annexes, which would appear to be the case even without an explicit statement to that effect. However, article 4 adds that any violation of the requirements of the Convention shall be prohibited and sanctioned by legislation enacted by the authority over the ship, that is, the flag State. There are also regional sea conventions drafted under the auspices of UNEP, but they do not contain rules additional to those set out in the MARPOL agreement. Instead, they generally declare that the states parties shall take all appropriate measures conforming to international law to prevent, abate, combat and control pollution caused by ships and ensure effective implementation for the zones in question of applicable international rules relating to vessel-source pollution (Kiss and Shelton, 1991).

3.3. The Environmental Management and Co-ordination Act, 1999

Kenya's Environmental Management and Coordination Act (EMCA) of 1999 is the most comprehensive, providing for the management, protection and conservation of the environment in general. For administrative purposes under this Act, the National Environment Council (NEC) has been established (Part III, section 4). The functions of the council, as specified in section 5, include policy formulation and direction; setting national goals and objectives and determining policies and priorities for the protection of the environment; and promotion of co-operation among public departments, local authorities, the private sector, non-governmental organizations and other organizations engaged in environmental protection. Also established under the Act (Part III, section 7), is the National Environment Management Authority (NEMA), the purpose of which is to exercise general supervision and co-ordination over all matters relating to the environment, and to be the principal instrument of Government in the implementation of all policies relating to the environment. Thus the functions of the Authority as outlined in section 9 include, among others, advising the Government on regional and international environmental conventions, treaties and agreements to which Kenya should be party, and following up the implementation of such agreements. Part XI (section 124) is devoted solely to international treaties, conventions and agreements (EMCA, 1999).

The enactment of the Act (EMCA) has therefore come to be viewed as a welcome panacea to any ineffectiveness in environmental legislation which could hitherto be attributed to fragmentation. In addition to providing for the co-ordination and harmonization of implementation of environmental management laws, the Act also provides for powers to enforce legislation for the protection of the environment, and the prosecution of offenders of environmental regulations. For instance, the control of ship-source oil pollution is covered under two sources of mandate (The Merchant Shipping Act, and The Kenya Ports Authority Act), with the potential of introducing inefficiency due to the fragmented implementation. By legislation, the Act is meant to harmonize implementation in such situations. Into the bargain, co-ordination of activities such as those required for the response to oil spill contingencies is provided for in this Act.

3.4. The National Oil Spill Response Committee/Contingency Plan

Oil spills into the coastal/marine environment come from a variety of sources other than vessels. These could come from other port activities such as those of oil refineries and accidents involving oil storage tanks at the port. Spills not necessarily of catastrophic nature could also emanate from land-based activities, and from operational or accidental discharges from small vessels such as fishing boats and recreational vessels. This explains the inclusion of a diverse array of participating organizations (e.g., Kenya Ports Authority, Kenya Petroleum Refineries, Kenya Pipeline Company, National Environmental Management Authority, Kenya Wildlife Services, Ministry of Transport and Communications, Mombasa Municipal Council, Fisheries Department, etc.,) into the National Oil Spill Response Committee/Contingency Plan. The primary objective of the contingency plan is to outline the procedures to be followed in combating and controlling oil spills in the territorial waters of Kenya. This area is under the jurisdiction of the Kenya Ports Authority (KPA), and hence the emanation of the mandate for the initiative from the KPA Act. Stakeholders (agencies) whose interests would in one way or the other be affected by oil spills are therefore incorporated to provide integrated support to KPA.

. The flow chart below depicts the interaction between KPA and some of these agencies in attempting to control oil spills in Kenya's territorial waters. Represented in the chart, for simplicity, are agencies responsible for oil refining and distribution, wildlife conservation, maritime activities, and fishery resources conservation.

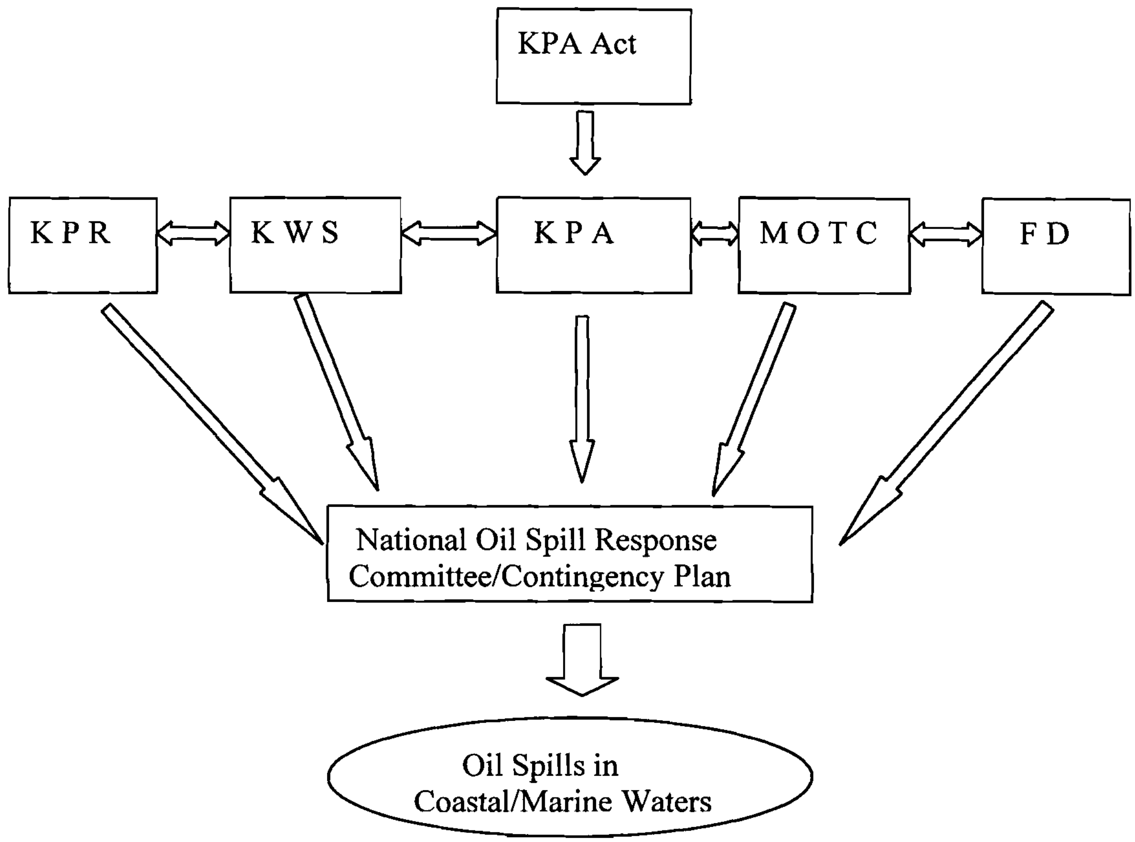

Fig. 3. Flow diagram showing how KPA (mandated by the KPA Act, Cap 391) interacts with some of the responsible agencies in the event of catastrophic oil spills in Kenyan coastal/marine waters.

Acronyms:

KPA.......Kenya Ports Authority

KPR.......Kenya Petroleum Refineries

KWS......Kenya Wildlife Services

MOTC....Ministry of Transport and Communications

FD.........Fisheries Department.

The role of the National Environmental Management Authority (NEMA), as provided for under the Environmental Management and Co-ordination Act, is to foster the co-ordination between KPA and the other participating agencies in during oil spill emergencies.

There are a number of aspects to be taken into consideration in the process of assessing the effectiveness of an environmental policy. Simply having the right regulatory regime in place is never a guarantee of the ultimate effectiveness of a policy. Environmental policy evaluations are often linked to the use of a specific instrument or set of instruments. An effectiveness evaluation is in that sense also an evaluation of those instruments in the policy context in which they are used (Gysen et al., 2002). The instrument, in the strict sense, is accompanied by other components. These include staff, administrative structures, financial means, training, awareness raising. Instruments are therefore defined as everything that is used by or through a government to attain the policy objectives. Thus the resources dedicated to the design and implementation of a measure are equally important and crucial considerations (Gysen et al., 2002).

Kenya is a party to a number of international treaties and conventions concerned with the protection of the coastal and marine environment from pollution, and seems to have in place a suitable institutional structure. However, inadequate instruments such as financial resources and operational capacity to deal with oil spills and other disasters in territorial waters and the EEZ (coastal and marine environment) have been acknowledged as major impediments to the achievement of the policy objectives (GOK, 2002).

CHAPTER 4

4.0. A review of indicators of effectiveness in oil pollution management

The deleterious effects of oil pollution in the world's coastal and marine environments confer upon all maritime nations the need to develop effective, practical and comparable monitoring strategies. The monitoring should generally be based on scientifically sound questions, and be capable of distinguishing between anthropogenic environmental damage and natural causative factors. A clear definition of monitoring goals is essential, as is the selection of suitable indicators of ecosystem status. Whereas the overall objective of monitoring environmental status/health is to identify the appropriate response to control its cause (Strain and Macdonald, 2002), in situations where the necessary measures are in place for such control the objective is solely to gauge the effectiveness of such measures.

Several quantitative indicators of the status of ecosystems have been proposed for evaluation and reporting on the status of marine ecosystems, and such indicators have roles in both communication and decision support. Thus there is a popular move towards constructing environmental indicators to reflect different aspects of the environment or of environmental policy. Areas of interest include environmental quality, the effectiveness of management strategies, and environmental commitment (i.e. the resources made available to managing the environment). The term indicator is used in several ways. It can imply a conceptual framework, a measurement or index, or an assessment statistic. However, to determine the best practices for selecting indicators for specific uses, a fairly good understanding of the information content of the various indicators is necessary (Nicholson and Fryer, 2002; Rice, 2003).

As the precautionary approach becomes broadly used as the basis for management decision-making, the role of indicators of ecosystem status becomes central. Thus it is necessary to identify values of an ecosystem indicator associated with adverse effects on the environment. The precautionary principle recognizes, among others, the fact that monitoring and research to determine the consequences of activities are essential for any management action to be taken (Ellis, 2003; Rice, 2003). In actual practice, however, many management actions are taken in total absence of monitoring and research. Indicators may be direct measurements of physical changes, contaminants or biological and ecological impacts specific to stressors or generic indicators of environmental damage, or proxy measurements of these attributes. They may also relate to particular species or habitats. There is therefore an increasing need for scientists to meet challenging objectives, such as to identify meaningful measures of "quality", and to recommend "indicators" to underpin implementation of directives, conventions, statutes and other more informal national and international initiatives (Strain and Macdonald, 2002; Hiscock et al., 2003).

In pollution studies and general environmental reporting, the use if indicator species is more widespread, and has a long history. Whether the species is reliably informative about the status of the environment is partially determined by the way in which the indicator species is chosen and used. When the indicator species approach is used to evaluate the impact of a specific activity, or the effectiveness of a particular management measure/policy on an ecosystem/environment, there may be enough information to select an indicator that is most likely to be both fully exposed to the activity and highly sensitive to it. In circumstances when reporting on the status of individual species is required to meet legal or policy commitments, more often than not such species are either designated as threatened or endangered by a recognized agency. For instance in Canada, this can be achieved via COSEWIC (Committee On the Status of Endangered Wildlife In Canada) initiative, in which case an environmental offender can be charged under the endangered species Act. In such cases indicators of the status of individual species will influence environmental decision-making, and may be interpreted as a measure of the health of the environment (Rice, 2003).

