RESTORATION AND MANAGEMENT FOR DAMAGED ECOSYSTEMS IN THE STATE OF KUWAIT

A Dissertation

by

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ABSTRACT

Land degradation and desertification is a serious global issue facing arid ecosystems. Problems of land degradation in Kuwait deserts have accelerated throughout the world, leading to loss of vegetation cover and topsoil fertility, increasing the intensity of desertification. Environmental disasters had occurred as a result of the Iraq's unlawful invasion and the occupation of Kuwait in 1991 impacted multiple ecosystems by through oil spills and military activities. Therefore, the Kuwaiti government selected six future protected areas, which are damaged and will be managed under a restoration plan. Umm Nigga, which is considered one of these future protected areas, was selected as a study area for our research. The northern portion of Umm Nigga, containing both coastal and desert ecosystems, falls within the boundaries of the De-Militarized Zone (DMZ) adjacent to Iraq, and has been fenced off to restrict public access since 1994. The central objective of this research is was to assess and design a conceptual framework for restoration planning. The specific objectives of this research were to: (1) utilize remote sensing, field assessment, and GIS spatial data to develop a site history for restoration planning, (2) utilize GIS and remote sensing to compare soil erosion models by water including MPSIAC, EMP, and RUSLE, and (3) assess the soil condition at the site by conducting soil and vegetation sampling, and to determine suitable locations for re-vegetation using GIS.

Results showed that vegetation cover increased in the unfenced damaged site after the 1991 Gulf War from 2% in 1988 to 37% in 1998, but then it decreased to 23%

in 2013. In the DMZ (fenced site), the vegetation cover also increased from 0% in 1988 to 40% in 1998, but it continued increasing through 2013 to 64%. We conclude that overgrazing and destructive camping are the major source of disturbance in the damaged areas. Our results also showed that the MPSIAC and EMP models were similar in spatial distribution of erosion, though the MPSIAC had more variability. However, the RUSLE presented unrealistic results. We then identified the amount of soil loss between coastal and desert areas, and fenced and unfenced sites for each model. In the MPSIAC and EMP models, soil loss was different between fenced and unfenced sites at the desert areas, which was higher at the unfenced due to the low vegetation cover. The overall results implied that vegetation cover played an important role in reducing soil erosion. According to the soil sampling and vegetation assessment in the field, we found that the vegetation in the coastal ecosystem site was not damaged, due to difficulty of access by people and grazing animals. However, in the desert ecosystem site, phosphors, potassium, and organic matter were higher at the reference area, and correlated with the higher vegetation cover. We conclude that soil remediation and re-vegetation may not be necessary to restore the damaged sites, given that damaged sites still contain concentration of nutrients which is likely sufficient to support native desert plant growth. Therefore, we believe that fencing alone will likely release the ecosystem in Umm Nigga from the former disturbance and allow recovery. However, if natural recovery does not begin within a few years, then re-vegetation should take place as a secondary option.

DEDICATION

This dissertation is dedicated to my mom and dad for their endless love, encouragement, and financial support during my graduate school. Without their love and support I would never be where I am today. I would also like to dedicate this work to Hoda, who has been a constant source of support and encouragement during the challenges of graduate school and life. I am truly thankful for having you in my life, I love you. I would also like to dedicate this work to my beautiful daughters Aseel and Layal. Further, I would like to dedicate this work to my older brother Jasem Al Safar for his support, and being my sponsor during my graduate school, it is finally over, you can rest now. Finally, I would like to dedicate this work to my brothers Ali Abdullah Mohammad Al Safar, and Ali Haidar, and my sisters Samar, Sara, and Hawraa for their love, support, and encouragement.

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CHAPTER I

INTRODUCTION

Land degradation and desertification are considered serious global issues. Arid and semiarid lands occur in regions with low rainfall and limited water resources, as the water lost through evapotranspiration exceeds the water gained from precipitation (Sjoholm et al. 1989; Allen 1995; Sowell 2001; Bainbridge 2007). Such a harsh climate and limited water resources make arid regions more vulnerable for disturbance and their recovery may be very slow, which means that they have less resilience and resistance compared with other ecosystems (Whisenant 1999; Bainbridge 2007). Drylands cover one-third of the worlds total land area, and more than half of them are located in the Arabian Gulf countries (Van Andel & Aronson 2012; Busby 2014). More than 2 billion people are living and using these regions.