The criteria for the selection of indicators for a monitoring program do have a large impact on the cost of the program and on the applicability of the data. A program may have a number of different indicators, but all should be selected in the context of available resources. In general, ideal indicators should be simple, inexpensive, sensitive, and straightforward to interpret (Strain and Macdonald, 2002). A good indicator should be able to answer management questions and readily understood by decision-makers, easily measurable, and should be something for which a legal requirement to report (arising from international agreements or community bylaws) exists. In addition, there should be a sound theoretical basis in ecological theory for which the indicator should be reliably informative about effects of particular forcers (Nicholson and Fryer, 2002; Hiscock et al., 2003; Rice, 2003).

In oil pollution monitoring and evaluation of policy effectiveness, a number of indicators are being used worldwide. The most commonly used by many countries include beached bird surveys, tar ball monitoring, aerial surveillance and satellite remote sensing. These are individually reviewed in the following sections.

4.1. Beached bird surveys

Surveys of beached oiled birds are considered as a powerful and highly sensitive indicator for marine oil pollution, as the mortality of seabirds is one of the most obvious adverse effects of pollution of the world's oceans with oil. Systematic surveys of dead seabirds along coastlines and the proportion of oiled beached birds (oil rates) have been used in many parts of the world to document the effect of oil pollution, more so in Western Europe and in parts of North America (Camphuysen and Heubeck, 2001). It is a relatively cost-effective method that can be used to detect trends in chronic oil pollution (Camphuysen, 1998), to evaluate changes in policy or to assess the scale of oiling incidents (Seys et al., 2002). At a basic level, beached bird surveys may be set up just to establish the patterns of occurrence of seabird mortalities. They may also be used to identify the species affected and to assess the scale of damage associated with a particular oiling incident, or as a signal highlighting the adverse effects of oil pollution on seabirds. In the southern North Sea, for instance, this was the initial aim of most beached bird surveys (Camphuysen and Heubeck, 2001; Seys et al., 2002).

The extent of chronic oil pollution in the Grand Banks, south of Newfoundland off the east coast of Canada, was assessed by conducting systematic beached bird surveys between 1984 and 1999 (Wiese and Ryan, 2003). The region lies within major shipping routes from Europe to North America, and is a year-round feeding habitat for large concentrations of numerous species of seabirds, making them vulnerable to oil pollution. The results indicate that chronic oil pollution in this region is among the highest in the world. Over the 16-year period, 62% of all the dead birds found had oiled feathers. The mean number of oiled birds per kilometer was found to be 0.77 and thus higher than reported average for other parts of the world (0.02-0.33). There was a strong correlation between oiling rates and weather patterns, indicative of the likelihood of illegal dumping occurring year round.

In Central California, beached bird and oil pollution data gathered over a 10-year period (1993-2002) by Beach Watch, a long term shoreline monitoring programme, also indicated a strong correlation with weather patterns (Roletto et al., 2003). The temporal patterns in bird encounter rates along 32 beach segments showed a doubling of beached birds and a six-fold increase in oiled birds during the 1997/98 El Nino-Southern Oscillation (ENSO) event (Roletto et al., 2003). Over the 10-year survey period, 3.2% of the beached birds were found to be oiled. In surveys conducted to assess long-term changes in oil pollution along the Belgian coast for a period of ten years (1990-1999), the proportion of oiled birds represented 44.7% of the beachcast corpses (Seys et al., 2002). Comparable results from a 20-year survey period (1977-1997) on the Netherlands coast indicated that 46% of beached birds were oiled (Camphuysen, 1998).

Thus, there exists great variability in the rates of seabird mortality that may be attributed to oil pollution at sea (Table 1).

Table 1

Summary of oiled birds (percent of total encountered) recorded by beach survey programs in the Atlantic and Pacific Oceans (Source: Roletto et al., 2003).

Year	Site	Oiling Fraction	Reference
1977-1997	Holland	46%	Camphuysen, 1998
1984-1999	Newfoundland	62%	Wiese & Ryan, 2003
1978-1979	Inland Washington	< 1%	Speich & Wahl, 1986
1980-1986	Central California	6.3%	Bodkin & Jameson, 1991
1990-1999	Belgium	44.7%	Seys et al., 2002
1993-2002	Central California	3.2%	Roletto et al., 2003
2000-2003	Oregon and Washington	2.2%	Hass & Parrish unpublished

The variability in the oil fractions is probably attributable to the amount of ship traffic in these areas. For instance, the North Sea has some of the busiest shipping routes in the world, with about 420,000 shipping movements per year, in addition to numerous ferry crossings and fishing trips (Volckaert et al., 2000). The high oiling rates shown for the North Sea countries (Holland and Belgium) can thus be accounted for on this basis. The Grand Banks south of Newfoundland also has dense ship traffic routes, being traversed by vessels traveling the Great Circle Route between Europe and North America (Wiese and Ryan, 2003). Thus the highest oiling fraction shown for Newfoundland can be ascribed to this factor. The ship traffic traversing the Kenyan coast from the Middle East through the

Mozambique Channel is not as heavy as in the above two cases. However, the nearly 5000 tanker voyages per year carrying 30% of the world's crude oil production (ITOPF, 2003) present a constant, yet relatively low threat to the marine environment.

Beached bird surveys provide relatively cost-effective additional information on the occurrence of oil at sea and the frequency of illegal spills that would be much more expensive, or could not be collected by other methods such as aerial surveys. Theoretically, the oil rate of beached birds should decline if measures to reduce oil pollution are being effective, and should remain stable or increase if they are not, or if pollution levels increase. The sensitivity of beached bird surveys to management interventions has been evident in the North Sea, where a reduction of illegal discharges of oil at sea were recorded as a reduction of oiled bird encounter rates on beaches in the German Bight and the northern isles of Scotland. Similarly, an increase in the penalties for polluting of coastal waters was reflected in a marked reduction in oiled bird encounter rates on Polish beaches. Conversely, a concerted multi-agency effort to combat illegal discharges of oil off Newfoundland reportedly coincided with an increase in oiled bird encounter rates on south coast shorelines (Wiese and Ryan, 1999).

Subtle changes in oiled bird encounter rates recorded along the beaches of the Wadden Sea were considered indicative of the positive results of attempts to protect certain areas of the North Sea, and a decline in oil rates over time (Camphuysen, 1998). Beached bird surveys on the Central California coast reportedly contributed to the discovery, Watch baseline data set was used to help determine the level of impacts from an oil spill in San Francisco, and resulted in a restoration settlement of over $3.6 million for seabirds and impacted habitats (Roletto et al., 2003). Beached bird surveys therefore provide a powerful tool for protection of coastal/marine environment from the adverse effects of oil pollution.

4.2. Tar ball monitoring

Tar balls from routine tanker operations, accidental oil spills, leakage of crude oils from inland and off-shore oil fields, and run-off from land-based activities continue to contaminate many coastlines and amenity beaches all over the world. Tar balls are water-insoluble fragments or lumps of oil weathered to a semi-solid or solid consistency, deposited on beaches or the sea bottom. They range in size from a pinhead to approximately 30 cm in diameter. They are formed during the weathering process through the combination of high molecular weight and persistent components of oil with debris present in the water column. Quantitative measurement of beach tar has been used as an indicator of chronic petroleum pollution of the oceans from a variety of sources (Butler et al., 1998). Systematic beach surveys of stranded tar balls may also be used for the identification of oil pollution trends. The chemical characterization of the components of the petroleum residues can be used to identify the sources of the pollution. The identification of the exact sources of the oil pollution contributes to the effectiveness of regulation and prosecutions (Zakaria et al., 2001; Owens et al., 2002).

The numerous pelagic tar ball studies reported in the literature indicate a high temporal and spatial variability of tar ball concentrations. Concentrations have been shown to vary by a factor of 10 or more at a single station during the course of a single day, and by as much as a factor of 500 in the course of a year. This variability may be partially attributable to sampling error, but many factors interact to determine the distribution and concentration of stranded pelagic tar balls. Tar ball deposits are not cumulative (distribution is dynamic), and represent only the material deposited in the intertidal portion of the beach during the last tidal cycle. In addition, the amount of tar in the supratidal zone is dependent on recent winds and waves which constantly redistribute sand to alternately expose and bury tar balls (Owens et al., 2002). The dynamic nature of tar ball distributions and concentrations limit the use of the observations, and the data derived from the field surveys should be regarded primarily as descriptive information, or semi-quantitative at best. Hence the reference to the tar balls counted on survey beaches in the Central California by Beach Watch program as a secondary indicator of chronic oil pollution (Roletto et al., 2003).

Despite the limitations in the use of tar ball data as quantitative indicators of petroleum pollution, the analyses of the chemical components have been successfully used in fingerprinting spilled oils in a number of incidents. Their analysis provides additional information on the source of hydrocarbon contamination and the extent of degradation of the oil spill (Tolosa et al., 2004). Among petroleum pollutants, polycyclic aromatic hydrocarbons (PAHs) are a major class of marine environmental contaminants. The investigation of these compounds is of great interest since several of them exhibit high carcinogenic and mutagenic activities. They arise from the incomplete combustion of organic material, especially fossil fuels (pyrolitic origin), from the discharge of petroleum and its products (petrogenic origin) and from the post-depositional transformation of biogenic precursors (diagenic origin). Generally, crude oils contain 0.2-7% PAHs, and different sources of crude oil give different concentrations of PAHs (Zakaria et al., 2001; Tolosa et al., 2004).

Crude oils are also characterized by aliphatic hydrocarbons such as alkanes, isoprenoids, cyclic aliphatic compounds and other aromatic hydrocarbons, including geochemical biomarkers such as hopanes (norhopanes and homohopanes), steranes, and plant-derived pentacyclic triterpanes such as oleananes and bicardinanes. Thus the molecular marker approach has been applied in the distinguishing of tar ball origins in some investigations on oil spill accidents, in addition to providing a better resolution to the state of petroleum pollution in the marine and coastal environments. In effect, the approach provides a more definitive confirmation of the sources of oil pollution (Barakat et al., 1999; Zakaria et al., 2001; Owens et al., 2002; Tolosa et al., 2004).

The long-term and systematic monitoring of ocean-facing beaches for tar ball distribution provides an expansion of the understanding of possible sources of chronic oil pollution to the fragile coastal/marine ecosystems. In addition, such sampling programmes furnish us with a simple and inexpensive way of verifying changes in pollution input from, say, shipping, and the effectiveness of international laws such as MARPOL 73/78. Golik and Rosenberg (1987) quantitatively evaluated beach-stranded tar balls along the Israeli coastline between 1975 and 1985. The results showed that there was a drastic reduction in tar ball quantity during the period, an occurrence partially attributed to enforcement of regulations against oil pollution. Thus monitoring and quantification of beach tar is a good indicator of the problem of petroleum pollution, and of the effectiveness of oil pollution control policies put in place by national and international authorities (Butler et al., 1998; Zakaria et al., 2001).

4.3. Aerial surveillance of oil spills

Aerial surveillance of oil spills from vessels and platforms is conducted by a number of countries in order to deter illegal operational discharges and to evaluate marine pollution. It is therefore a valuable indicator of the state of pollution of the marine environment, and the effectiveness of the pollution control measures. The monitoring approach is widely used over the North Sea, Baltic Sea, and the Mediterranean Sea (International Maritime Organization "special areas"). In these, illegal discharges of oil from maritime transport, off-shore installations, and refineries (both offshore and shore-sited) -the three main sources of oil pollution into the marine environment - is prohibited. Aerial surveillance is therefore conducted to prevent and detect any violations of these regulations on tankers and platforms (EEA, 2001).

The North Sea (designated a "special area" since 1 August 1999) has some of the busiest shipping routes in the world, with about 420,000 shipping movements per year within its limits. Traffic is particularly heavy through the Traffic Separation Scheme (TSS) with, at times, 150 ships per day sailing along the axis to and from the North Sea, in addition to numerous ferry crossings and fishing trips. A pernicious consequence of this dense shipping traffic is chronic oil pollution (Volckaert et al., 2000). However, despite important international legal efforts to prevent and repress oil pollution from ships (United Nations, 1982), the number of accidental and illegal operational oil spills from ships has remained unacceptably high (Volckaert et al., 2000).

Within the framework of the Bonn Agreement, eight European Union (EU) countries (Belgium, France, Germany, Denmark, The Netherlands, Sweden, United Kingdom and Norway) execute aerial surveillance over the North Sea as an aid to detecting and combating oil pollution and to prevent anti-pollution regulations. Under the HELCOM Convention, nine countries (Denmark, Germany, Finland, Sweden, Poland, Russia, Latvia, Lithuania and Estonia) participate in aerial surveillance over the Baltic Sea with the same objective (EEA, 2001). For instance Belgium, under the Bonn Agreement and the Belgian law on the prevention of pollution of the sea by ships, conducts an aerial surveillance program initiated in 1991. The aims include, among others, deterring illegal operational discharges and evaluating marine pollution in the Belgian's Maritime Zone of Interest (BMZI), which overlaps with areas of the Bonn Agreement Zone of Joint Responsibility (Volckaert et al., 2000). The results obtained from aerial surveillance in these areas show that the number of oil slicks exhibited a general decline between 1991 and 1996 (Table 2).

Table 2:

Total annual number of oil slicks by country observed during hours of aerial surveillance (number of observations per flight hours). Source: EEA, 2001, adopted from the Bonn Agreement and HELCOM.

Country	1991	1992	1993	1994	1995	1996
Belgium	16/75 (0.213)	60/189 (0.317)	60/214 (0.280)	82/207 (0.396)	57/206 (0.277)	58/220 (0.264)
Denmark	137/294 (0.466)	45/294 (0.153)	21/294 (0.071)	40/304 (0.132)	65/296 (0.220)	49/361 (0.136)
Netherlands	273/703 (0.388)	288/815 (0.353)	279/721 (0.387)	283/949 (0.298)	238/819 (0.291)	247/897 (0.275)
Finland	23/1820 (0.013)	22/2733 (0.008)	26/2285 (0.011)	23/2313 (0.010)	26/2353 (0.011)	39/2523 (0.015)
Sweden	221/1677 (0.132)	284/1817 (0.156)	256/2011 (0.127)	381/2220 (0.172)	461/2113 (0.218)	262/1924 (0.136)
United Kingdom	135/548 (0.246)	191/636 (0.300)	180/677 (0.266)	147/675 (0.218)	176/868 (0.203)	108/981 (0.110)

It can be gleaned from the computed encounter rates (values in parentheses) that there is a general decline in the number of observed oil slicks in the period 1991-1996. For instance in the case of the United Kingdom, the data indicate that there was a steady decline between 1992 and 1996. This trend could be the result of a decline in illegal discharge of oil from ships and platforms as a result of aerial surveillance in these areas, which increased steadily from 548 to 981 h during the six year period. The Black Sea is one of the IMO "special areas" where operational discharges by ships are prohibited. However, the ports and river mouths along its coastline, as well as the shipping lanes are allegedly severely polluted, a state partially attributed to the absence of aerial surveillance (Pakin, 2004). Monitoring of oil spills by aerial surveillance can thus partially serve an aid to detecting and combating pollution, and to prevent violations of anti-pollution regulations.

4.4. Satellite remote sensing

Space-borne remote sensing has proved to be an important tool in monitoring marine oil spills due to its wide spectral coverage, insensitivity to weather and regular revisit capability. Operational satellite surveillance services are thus being used by a number of pollution control agencies around the world in an effort to control chronic oil pollution through detection and apprehension. National pollution control authorities have the responsibility to maintain some level of monitoring of illegal ship discharges to ensure compliance with the marine protection regulations/legislation and the general protection of the coastal/marine environment (CRISP, 2001). For instance in Norway, it has been shown that ship observations and aircraft surveillance using Side Looking Airborne Radar (SLAR), have proved useful, but they are of limited coverage and high operational cost. Therefore, pollution control authorities in various countries have resorted to satellite Synthetic Aperture Radar (SAR) for their oil spill detection systems (Espedal and Johannessen, 2000).

The large amount of data generated by space-borne SAR systems provides a unique opportunity for global scale oil spill detection, compared to scattered ship observations or aircraft surveillance in limited areas. Due to its large coverage, satellite based SAR has proved very useful in large scale oil spill detection and monitoring. In addition, it has been documented that SAR is capable of detecting oil spills at high spatial resolution, independent of daylight and cloud conditions (Espedal and Wahl, 1999; Espedal and Johannessen, 2000). High resolution SAR has also proved to be a valuable tool for monitoring ships, since much information about detected ships can be derived from the SAR image, including the ships location, heading, length, velocity, and possibly ship type (CRISP, 2001). The application of satellite technology greatly enhances monitoring and enforcement capabilities if used in combination with aerial surveillance. Linking satellite data with aerial surveillance programmes makes the scheduling of surveillance flights more effective by providing vector information to the site of illegal discharge incidents, ultimately enhancing surveillance efforts (CCG, 2003).

Monitoring and evidence gathering involves three processes: (1) routine surveillance to detect possible oil slicks, (2) detailed reconnaissance of detected slicks to verify the presence of oil and (3) identification of the polluter. Suitably equipped aircraft can accomplish all the three tasks, but this is expensive to operate (especially the routine surveillance) and the aircraft rarely detect oil slicks at the time the polluter is present. Effective prevention, response and deterrence therefore require integration of the tools available to pollution control organizations. The use of satellite data allows significant improvement in the surveillance capability for only a very small additional investment. For instance, a combination of one patrol aircraft and a satellite image provides similar performance as, at least, a doubling of the available airborne survey resources. The satellite surveillance is therefore carried out in tandem with aircraft reconnaissance of any detected oil slicks and possible identification of the polluter (Espedal and Wahl, 1999).

A simplified illustration of how a pollution control agency would respond to an illegal oil spill using observation satellite is presented in Fig.4. It provides the context for the exploration of satellite data, showing how elements for surveillance, reconnaissance and polluter identification would interact.

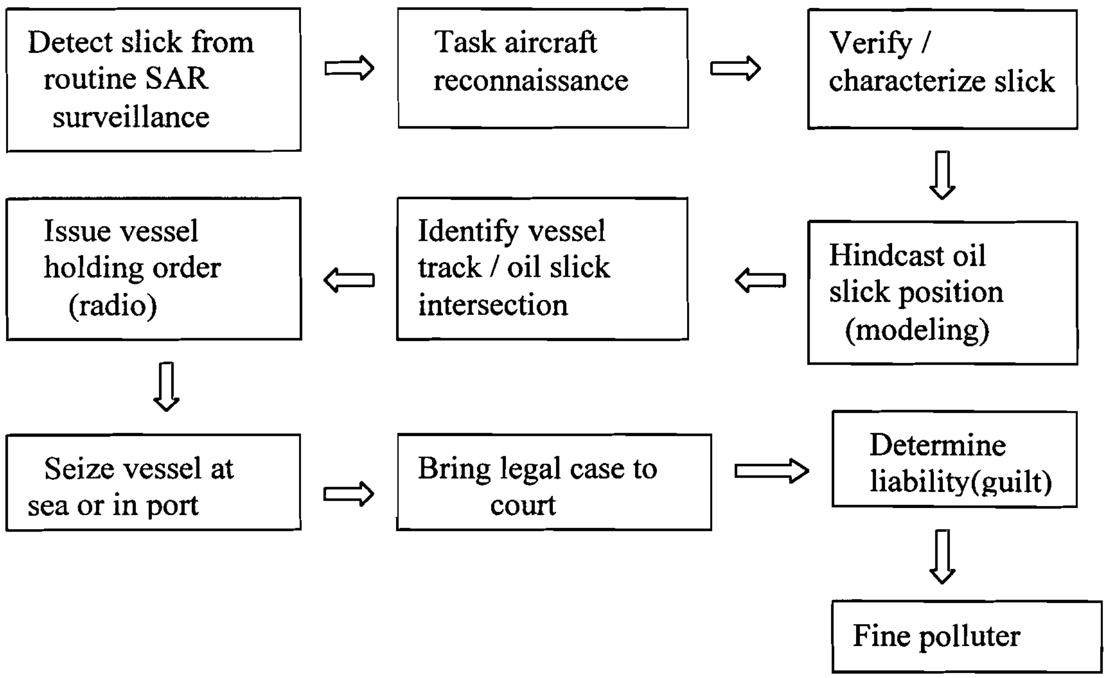

Figure 4: A flow diagram showing how marine pollution control agency would respond to an illegal oil spill following evidence from space-borne satellite data. Source: Espedal and Wahl, 1999.

Further information leading to verification/characterization of oil slicks and their linkage to ship targets (hindcasting oil slick position), is derived from meteorological data such as wind speed and direction, ocean current data, and other features observed on the SAR imagery (Espedal, 1998; Espedal and Wahl, 1999).

Some of the countries currently using satellite surveillance operationally for oil pollution management include Norway (Espedal and Johannessen, 2000), the United Kingdom

(Jones and Mitchelson-Jacob, 1998), the United States of America (Simecek-Beatty and Clemente-Colon, 2004), Canada (CCG, 2003), Singapore (covering a large part of the South East Asian coastal waters), and Denmark. For instance Canada is using RADARSAT-1 SAR technology under the I-STOP (Integrated Satellite Tracking of Polluters) program as a deterrent to chronic oil pollution arising from illegal discharges at sea. The project is meant to enhance the surveillance of the waters off the East Coast through the use of different resources among other objectives. The project collaborators and contributors include Environment Canada (EC), Canadian Space Agency (CSA), Canadian Coast Guard (CCG), Transport Canada (TC), Radar Sat International (RSI) and the Department of National Defense (DND) (CCG, 2003).