Generally, the major source of disturbance affecting arid ecosystems is overgrazing, which is increasing with the rapid increase in population around the desert areas leading to more economic pressure (Barrow & Havstad 1996; Papanastasis 2009; Tongway & Ludwig 2011). However, there are other sources affecting arid ecosystems such as military operations, camping, inappropriate farming, and poor irrigation management. It is important to understand that camping in Kuwait differ from other countries. Camping in Kuwait is connected to the weather, which starts from the beginning of November and last to March. A large number of huge tents are placed in the desert containing huge electricity generation, as well as other entertainment activities such as off road deriving and four wheelers, which influence the possibility of plants to grow during this period (Fig 1-1). Such activities may lead to serious environmental problems such as increased salinity, alkalinity, accelerated erosion, soil compaction, loss of productivity and waterlogging (Zaman 1997; Brown 2002; Misak et al. 2002).

Fig 1.1 Examples of camping and other entertainment activities in Kuwait desert.

Once an arid ecosystem is disturbed, it is very difficult to restore due to the harsh climatic conditions, which affects the natural recovery. Disturbed ecosystems also increase the risk of soil erosion and runoff, which affects the topsoil as it contains the highest amount of fertility, affecting plant establishment and it may take thousands of years to recover (Whisenant 1999; Bainbridge 2007). However, several studies have shown that rapid natural recovery could occur in only few years (Brown & Al-Mazrooei 2003). Thus, having slow or rapid natural recovery depends on the degree and level of damage. Therefore, understanding the condition of the soil is necessary to decide whether natural recovery could occur or other strategies need to be undertaken.

Re-vegetation of native plants has become a common goal for most restoration projects in recent years. In arid ecosystems, seeding of native species often fail due to planting in unsuitable sites, using a low number of seeds, or using poor seed quality. Re-vegetation is not an easy approach; it requires several assessments for soil condition, seed collection processing, seedbank assessment, and seed quality work. The outcome of using seeds depends on a combination of environmental indicators and the genetic characteristics of the species (Roundy & Biedenbender 1995; Whisenant 1999; Bainbridge 2007). Planted seeds could also follow different trajectories depending on environmental conditions such as the amount of precipitation and soil moisture. If the conditions are favorable for germination, then seeds could initiate and complete germination, otherwise, seeds would not germinate, but could remain viable in the seedbank (Blomquist & Lyon 1995; Barrow & Havstad 1996).

Therefore, no single assessment is appropriate for all restoration projects as ecosystems have unique combination of processes (Whisenant 1999). Restoration in arid systems can be costly and it has been estimated that the cost of arid lands restoration range from \$60 to \$3000/ha (Berger 1990). Therefore, it is necessary to determine the cost before selecting the objectives. A large cost does not necessary yield better results, and it may also fail (Berger 1990).

Successful restoration requires a holistic view of the interactions between humans and the environment through time. Successful restoration projects are correlated with the understanding of the environment including soils, animals, plants, and human activities or people involved (Berger 1990; Allen 1995; Van Andel & Aronson 2012). They also require an adequate description of the ecological site before work can begin. Often, the first step is to assess the disturbed ecosystem including identifying surface soil conditions, relevant hydrological processes (infiltration and runoff), and nutrient cycles (Whisenant 1999; Tongway & Ludwig 2011). Additionally, it is necessary to understand the history of the location in terms of the type, nature, location, and intensity of disturbances. Developing a site history is one of the most important and useful steps in restoration planning as it helps in understanding what we are seeing today, guides our choice of assessment tools, and informs our views in determining the best restoration strategy (Berger 1990; Bainbridge 2007).

Geographical Information Systems (GIS) and remote sensing are powerful technologies in assessing and modeling ecological problems, as they help in exploring and analyzing spatial data, and support in finding appropriate solutions for spatial problems. GIS can be used a decision support system to model spatial processes and solve problems analytically. Spatial analysis in GIS examines relationships between geographic features collectively and uses those relationships to describe the real-world phenomena (Clarke et al. 2002; Fotheringham & Rogerson 2013). Remotely sensing is also becoming a widely used technology in understanding site history and ecosystem changes over years (Herold et al. 2002; Groom et al. 2006; Jia et al. 2008). Such technologies can also help in generating necessary ecological information that can help in designing suitable restoration strategies, as well as modeling and simulating future changes. Remote sensing is an essential part of GIS, as most GIS data are generated from remote sensing. Remote sensing data such as satellite imagery helps researchers in mapping vegetation, water, and geology in both space and time. Therefore, the integration of these technologies is considered a powerful tool in assessing, solving, and managing ecological restoration projects (Frohn 1997; Skidmore 2003).

Over past decades, land degradation in the Kuwait desert has accelerated, leading to loss of vegetation cover and topsoil fertility (Brown 2003; Omar 2014). The world's largest hydrocarbon spill, and one of the worst environmental disasters in history, occurred as a result of Iraq's unlawful invasion and occupation of Kuwait in 1991. Multiple ecosystems in Kuwait were contaminated by these spills and associated military activities. Six million barrels of crude oil were spilled into the marine and terrestrial environment and approximately 2-3 million barrels of crude oil were burned (Khordagui & Al-Ajmi 1993). Oil lakes and tarcrete were deposited throughout Kuwait. Groundwater integrity was threatened as a result of the heavily polluted ground surface (Hadi et al. 2006; Omar & Bhat 2008). The observed genotoxicity of oil pollution altered plant growth parameters, such as photosynthetic pigments, proteins, free amino acids, phenols, and reduced sugar levels (Malallah et al. 1998).