The I-STOP project, triggered in part by concern for the deaths of large numbers of seabirds on the shores of Atlantic Canada, was started in September 2002. Only nine days into the project, a ship was apprehended illegally discharging a large amount of oil off the coast of Newfoundland under cover of darkness. The captain of the seized vessel was taken to court and charged for the offence under the Migratory Birds Act (Auld, 2002; Hilliard, 2002). The arrest was a result of the integration of activities following satellite detection of the oil spill, and provision of information to the Canadian Coast Guard pollution surveillance flight. There is evidence of a significant reduction in illegal oily discharges at sea since that incident, a fact attributed to awareness of potential polluters that satellite surveillance is being used as an additional deterrent (Pomeroy, personal communication). Detection of oil discharges using satellite technology thus offers the most promising method for monitoring spills, and hence a good deterrent to illegal discharges of oil at sea.

Documentary evidence points to the four indicators discussed above as the best developed and widely applied so far. A summary of these indicators, with a brief outline of data acquisition methods and their appropriateness for Kenya is presented in Table3.

Table 3:

Best developed indices of effectiveness in oil pollution management, respective data collection methods and possible applicability in Kenya.

Index	Data acquisition method	Appropriateness to Kenya
Beached bird monitoring	Determination of the number of dead and oiled birds per Km of beach surveyed per given time (e.g., per year)	Applicable in Kenya due to its cost-effectiveness
Tar ball monitoring	Determination of the weight of tar balls per unit area (or per unit length) of beach front surveyed during, say, spring low tide	Applicable in Kenya since it is relatively cost-effective
Aerial surveillance	Determination of the number and size of oil spills detected in the territorial sea by surveillance aircraft per unit time (e.g., per flight, per year)	Not very applicable on routine basis due to the high cost implications, but limited use during emergency situations such as massive accidental spills feasible
Satellite remote sensing	Determination of the number and size of oil spills detected by satellite per unit time (e.g., per image, per month)	Can be applicable with good inter-agency co-ordination for pollution data from, say, the European satellite Meteosat Second Generation (MSG)

Four well established indices of the effectiveness of illegal oil pollution control regulations, legislation and policies have thus been identified. Of these, three (i.e., beached bird surveys, tar ball monitoring, and use of satellite-acquired data) can potentially be applied routinely in Kenya. The first two of the three can actually be carried out by scientists from the Kenya Marine and Fisheries Research Institute (KMFRI) if financial resources for regular and consistent surveys can be availed. As for space-borne remote sensing, there is the potential to take advantage of the newly launched Meteosat Second Generation (MSG) satellite and its ground receiving station, an advanced satellite-linked weather monitoring and prediction facility.

The utility is funded by the European Union under the auspices of the Meteorological Transition in Africa Project (MTAP), being hosted by Kenya. The data to be acquired from the facility can find applications not only for meteorological purposes, but also for other fields such as marine environmental monitoring (Agutu, 2004). Thus with good inter-departmental co-ordination, say, between the Meteorological Department and oil pollution control agencies such as the Kenya Ports Authority, arrangements can be made for acquisition of oil spill data to be analyzed for the purpose of deterring illegal discharges in the territorial waters of Kenya. This of course requires a concomitant investment in personnel training and infrastructural improvement.

CHAPTER 5

5.0. A review of some effective regulatory regimes in oil pollution control

There are a good number of countries around the world with effective environmental management regimes, specifically with regard to marine oil pollution from various illegal activities that lead to spillage of oil into the marine environment. These countries have utilized a number of indicators (monitoring tools) to signal the effects of chronic marine oil pollution, and hence to gauge the effectiveness of the available regulatory measures. There are some standardized methods, which lead to highly sensitive monitoring instruments of marine oil pollution. Systematic beached bird surveys have been used in many parts of the world to document the effect of oil pollution, but particularly so in parts of North America and Western Europe.

Several countries (especially in the European Union) use aircraft to monitor shipping lanes for oil pollution, but this is generally too expensive for most developing countries. Beached bird surveys have been found to provide relatively cost-effective information regarding the occurrence and frequency of illegal spills. Theoretically, oil rates (the proportion of dead birds having oiled plumage) should decline if regulatory measures to control oil pollution are being effective. The indicator has been used successfully in the United Kingdom, Belgium, Germany, Poland and the Netherlands (Camphuysen, 1998). A number of satellites have been launched by various countries for space-borne remote sensing of oil spills, and have proved quite useful for spill monitoring. Some of these include Envisat-1 (Europe), SPOT-5 (France), Radarsat-1 (Canada), NOAA (United States of America), ADEOS-2 (Japan), among others (Sherbinin and Giri, 2001). Some of the countries with effective regulatory regimes and the approaches they adopt for monitoring and control are briefly reviewed in this section.

5.1. New Zealand

New Zealand is one of the countries with the longest series of beached bird surveys in the world. These have been organized since 1943 by the Ornithological Society of New Zealand primarily to collect information on the occurrence of storm driven birds. There has been no major oil spill in New Zealand waters, although relatively minor incidents near harbours have affected penguins and shags. The results of beached bird surveys suggest that chronic oil pollution is slight around New Zealand with, for example, only one out of 5000 dead seabirds found in 1994 having an oiled plumage (Camphuysen and Heubeck, 2001). This could only be indicative of an effective regulatory regime in place to combat illegal discharges of oil into the marine environment.

In addition to the organized beached bird surveys, there is a requirement that every ship entering one of the ports in New Zealand has to contribute to a "disaster fund." The purpose of this fund is to cater for emergencies in case of major accidental oil spills, and also to put in place a consistent coastal monitoring programme for the purpose of collecting baseline data. The principal source of funds for the agency responsible for the protection of the marine environment from oil pollution, the Maritime Safety Authority of New Zealand (MSANZ) is a direct levy on vessels that use its safety services. A lesser proportion also comes from direct charges (for instance from registering of ships). Aerial surveillance of the marine and coastal waters is also in place for proper monitoring. However, the practical usage of satellite for oil spill surveillance is still not feasible owing to a number of constraints, such as funding for the purchase of data (Belliss and McNeill, 2004).

With respect to chronic oil pollution emanating from operational spills in the commercial sector and among recreational users, the Maritime Safety Authority of New Zealand initiated an oil spill prevention campaign in 2001. The principal objective of MSANZ is to undertake activities that, among others, provide effective marine pollution prevention and an effective marine oil spill response systems, at reasonable cost. The primary approach of the initiative was a comprehensive educational campaign with the objective of raising awareness on the following issues:

(1) The meaning of the words "oil spill", for instance to boat owners. To most people, mention the words "oil spill" and they will immediately think of, say, the Exxon Valdez spill in Alaska. A small spill of diesel never registers as an oil spill to many people. So the tag line of the campaign adopted by MSANZ was "every drop counts". The aim is to make people realize that preventing the minor spills that occur routinely on boats can have a huge positive impact on the marine environment.

(2) The potential consequences. Given that people do not necessarily consider minor operational spills to be a problem, they are not aware of the potential consequences. It is

therefore important to make people recognize the fact that the cumulative environmental effects of many small discharges are far greater than the so-called "catastrophic" discharges as in accidental spills. Minor spills can be more harmful than the highly visible oils as they tend to be more toxic in the water, and together they represent a far greater volume of oil being discharged into the oceans. Also they are generally released closer to the shore.

(3) How to reduce the spills. This was a key message MSANZ had to get across to boat owners. Thus the use of adsorbent pads in bilges and of products to collect the overflow of fuel from fuel tank breather pipes was promoted, for the prevention and cleaning up of minor spills before entering the marine environment (MSANZ, 2004).

5.2. The North Sea coastal states

Countries bordering the North Sea, namely Belgium, Denmark, France, Germany, the Netherlands, Norway, Sweden, and the United Kingdom, together with other members of the European Union, are members of the Bonn Agreement. The Bonn Agreement is an international agreement by North Sea coastal states, together with the EC to:

- Offer mutual assistance and co-operation in combating pollution;
- Execute surveillance as an aid to detecting and combating pollution and to prevent violations of anti-pollution regulations.

The Agreement has assisted with the development of national capabilities around the North Sea, and emphasis has been on the coordination of surveillance activities, including the opportunities offered by satellite surveillance. Under the Agreement, the countries carry out surveillance flights in their own maritime area. In addition, they co-ordinate the following flights to maximize the benefits of the surveillance aircraft:

- A monthly flight carried out by each country in turn to survey the offshore area of the North Sea where offshore oil and gas activities are taking place.
- Intensive surveys where a few neighbouring countries co-operate to survey an area with a high traffic density.

The North Sea has some of the busiest shipping routes in the world, the consequence of which is chronic oil pollution (Volckaert et al., 2000). These intensive surveys are judged to be particularly useful because they are high profile with several aircraft in an area with high shipping intensity. The continuous sequence of aerial surveillance is supported by sea-borne assistance and where possible, with data from satellite observations. Thus a permanent presence (e.g., over a period of 24 hours) in the sea is ensured. This raises the profile of the surveillance operations, enhances the enforcement of discharge provisions at sea, and increases the deterrent effect of the aerial surveillance efforts. The exercises are also an ideal way of improving the co-operation between the participating authorities (Lunel et al., 2001).

Due to the highly organized way of operation, many European countries are leaders in the field of surveillance and combating marine pollution. The way in which surveillance and prosecution is dealt with through the Bonn Agreement offers an ideal model to be adopted elsewhere in the world. The increased likelihood of discovery and prosecution has also given the incentive to put in place improvements in operating procedures on board tankers and on off-shore platforms. The general level of maintenance of equipment and vessels has been improved through Port State Control and the number of Port reception facilities has also increased significantly. All these factors have played a significant role in the decline in operational discharges of oil from the crude oil tanker fleet and oil production platforms into the marine environment (Carpenter and Macgill, 2001; Lunel et al., 2001).

Chronic oil pollution in the North Sea has been responsible for the death of countless numbers of seabirds since the beginning of the twentieth century, as signaled by the strandings of oiled birds. Many thousands of oiled seabirds have been known to wash ashore in the coastal states of the North Sea such as the Netherlands, Belgium, Germany, Denmark, and along the British east coast (Stowe and Underwood, 1984; Camphuysen, 1998). Extensive beached bird surveys were conducted in the Netherlands over a 20 year period between 1977 and 1997, covering 27,251 km of the coast. Analysis of the results revealed consistent declines in oil rates in virtually all species and groups of birds, and nearly all trends were significant. This overall decline in oil rates is interpreted as being indicative of a significant decline in chronic oil pollution in (parts of) the (southern) North Sea (Camphuysen, 1998).

Beached Bird Surveys are shown to result in valuable data that may be used as a quite sensitive tool to study the effectiveness of measures to reduce oil pollution at sea. It is a cost-effective method that demonstrates long-term changes in oil pollution, which in turn can be used to evaluate changes in policy. A complete picture of the recent changes in marine oil pollution in European waters could definitely emerge from a trend analysis of data available in other European countries. For instance, similar surveys have indicated recent reductions in oil pollution in the north British archipelagos (Orkney and Shetland), Poland and Germany (Camphuysen, 1998; Camphuysen and Heubeck, 2001; Seys et al., 2002). The long history of the use of beached bird surveys in many of the European countries as an indicator of the problem of chronic oil pollution in the North Sea, and the institution of monitoring instruments such as aerial surveillance that have led to drastic reduction of illegal discharges into the marine environment, are therefore closely intertwined.

5.3. Canada

Chronic oil pollution has been a serious problem in eastern Canada since at least the 1940s. Despite this, and some large oiling incidents during the period when beached bird surveys were becoming consolidated in Europe (Levy, 1980), only two schemes have been initiated on the Atlantic coasts of North America, in Newfoundland sporadically since the 1950s and more systematically there since 1984 by the Canadian Wildlife Service (Wiese and Ryan, 2003). The results of beached bird surveys in Newfoundland by Environment Canada, the Canadian Coast Guard and Operation Clean Feather (1980-1998) suggested that chronic oil pollution was a more serious threat to seabirds in these Atlantic waters than for instance along the west coast (Simons, 1985). Recent analysis showed most alarmingly that the oil rate in beached birds in Newfoundland has consistently increased, and that chronic oil pollution off the coast of Newfoundland (with an overall oiling rate of 62%) is among the highest documented in the world (Wiese and Ryan, 2003).

Beached bird surveys and oiling rates are thus valuable indicators of chronic oil pollution in this region, with wider implications and applications. Declining oiling rates of seabirds in parts of the North Sea over the last two decades or so were interpreted as a decline in overall chronic pollution, and attributed to more extensive aerial surveillance and decisions to clean-up oil slicks rather than wait for natural dissipation (Camphuysen, 1998; Camphuysen and Heubeck, 2001). However, for the Newfoundland region, detailed investigation of the data showed that increasing trends in oiling rates were mainly due to a reduction in the numbers of clean birds found on beaches, and this was related to increasingly warmer winter ambient air temperatures. Improved weather conditions have the effect of reducing natural mortality. Despite the underlying patterns, for instance weather, influencing oiling rates, the data collected during these surveys was useful in determining trends in chronic oil pollution (Wiese and Ryan, 2003).

Bird oiling rates in Newfoundland in the early 1980s were similar to, or occasionally even higher than, those in the more polluted parts of the North Sea, but with variable levels after the mid 1980s (Camphuysen and Heubeck, 2001). Recent analyses of Newfoundland data, however, suggest a gradual increase in oil rates during the 1990s (Wiese and Ryan, 1999; Camphuysen and Heubeck, 2001). To help reduce and hopefully eliminate illegal dumping of oil at sea, a strong year-round enforcement presence, awareness creation, and establishment of accessible and convenient oil disposal facilities in Atlantic Canada, have been recommended (Wiese and Ryan, 2003). Other approaches which have been recommended to curb this chronic oil pollution problem include, but are not limited to: increase in fines to act as deterrent to polluters; establishment of a system to reward people who file pollution reports leading to convictions; appointment of pollution prevention officers (PPOs); increasing the effectiveness of aerial surveillance; and the use of satellite imaging as a full-time tool to detect ship-source oil pollution (Wiese, 2003).

The Canadian government has responded to this chronic marine oil pollution problem by implementing some of these recommendations. Thus the current response includes a surveillance programme that uses aircraft and Pollution Prevention Officers to survey the areas with dense ship traffic, and document evidence of ships illegally discharging oil into the marine environment. However, due to the very long coastline, the mobilization of adequate resources to ensure adequate surveillance coverage has always been a challenge (CCG, 2003). To enhance surveillance capabilities, the various government stakeholder agencies have suggested the use of satellite technology to be incorporated in the monitoring and enforcement programme. To this end, a multi-agency project known as Integrated Satellite Tracking of Polluters (I-STOP) has been initiated. The major goal of the ISTOP project was to determine if Radarsat technology could be harnessed to the task of reducing chronic oil pollution on the east coast of Canada (CCG, 2003).

The I-STOP project started in September 2002, and had a number of specific objectives. These included but were not limited to the following: - to increase or improve the surveillance of the waters off the east coast of Canada through the use of different resources; to determine if Radarsat can be linked with the Canadian Coast Guard's (CCG's) National Aerial Surveillance Programme (NASP) to make the scheduling of the CCG pollution surveillance flights more effective by providing vector information to cite illegal discharge incidents, ultimately enhancing surveillance efforts; and to provide an additional deterrent to illegal discharges of oil at sea (CCG, 2003). Only nine days after inception, it led to the arrest of a ship found illegally discharging a large quantity of oil off the coast of Newfoundland. The captain of the ship was taken to court and charged with the offence (Auld, 2002; Hilliard, 2002). Following this incident, there seems to be a drastic decline in illegal discharges as evidenced from the Radasat-1 data being acquired on a daily basis. The decline can partially be ascribed to the effectiveness of satellite technology as a deterrent to illegal discharges of oil at sea (Joe Pomeroy, personal communication).

Although some countries have seen the benefit of space-based data contribution to oil spill surveillance, there are several major constraints to the acceptance of this technology. These include difficulty in the practical incorporation of the technology into the existing oil spill strategy, and the slow delivery of data occasioned by the time lapse between detection and acquisition of information due to absence of fast processors. More importantly, funding for the purchase of data is a major problem for most countries. For instance, base mapping of New Zealand ports by ERS-2 SAR (European Remote Sensing satellite) has raised the profile of satellite monitoring of oil spills in the country's area of interest. However, the above constraints have acted against the acceptance of this technology. For practical usage, oil spill surveillance from satellites requires the availability of suitably fast processors where the data can be downloaded. Such facilities are currently available in North America, Norway, the United Kingdom and Singapore (Belliss and McNeill, 2004).

CHAPTER 6

6.0. The potential risk of oil spills and Institutional capacity to deal with them in Kenya

The potential for an oil spill refers to the possibility or likelihood of a spill actually occurring. Being aware of the concern about the possible impact of oil spills on the marine environment is important for stepping up efforts to protect the ecosystems surrounding the operations of the oil-associated industries. Thus the consideration of the potential of spill occurrence is a pre-requisite for putting the risks and responsibilities associated with oil in the sea in their true perspective.

Institutional capacity refers to the attributes that make for an effective and sustainable institution. These attributes can generally be organized into three clusters, namely: institutional resources, institutional performance and institutional sustainability. Institutional resources represent the attributes an organization possesses or controls and consists of its basic legal structure, assured access to human, financial, technical, and other resources, and its management systems and structure, including performance management systems. Institutional performance measures an institution's program, services, or other impacts as a result of how effectively it employs its institutional and technical resources. Institutional sustainability incorporates more forward-looking attributes such as organizational autonomy, leadership, and learning capacity which, in turn, help ensure sustainability and self-reliance in the future.

6.1. Potential risk of oil spills

The risk of oil spills occurring in Kenya's territorial waters is potentially high due to the fact that the Kenyan coast and EEZ is adjacent to the busy tanker route from the Middle East to the Cape of Good Hope and beyond, and to the high frequency with which coastal tankers call into the port of Mombasa (ITOPF, 2003; NMOSCP, 2003). Other contributors to risk are operations of the Kenya Petroleum Refineries Limited, the Independent Power Producers, and of a large fleet of motorized and passenger vessels and fishing boats operating nearshore (NMOSCP, 2003).

6.1.1. Ocean-going commercial ships

The Kenya coastal area is bordered offshore by a major tanker route from the Middle East to other parts of the world. About 5000 tanker voyages per year, carrying 30% of the world's crude oil production, have been reported to pass along the route along the East African coast (ITOPF, 2003). It is estimated that 50 ships of various types are in the major shipping lanes off the coast of Kenya at any given time. Of these, up to 9 are likely to be oil tankers with capacities ranging from 50,000 to 250,000 tonnes (UNEP, 1998). Despite the fact that most of this traffic passes about 250 nautical miles offshore, tar balls originating from bilge discharges and tank washings have been observed on the shores occasionally (UNEP, 1998; NMOSCP, 2003). This indicates that an oil spill occurring offshore beyond the EEZ has the potential to reach the shoreline, depending on the prevailing monsoon winds and currents (UNEP, 1982; NMOSCP, 2003).

6.1.2. Coastal trade vessels

A more significant risk of oil spills occurring in Kenya's territorial waters is from coastal tankers en-route to or from the port of Mombasa (NMOSCP, 2003). The entrance to the port of Mombasa is fairly narrow due to the presence of the fringing coral reefs that must be negotiated (UNEP, 1998). The port is congested with the multitude of small inshore vessels that operate there. This enhances the risk of collisions and groundings as there is less room to manoeuvre the vessels within the port (UNEP, 1998; NMOSCP, 2003). The available statistics indicate that the major oil spills emanate from groundings near coral reef areas and narrow creeks. For instance between 1973 and 1994, the amount of oil spilled in the territorial waters of Kenya due to cargo vessel and tanker groundings was estimated to be 396,720 metric tonnes (NMOSCP, 2003).

6.1.3. Petroleum refineries, power producers, and fishing boats

Operations within the Kenya Petroleum Refineries could also result in major oil spills into the coastal waters. The refinery processes approximately 2 million tonnes of imported crude oil annually (NMOSCP, 2003). Improper handling procedures at the oil terminals could therefore result in major spills. Operations of the independent power producers also have the potential to result in catastrophic oil spills. For instance, the accidental rupture of a fuel storage tank at Kipevu Power Station in 1988 resulted in the spillage of 3,000 – 5,000 tonnes of oil into an ecologically sensitive sheltered Makupa creek. The spill resulted in the death of mangroves and seagrasses, with no recovery to date (Obura, 2001; Ochieng and Erftemeijer, 2003; NMOSCP, 2003). Marine fisheries in Kenya are mainly artisanal and undertaken mostly from small non-motorized boats such as outriggers and dhows (UNEP, 1998). The motorized fishing craft are comparatively small in number, and as such the potential threat of oil spills from their operations is bound to be minimal in comparison to refineries and power stations.

6.2. Institutional capacity

The Kenya Ports Authority (KPA) under the KPA Act (Cap 391 of the Laws of Kenya), is responsible for the control of oil pollution within the territorial waters of Kenya. However, for expert information regarding the sensitivity of the shoreline with respect to oil pollution, say, environmental data of those areas that are likely to be impacted by oil spills along the coast, KPA enlists the support of the Kenya Wildlife Service (KWS) and Kenya Marine and Fisheries Research Institute (KMFRI) (NMOSCP, 2003). Kenya Wildlife Service is a state corporate body established by the Wildlife (Management and Conservation) Act of 1976 (Cap 376 of the Laws of Kenya). The broad mandate of KWS is the conservation and management of wildlife resources (both aquatic and terrestrial) in Kenya. By way of a 1989 amendment of the Wildlife Act, the responsibilities of KWS within this mandate include, but are not limited to: 1) the promotion of conservation through various community programmes, and 2) conducting and coordinating research activities in the fields of wildlife conservation and management.

Kenya Marine and Fisheries Research Institute (KMFRI) is a government parastatal organization with a research mandate defined by article No. 4 of the Science and Technology Act of 1979 (Cap 250 of the Laws of Kenya). The research activities KMFRI is empowered to carry out include, among others, the monitoring of pollution in Kenya's aquatic environments (fresh and marine waters). KMFRI also involves communities in some of its research programmes. Recommendations emanating from the research findings are forwarded to policy makers for further action. The calibre of technical and research personnel available in these two institutions (KWS and KMFRI) is just what is needed for (oil) pollution monitoring of Kenya's coastal marine environment. The same personnel are capable of conducting educational campaigns aimed at controlling operational oil spills from, say, land-based activities and small boats, if provided with the other necessary resources.

In terms of the availability of equipment, the KPA is equipped to adequately respond to tier 2 oil spills of up to 3,000 tonnes. In addition to the resources of the KPA, the Oil Spill Mutual Aid Group (OSMAG) supports the KPA in responding to and mitigating the effects of spills greater than tier 2 that affect them. OSMAG is a cooperative group consisting of the various oil companies such as ExxonMobil Kenya Ltd., Caltex Oil (Kenya) Ltd., BP Kenya Ltd., Total Kenya Ltd., Kenya Petroleum Refineries Ltd., and Kenya Pipeline Company Ltd. (NMOSCP, 2003). In case of a major oil spill that affects an OSMAG member, the OSMAG will be mobilized at the request of the KPA to provide integrated support to the latter in combating the spill. For larger oil spills beyond the capability of the KPA and the oil companies (OSMAG) i.e., tier 3, international assistance will be sought, either by the Government through the established international protocols, or by the oil companies who subscribe to international response organizations such as the Oil Spill Response Ltd. (ORSL) in Southampton, England, or East Asia Response Ltd. (EARL) in Singapore (NMOSCP, 2003). Tier 1 sites, which include shore-side industries with oil transfer sites and vessels, are expected to plan for and be able to provide a clearly identifiable first aid response to pollution incidents for which they are responsible (NMOSCP, 2003).

Under the Environmental Management and Co-ordination Act (EMCA), there is the Environmental (Prevention of Pollution in the Coastal Zone and other Segments of the Environment) Regulation (Kenya Gazette Supplement, 2003). The provisions of this regulation require that oil and other ship wastes be off-loaded at the certified Port Waste Reception Facility available at Mombasa port. Every ship is required to carry an Oil Record Book, and to obtain a certified Port Waste Reception Facility in accordance with MARPOL. No ship is expected to leave a Kenyan port without producing a valid discharge certificate (Kenya Gazette Supplement, 2003). This waste management facility, constructed on recommendations carried in a report on Mombasa by International Maritime Organization (IMO), is a joint venture between a Dutch company Atracon and Kenyan stakeholders (NATION Reporter, 2003).

The KPA and the oil companies provide the financial resources for the operation and maintenance of the available oil spill response equipment. The other agencies involved in research activities such as KMFRI and KWS are allowed to solicit for research funds (grants) from other donor organizations. For instance, KMFRI has undertaken quite a number of research programmes in collaboration with scientists from some European countries such as Belgium, the Netherlands, and Sweden, funded by agencies from the latter's governments. KMFRI and KWS have for instance been involved in a joint research project funded by the Netherlands Wetland Program through KWS. In addition, there are regional organizations such as the Western Indian Ocean Marine Science Association (WIOMSA) from which individual or groups of scientists can solicit for research grants. Thus the possibility exists for acquisition of funds to undertake oil pollution monitoring along the Kenyan coast by scientists from either KMFRI, or in collaboration with scientists from KWS. All that is needed is to present a convincing proposal to potential research grant donors.

CHAPTER 7

7.0. Recommendations

Based on the preceding analyses, a few viable oil pollution management approaches can be recommended for Kenya. Although some approaches used elsewhere are quite effective in oil spill surveillance and prevention, they can not be implemented easily by most countries. This is mainly due to the fact that the technologies involved are more often than not very expensive. To this end, a developing country like Kenya can only do with cost-effective measures at the lower end of the absolute cost scale, some of which have been shown to be quite valuable as gauges of effectiveness of national, regional, and international regulations and policies relating to the control of marine oil pollution. The following measures are therefore recommended for Kenya:

(i) Tar ball monitoring

The quantity of tar balls littering beaches has been shown to be a good measure of chronic oil pollution of the open ocean in many parts of the world. Systematic monitoring of beaches for stranded tar balls has therefore been recommended as a simple and inexpensive way of verifying changes in pollution input form shipping and the

al., 2002). There has been evidence of tar ball pollution along the tourist beaches in Kenya (Mombasa and Malindi), according to surveys conducted earlier by the Kenya

Marine and Fisheries Research Institute (UNEP, 1982). No systematic surveys have been in place on a continuous basis for quite some time, presumably due to the dearth of financial resources. However, due to it's cost-effectiveness and the validity of the method as an indicator of the state of petroleum pollution of the coastal marine environment, tar ball monitoring is an approach KMFRI should seriously consider continuing with on a regular basis. KMFRI is the government agency mandated by the Science and Technology Act, Cap 250 of the Laws of Kenya, to undertake pollution monitoring in Kenya's coastal marine environment. The results of such studies and recommendations thereof can then be forwarded to policy makers for further action or inaction.

(ii) Education

Minor operational spills from fishing, recreational and the larger coastal cargo vessels have the potential to contribute to the problem of chronic oil pollution of the coastal marine environment. More often than not the operators of these vessels never have the slightest idea of the deleterious effects of these minor spills of oil from their vessels. There is also documentary evidence to the effect that used lubricating oils from garages and petrol stations (land-based activities) are not disposed of in any organized manner, as disposal varies from burying underground to discharging into drains, rivers and creeks (UNEP, 1982). The best approach would be to mobilize some resources and organize comprehensive educational campaigns for operators of these vessels, garages and petrol stations. Operators of industries situated along the coast, such as slaughterhouses, fish-processing plants and others which dispose of waste oil directly into the sea, should also be included in the campaigns. As indicated in Chapter 6, the research personnel at KMFRI are capable of conducting such campaigns if provided with the necessary resources, as they have the power and the entree to work with stakeholder communities.

The objective of the campaigns should be to sensitize these groups of people about the potential impacts of these minor spills on the marine environment. Thus awareness should be raised on issues such as the meaning of the words "oil spill" and the cumulative environmental effects of these minor discharges. They should also be educated on how to reduce and prevent spills, and on use of safe disposal methods. Disposal of waste oil in designated reception facilities should be recommended, and the possibilities of recycling of such waste be explored. Such educational campaigns aimed at preventing operational oil spills have been implemented in New Zealand for example, one of the countries where beached bird surveys have indicated that chronic oil pollution is slight (Camphuysen and Heubeck, 2001; MSANZ, 2004).

(iii) Satellite data

Another recommendation that may sound over-ambitious is the use of satellite data as a way of monitoring illegal discharges from ships. Under the auspices of the Meteorological Transition in Africa Project (MTAP), a new advanced satellite-linked weather monitoring and prediction facility (Meteosat Second Generation (MSG) ground receiving station) has been launched. The project is funded by the European Union and is being hosted by Kenya. The project is envisaged to improve Africa's access to data that be included in the campaigns. As indicated in Chapter 6, the research personnel at KMFRI are capable of conducting such campaigns if provided with the necessary resources, as they have the power and the entree to work with stakeholder communities.

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REFERENCES

Agutu, M. 2004. Failure to heed forecasts "cause of floods chaos". Daily Nation, Kenya, 6 May 2004. Nation Media Group Ltd., Nairobi, Kenya.

Al-Yakoob, S., Saeed, T., and Al-Hashash, H. 1993. Polycyclic aromatic hydrocarbons in edible tissues of fish from the Gulf after the 1991 oil spill. Marine Pollution Bulletin 27: 297-301.

Ansari, Z. A., and Ingole, B. 2002. Effect of an oil spill from MV Sea Transporter on intertidal meiofauna at Goa, India. Marine Pollution Bulletin 44: 396-402.

Auld, A. 2002. Satellite detects oil spill off Nfld. The Chronicle-Herald, Halifax, Wed. 11 Sept. 2002.

Bak, R. P. M. 1987. Effects of chronic oil pollution on a Caribbean coral reef. Marine Pollution Bulletin 18(10): 534-539.

Barakat, A. O., Mostsfa, A. R., Rullkotter, J. and Hegazi, A. R. 1999. Application of a multimolecular marker approach to fingerprint petroleum pollution in the marine environment. Marine Pollution Bulletin 38: 535-544.

Barron, M. G., and Ka'Aihue, L. 2001. Potential for photoenhanced toxicity of spilled oil in Prince William Sound and Gulf of Alaska waters. Marine Pollution Bulletin 43(1-6): 86-92.

Belliss, S., and McNeill, S. J. 2004. The ERS oil-spill research project in New Zealand: Demonstration to the commercial market. http://earth.esa.int/pub/ESA DOC/gothenburg/Oral/services.pdf, accessed on 16 June 2004.

Berdugo, V., Harris, R. P., and O'Hara, S. C. 1977. The effect of petroleum hydrocarbons on reproduction of an estuarine planktonic copepod in laboratory cultures. Marine Pollution Bulletin 8(6): 138-143.

Born, A. F., Espinoza, E., Murillo, J. C., Nicolaides, F., and Edgar, G. J. 2003.Effects of the Jessica oil spill on artisanal fisheries in the Galapagos. Marine Pollution Bulletin 47(7-8): 319-324.

Boto, K. G., Alongi, D. M., and Nott, A. L. J. 1989. Dissolved inorganic carbon-bacteria interactions at sediment-water interface in a tropical mangrove system. Marine Ecology Progress Series 51: 243-251.

Bryant, D., Burke, L., McManus, J., and Spadling, M. 1998. Threats to reefs. In: Reefs at Risk: A map-based indicator of potential threats to the world's coral reefs. World Resources Institute, Washington DC, USA, 56pp.

Burns, K. A., Garrity, S. D., and Levings, S. C. 1993. How many years until mangrove ecosystem recover from catastrophic oil spills? Marine Pollution Bulletin 26(5): 239-248.

Burns, K. A., Garrity, S. D., Jorissen, D., McPherson, J., Stoelting, M., Tierney, J., and Yelle-Simmons, L. 1994. The Galeta oil spill. II. Unexpected persistence of oil trapped in mangrove sediments. Estuarine, Coastal and Shelf Science 38: 349-364.

Burns, K. A., Codi, S., Pratt, C., and Duke, N. C. 1999. Weathering of hydrocarbons in mangrove sediments: testing the effects of using dispersants to treat oil spills. Organic Geochemistry 30:1273-1286.

Burns, K. A., Codi, S., and Duke, N. C. 2000. Gladstone, Australia Field Studies: Weathering and degradation of hydrocarbons in oiled mangrove and salt marsh sediments with and without the application of an experimental bioremediation protocol. Marine Pollution Bulletin 41(7-12): 392-402.

Butler, J. N., Wells, P. G., Johnson, S., and Manock, J. J. 1998. Beach Tar on Bermuda: Recent observations and implications for global monitoring. Marine Pollution Bulletin 36(6): 458-463.

Camphuysen, K. C. J. 1998. Beached bird surveys indicate decline in chronic oil pollution in the North Sea. Marine Pollution Bulletin 36(7): 519-526.

Camphuysen, C. J., and Heubeck, M. 2001. Marine oil pollution and beached bird surveys: the development of a sensitive monitoring instrument. Environmental Pollution 112(3): 443-461.

Carpenter, A., and Macgill, S. 2001. Charging for port reception facilities in the North Sea ports: putting theory into practice. Marine Pollution Bulletin 42(4): 257-266.

Catriona, M. O., Macinnis-Ng, and Ralph, P. J. 2003. In situ impact of petrochemicals on the photosynthesis of the seagrass *Zostera capricorni*. Marine Pollution Bulletin 46: 1395-1407.

CCG. 2003. Integrated Satellite Tracking of Polluters (I-STOP): Final report, Canadian Coast Guard, Government of Canada, Ottawa, 24pp + appendices.

Cincinelli, A., Stortini, A. M., Perugini, M., Checchini, L., and Lepri, L. 2001. Organic pollutants in sea-surface microlayer and aerosol in the coastal environment of Leghorn – (Tyrrhenian Sea). Marine Chemistry 76(1-2): 77-98.

Clark, R. B., Frid, C., and Attrill, M. 1997. Marine Pollution, 4th Edition. Claredon Press, Oxford, 161 pp.

CRISP. 2001. Satellite Remote Sensing for Oil Spill Monitoring. Center for Remote Imaging, Sensing and Processing.

CRISP. 2001. Ship Detection by Synthetic Aperture Radar. Center for Remote Imaging, Sensing and Processing.

Cross, J. N., Hardy, J. T., Hose, J. E., Hershelman, G. P., Antrim, L. D., Gossett, R. W.,and Crecelius, E. A. 1987. Contaminant concentrations and toxicity of sea-surface microlayer near Los Angeles, California. Marine Environmental Research 23(4): 307-323.

Da Silva, E. M., Campos, P. M., Teixeria, N. M., and Azevedo, C. C. 1997. Impact of petroleum pollution on aquatic coastal ecosystems in Brazil. Environmental Toxicology and Chemistry 16(1): 112-118.

Dean, T. A., Stekoll, M. S., Jewett, S. C., Smith, R. O., and Hose, J. E. 1998. Eelgrass (*Zostera marina L.*) in Prince William Sound, Alaska: Effects of the Exxon Valdez Oil Spill. Marine Pollution Bulletin 36(3): 201-210.

De Sherbinin, A., and Giri, C. 2001. Remote sensing in support of multilateral environmental agreements: what have we learnt from pilot applications? A paper presented at the open meeting of the Human Dimensions of Global Environmental Change Research Community, Rio de Janeiro, Brazil, 6-8 October 2001, 20pp.

Dicks, B. 1984. Oil pollution in the Red Sea – Environmental monitoring of an oilfield in a coral area, Gulf of Suez. Deep Sea Research Part A: Oceanographic Research Papers 31(6-8): 833-854.

Duke, N. C., Burns, K. A., Swannell, R. P. J., Dalhaus, O., and Rupp, R. J. 2000. Dispersant use and a bioremediation strategy as alternate means of reducing impacts of large oil spills on mangroves: The Gladstone Field Trials. Marine Pollution Bulletin 41(7-12): 403-412.

Duke, N. C., and Watkinson, A. J. 2002. Chlorophyll-deficient propagules of *Avicennia marina* and apparent longer term deterioration of mangrove fitness in oil-polluted sediments. Marine Pollution Bulletin 44: 1269-1276.

Durako, M. J., Kenworhty, W. J., Fatemy, S. M. R., Valavi, H., and Thayer, G. W. 1993. Assessment of the toxicity of Kuwait crude oil on the photosynthesis and respiration of seagrasses of the northern Gulf. Marine Pollution Bulletin 27: 223-227.

Dutrieux, E., Martin, F., and Debry, A. 1990. Growth and mortality of Sonneratia caseolaris planted on an experimentally oil-polluted soil. Marine Pollution Bulletin 21(2): 62-68.

EEA. 2001. Accidental and illegal discharges of oil at sea. European Environment Agency.
http://themes.eea.eu.int/sectors_and_activities/transport/indicators/consequences/discharges, 2001/Oil_Spills_TERM_2001.doc.pdf, accessed on 7 June 2004.

Eid, E-M. E., and Fawzi, M. A. 1991. Egyptian approach towards appropriate use of coastal zones on the Red Sea. Marine Pollution Bulletin 23: 331-337.

Elgershuizen, J. H. B. W., and De Kruijf, H. A. M. 1976. Toxicity of crude oils and a dispersant to the stony coral *Madracis mirabilis*. Marine pollution Bulletin 7(2): 22-25.

Ellis, D. V. 2003. The precautionary principle and environmental monitoring. Marine Pollution Bulletin 46: 933-934.

EMCA. 1999. Environmental Management and Coordination Act No. 8 of 1999. Government Printer, Nairobi, Kenya, 81pp.

Epstein, N., Bak, R. P. M., and Binkevich, B. 2000. Toxicity of third generation dispersants and dispersed Egyptian crude oil on Red Sea coral larvae. Marine Pollution Bulletin 41(6): 497-503.

Espedal, H. A. 1998. Oil spill and its look-alikes in ERS SAR imagery. Earth Observation and Remote Sensing, Russian Academy of Science 5: 94-102.

Espedal, H. A., and Wahl, T. 1999. Satellite SAR oil spill detection using wind history information. International Journal of Remote Sensing 20(1): 49-65.

Espedal, H. A., and Johannessen, O. M. 2000. Detection of oil spills near offshore installations using synthetic aperture radar (SAR). International Journal of Remote Sensing 21(11): 2141-2144.

Fingas, M. F. 1993. An introduction to oil spill behaviour, chemistry, and mathematical modeling. Spill Technology Newsletter 18(4), 12pp.

Fricke, A. H., Henning, H. F-K. O., and Orren, M. J. 1981. Relationship between oil pollution and psammolittoral meiofauna density of two South African beaches. Marine Environmental Research 5(1): 59-77.

Garrity, S. D., Levings, S. C., and Burns, K. A. 1994. The Galeta oil spill. I. Long-term effects on the physical structure of the mangrove fringe. Estuarine, Coastal and Shelf Science 38: 327-348.

GOK. 2002. National Assessment Report for the World Summit on Sustainable Development (RIO + 10), Johannesburg, South Africa. Government of Kenya, 54+ pp.

Golik, A., and Rosenberg, N. 1987. Quantitative evaluation of beach-stranded tar balls by means of air photographs. Marine Pollution Bulletin 18(6):289-293.

Gullstrom, M., de la Torre, C., Maricela, B., Salomao, O., Bjork, M., Dahlberg, M., Kautsky, N., Ronnback, P., and Ohman, M. C. 2002. Seagrass ecosystems in the Western Indian Ocean. AMBIO 31(7): 588-596.

Guzman, H. M., and Holst, I. 1993. Effects of chronic oil-sediment pollution on the reproduction of the Caribbean reef coral *Siderastrea siderea*. Marine Pollution Bulletin 26(5): 276-282.

Gysen, J., Bachus, K., Bruyninckx, H. 2002. Evaluating the effectiveness of environmental policy: An analysis of conceptual and methodological issues. Paper presented at the European Evaluation Society Seville Conference, October 10-12, 2002, Seville, Spain, 17pp.

Hardy, J., Kiesser, S., Antrim, L., Stubin, A., Kocan, R., and Strand, J. 1987. The sea-surface microlayer of Puget sound: Part I. Toxic effects on fish eggs and larvae. Marine Environmental Research 2394): 227-249.

Hardy, J. T., Crecelius, E. A., Antrim, L. D., Broadhurst, V. L., Apts, C. W., Gurtisen, J. M., and Fortman, T. J. 1987. The sea-surface microlayer of Puget sound: Part II. Concentrations of contaminants and relation to toxicity. Marine Environmental Research 23(4): 251-271.

Hardy, J. T., Crecelius, E. A., Antrim, L. D., Kiesser, S. L., Broadhurst, V. L., Boehm, P. D., Steinhauer, W. G., and Coogan, T. H. 1990. Aquatic surface microlayer contamination in Chesapeake bay. Marine Chemistry 28(4): 333-351.

Hatcher, A., and Larkum, A. W. D. 1982. The effects of short-term exposure to Bass Strait crude oil and Corexit 8667 on benthic community metabolism in *Posidonia australis* (*Hook f*) dominated microcosms. Aquatic Botany 12(3): 219-227.

Hatcher, B. G., Johannes, R. E., and Robertson, A. I. 1989. Review of research relevant to the conservation of shallow tropical marine ecosystems. Oceanogr. Mar. Biol. Annu. Rev. 27: 337-414.

Hegazy, A. K. 1998. Perspectives on survival, phenology, litter fall and decomposition, and calorific content of Avicennia marina in the Arabian Gulf region. Journal of Arid Environments 40: 417-429.

Hellou, J., Stenson, G., Ni, I-H., and Payne, J. F. 1990. Polycyclic aromatic hydrocarbons in muscle tissue of marine mammals from the Northwest Atlantic. Marine Pollution Bulletin 21(10): 469-473.

Henning, H. F-K. O., and Fricke, A. H. 1980. Recovery potential and chronic oil pollution on the South African coast. Marine Pollution Bulletin 11(10): 301-302.

Heras, H., Ackman, R. G., and Macpherson, E. J. 1992. Tainting of Atlantic salmon (Salmo salar) by petroleum hydrocarbons during a short-term exposure. Marine Pollution Bulletin 24(6): 310-315.

Hilliard, W. 2002. Huge oil spill found off South coast. The Telegram, Halifax, Tues. 10 Sept. 2002.

Hiscock, K., Elliot, M., Laffoley, D., and Rogers, S. 2003. Data use and information creation: challenges for marine scientists and for managers. Marine Pollution Bulletin 46: 534-541.

Hodgson, G. 1999. A global assessment of human effects on coral reefs. Marine Pollution Bulletin 38: 345-355.

Islam, S. M., and Tanaka, M. 2004. Impacts of pollution on coastal and marine ecosystems including coastal and marine fisheries and approach for management: a review and synthesis. Marine Pollution Bulletin 48: 624-649.

ITOPF. 2003. Regional Profiles: Eastern Africa. The International Tanker Owners Pollution Federation Limited (ITOPF), December 2003, 4 pp.

Jones, B., and Mitchelson-Jacob, E. G. 1998. On the interpretation of SAR imagery from the Sea Empress oil spill. International Journal of Remote Sensing 19(4): 789-795.

Kenya Gazette Supplement. 2003. Environmental (Prevention of Pollution in Coastal Zone and other Segments of the Environment) Regulation, 2003 (L.N. No. 159 of 2003), pp. 531-534.

Kiss, A., and Shelton, D. 1991. International Environmental Law. Transnational Publishers Inc., London, England, pp168-178.

Klekowski Jr., E. J., Corredor, J. E., Morell, J. M., and Del Castillo, C. A. 1994. Petroleum pollution and mutation in mangroves. Marine Pollution Bulletin 28(3): 166-169.

Krahn, M. M., Ylitalo, G. M., Buzitis, J., Bolton, J. L., Wigren, C. A., Chan, S., and Varanasi, U. 1993. Analyses for petroleum-related contaminants in marine fish and sediments following the Gulf oil spill. Marine Pollution Bulletin 27: 285-292.

Kucklick, J. R., and Bildeman, T. F. 1994. Organic contaminants in Winyah bay, South Carolina I. Pesticides and polycyclic aromatic hydrocarbons in subsurface and microlayer waters. Marine Environmental Research 37(1): 63-78.

Law, R. J., and Whinnett, J. A. 1992. Polycyclic aromatic hydrocarbons in muscle tissue of harbour porpoises (*Phocoena phocoena*) from UK waters. Marine Pollution Bulletin 24(11): 550-553.

Laws of Kenya. 1977. The Continental Shelf Act: Cap 312. Government Printer, Nairobi, Kenya.

Laws of Kenya. 1977. The Wildlife (Conservation and Management) Act: Cap 376. Government Printer, Nairobi, Kenya.

Laws of Kenya. 1978. The Kenya Ports Authority Act: Cap 391. Government Printer, Nairobi, Kenya.

Laws of Kenya. 1983. The Merchant Shipping Act: Cap 389. Government Printer, Nairobi, Kenya.

Laws of Kenya. 1991. The Fisheries Act: Cap378. Government Printer, Nairobi, Kenya.

Laws of Kenya. 1991. The Maritime Zones Act: Cap 371. Government Printer, Nairobi, Kenya.

Levy, E. M. 1980. Oil pollution and seabirds: Atlantic Canada 1976-1977 and some implications for northern environments. Marine Pollution Bulletin 11: 51-56.

Lunel, T., Capon, M., and Huisman, S. 2001. Prosecutions as a deterrent: Examples from the North Sea. 8pp. http://www.aip.com.au/amosc/papers/lunel t.doc, accessed on 15 June 2004.

Mathews, C. P., Kedidi, S., Fita, N. I., Al-Yahya, A., and Al-Rasheed, K. 1993. Preliminary assessment of the effects of the 1991 Gulf War on Saudi Arabian prawn stocks. Marine Pollution Bulletin 27: 251-271.

McClanahan, T. R., Nugues, M., and, S. 1994. Fish and sea urchin herbivory and competition in Kenyan coastal (Mwachireya) lagoons: the role of reef management. Journal of Experimental Marine Biology and Ecology 184(2): 237-254.

McLachlan, A., and Harty, B. 1982. Effects of crude oil on the supralittoral meiofauna of a sandy beach. Marine Experimental Research 7(1): 71-79.

MSANZ. 2004. Maritime Safety Authority of New Zealand. http://www.msa.govt.nz/Protection/operational.htm, accessed on 16 June 2004.

Munga, D., and Mwaguni, S. M. 2003. Physical alterations and destruction of habitats in Kenya: Policies and Institutional Framework. Technical Report, 32+ pp.

Mutiso, J. W. 2001. Integrated waste management and marine pollution in Kenya. Country report presented at the International Maritime Organization Southern / East African Networking Workshop. Cape Town, South Africa, pp 37-40.

Myers, R. A., and Worm, B. 2003. Rapid worldwide depletion of predatory fish communities. Nature 423: 280-283.

Nation Reporter. 2003. Collection of Ships' waste set to start. Daily Nation, Kenya, 26 September 2003. Nation Media Group Ltd., Nairobi, Kenya.

Naylor, R. L., Goldburg, R. J., Primavera, J. H., Kautsky, N., Beveridge, M. C. M., Clay, J., Folke, C., Lubchenco, J., Mooney, H., and Troell, M. 2000. Effect of aquaculture on world fish supplies. Nature 45: 1017-1024.

Negri, A. P., and Heyward, A. J. 2000. Inhibition of fertilization and larval metamorphosis of the coral *Acropora millepora* (Ehrenberg, 1834) by petroleum products. Marine Pollution Bulletin 41(7-12): 420-427.

Nicholson, M., and Fryer, R. 2002. Developing effective environmental indicators – does a new dog need old tricks? Marine Pollution Bulletin 45: 53-61.

NMOSCP. 2003. National Marine Oil Spill Contingency Plan (Kenya). Draft document.

Obura, D. O. 2001. Kenya. Marine Pollution Bulletin 42(12): 1264-1278.

Ochieng, C. A., and Erftemeijer, P. L. A. 2003. The seagrasses of Kenya and Tanzania. In: Green, E. P. and Short, F. T. (Eds.), World Atlas of Seagrasses (UNEP – WCMC), The University of California Press, Chapter 7, pp82-92.

Odido, M. 1997. Marine Science Country Profiles: Kenya. IOCINCWIO – IV / Inf.5. Intergovernmental Oceanographic Commission and Western Indian Ocean Marine Science Association (IOC / WIOMSA), 49pp.

Owens, E. H., Mauseth, G. S., Martin, C. A., Lamarche, A., and Brown, J. 2002. Tar ball frequency and analytical results from a along-term beach monitoring program. Marine Pollution Bulletin 44: 770-780.

Pakin, E. M. A. 2004. Assessment of the risk of oil pollution in the Black Sea. A paper presented at the 2nd Middle East Technical University Conference on International Relations, 9pp.

Pauly, D., Alder, J., Bennett, E., Christensen, V., Tedmeyers, P., and Watson, R. 2003. The Future for Fisheries. Science 302: 1359-1361.

Peterson, C. H. (Editor). 2001. The "Exxon Valdez" oil spill in Alaska: acute, indirect and chronic effects on the ecosystem. Advances in Marine Biology 39: 1-103.

Price, A. R. G. 1998. Impact of the 1991 Gulf war on the coastal environment and ecosystems: current and future prospects. Environmental International 24(1/2): 91-96.

Proffitt, C. E., Devlin, D. J., and Lindsey, M. 1995. Effects of oil on mangrove seedlings grown under different environmental conditions. Marine Pollution Bulletin 30(12): 788-793.

Ralph, P. J., and Burchett, M. D. 1998. Impact of petrochemicals on the photosynthesis of *Halophila ovalis* using chlorophyll fluorescence. Marine Pollution Bulletin 36 (6): 429-436.

Ramamurthy, V. D. 1991. Effects of oil pollution on bio-ecology and fisheries on certain enclosed coastal regions of Arabian Sea. Marine Pollution Bulletin 23: 239-245.

Reimer, A. A. 1975. Effect of crude oil on corals. Marine Pollution Bulletin 6(3): 39-43.

Rice, J. 2003. Environmental health indicators. Ocean and Coastal Management 46(3-4): 235-259.

Rice, S. D., Thomas, R. E., Carls, M. G., Heintz, R. A., Wertheimer, A. C., Murphy, M. L., Short, J. W., and Moles, A. 2001. Impacts to pink salmon following the Exxon Valdez oil spill: persistence, toxicity, sensitivity, and controversy. Reviews in Fisheries Science 9(3): 165-211.

Roletto, J., Mortenson, J., Harrald, I., Hall, J., and Grella, L. 2003. Beached bird surveys and chronic oil pollution in Central California. Marine Ornithology 31: 21-28.

Saharuddin, A. H. 2001. National ocean policy – new opportunities for Malaysian ocean development. Marine Policy 25: 427-436.

Salm, R. V. 1996. The status of coral reefs in the Western Indian Ocean. Working paper prepared for the International Coral Reef Initiative (ICRI) Workshop, Seychelles, March 1996.

Semesi, A. K., and Howel, K. 1992. The mangroves of the Eastern African region. UNEP, 45pp.

Seys, J., Offringa, H., Waeyeberge, J. N., Meire, P., and Kuijken, E. 2002. An evaluation of beached bird monitoring approaches. Marine Pollution Bulletin 44: 322-333.

Shafir, S., Van Rijn, J., and Rinkevich, B. The use of coral nubbins in coral reef ecotoxicology testing. Biomolecular 2003 Engineering 20: 401-406.

Simecek-Beatty, D., and Clemente-Colon, P. 2004. Locating a sunken vessel using SAR imagery: detection of oil spilled from the SS Jacob Luckenbach. International Journal of Remote Sensing 25(11): 2233-2241.

Simons, M. M. 1985. Beached bird survey project on the Atlantic and Gulf coasts. American Birds 39: 358-362.

Sol, S. Y., Johnson, L. L., Horness, B. H., and Collier, T. K. 2000. Relationship between oil exposure and reproductive parameters in fish collected following the Exxon Valdez oil spill. Marine Pollution Bulletin 40(12): 1139-1147.

Stowe, T. J., and Underwood, L. A. 1984. Oil spillages affecting seabirds in the United Kingdom, 1966-1983. Marine Pollution Bulletin 15: 147-152.

Strain, P. M., and Macdonald, R. W. 2002. Design and implementation of a program to monitor ocean health. Ocean and Coastal Management 45(6-7): 325-355.

Thomas, P., and Bundiantara, L. 1995. Reproductive life history stages sensitive to oil and naphthalene in Atlantic croaker. Marine Environmental Research 39: 147-150.

Thorhaug, A., Marcus, J., and Booker, F. 1986. Oil and dispersed oil on subtropical and tropical seagrasses in laboratory studies. Marine Pollution Bulletin 17: 357-361.

Tolosa, M., de Mora, S., Sheikholeslami, M. Z., Villeneuve, J-P., Bartocci, J., and Cattini, C. 2004. Aliphatic and aromatic hydrocarbons in coastal Caspian Sea sediments. Marine Pollution Bulletin 48(1-2): 44-60.

Trust, K. A., Esler, D., Woodin, B. R., and Stegeman, J. J. 2000. Cytochrome P450 1A induction in sea ducks inhabiting nearshore areas of Prince Edward sound, Alaska. Marine Pollution Bulletin 40(5): 397-403.

UNCLOS. 1982. United Nations Convention on the Law of the Sea. United Nations, New York.

UNEP. 1982. Marine pollution in the East African region. UNEP Regional Seas Reports and Studies No. 8, 54pp.

UNEP. 1982. Environmental problems of the East African region. UNEP Regional Seas Reports and Studies No. 12, 88pp.

UNEP. 1998. Eastern Africa Atlas of coastal resources: Kenya. UNEP / Belgian Administration for Development Cooperation, Nairobi, Kenya, 119 pp.

UNEP. 2003. Global Marine Oil Pollution Information Gateway. Eastern Africa: Regional state of the environment reports. http://oils.gpa.unep.org/framework/region-11.htm, accessed on 18 December 2003.

Volckaert, F. A. M., Kayens, G., Schallier, R., and Jacques, T. G. 2000. Aerial surveillance of operational oil pollution in Belgium's Maritime Zone of Interest. Marine Pollution Bulletin 40(11): 1051-1056.