As compensation, Kuwait was awarded over $460 million USD to restore its damaged terrestrial ecosystems (UNCC 2002). As a portion of this restoration effort, the re-vegetation of damaged ecosystems will be critical to stabilize the desert surface, regulate the distribution of rainfall, ensure the continued viability of multiple endangered species, and provide sustenance for endemic wildlife. Therefore, Kuwait suggested six locations for future protected areas. These locations are currently damaged and need to be managed under the restoration plan. However, the government approved only four locations as future terrestrial protected areas.

The central objective of this research was to assess and design a conceptual framework for restoration planning. The specific objectives of this research were to:

1. Utilize remote sensing, field assessment, and GIS spatial data to develop a site history for restoration planning.
2. Utilize GIS and remote sensing to compare soil erosion models including Modified Pacific South West Inter Agency Committee (MPSIAC), Erosion Potential Method (EMP), and Revised Universal Soil Loss Equation Method (RUSLE), and to determine their applicability for arid regions such as Kuwait.
3. Assess the soil condition at the site by conducting soil and vegetation sampling, and to determine suitable locations for re-vegetation using GIS.

The findings of this research yielded an understating of the damaged ecosystems in Kuwait and provided stakeholders with tools and technologies to assess and evaluate the damaged locations. In the future, our methods could also be applicable for assessing other restoration projects.

CHAPTER II

THE USE OF REMOTE SENSING TO DEVELOP A SITE HISTORY FOR RESTORATION PLANNING IN AN ARID LANDSCAPE

Overview

Developing a site history and ecological site description is one of the critical steps in restoration planning. This study focuses on Umm Nigga, Northeast of Kuwait, which, was damaged by various anthropogenic activities. The northern portion of Umm Nigga falls within the boundaries of the De-Militarized Zone (DMZ) adjacent to Iraq, and was fenced off to restrict public access since 1994. The central objective of this project was to utilize remote sensing, field assessment, and GIS spatial data to develop a site history for restoration planning of Umm Nigga. Field observation and GIS analysis indicated that the landscape could be divided into three units along a gradient ranging from the coast to inland locations, based on geology, soil properties, and dominant vegetation. Reference sites in the DMZ were also matched for each unit. Remote sensing was used to compare vegetation cover between damaged and reference sites at selected units. Results showed that vegetation cover increased in the unfenced damaged site after the 1991 Gulf War from 2% in 1988 to 37% in 1998, but then it decreased to 23% in 2013. In the DMZ reference site, the vegetation cover also increased from 0% in 1988 to 40% in 1998, but it continued increasing through 2013 to 64%. We conclude that overgrazing and destructive camping are the major source of disturbance in the damaged areas.

Introduction

The development of a successful restoration plan typically requires an adequate description of the ecological site before work can begin. Often, the first step is to assess the disturbed ecosystem including identifying surface soil conditions, relevant hydrological processes (infiltration and runoff), and nutrient cycles at the site (Whisenant 1999; Tongway & Ludwig 2011). Additionally, it is necessary to understand site history in terms of the type, nature, location, and intensity of disturbances. A site history helps in understanding what we are seeing today, guides our choice of assessment tools, and informs our views in determining the best restoration strategy. It also reduces the cost of field assessment and facilitates determining key sources of disturbance (Bainbridge 2007). Understanding the history can also help in identifying appropriate reference sites with similar elevation, aspect, topography, soil, and vegetation community, which is crucial in specifying restoration goals (Whisenant 1999; Cooke & Johnson 2002; Van Andel & Aronson 2012).

Remotely-sensed imagery is widely used to reconstruct parts of the site history. Remote sensing (RS) also enables researchers to analyze dynamic changes in landscapes (Herold et al. 2002; Groom et al. 2006; Jia et al. 2008; Hadeel et al. 2010). According to previous studies, RS can help generate a substantial information needed to evaluate the distribution of vegetation cover (Muthumanickam et al. 2011; Im et al. 2012; Harris et al. 2014) and monitoring landscape degradation in arid and semi-arid environments (Tueller 1989; Washington-Allen et al. 1998; Diouf & Lambin 2001; Washington-Allen et al. 2004; Washington-Allen et al. 2006; Chen et al. 2013). The Normalized Difference

Vegetation Index (NDVI) is a widely used method in RS to evaluate vegetation and measure the amount of photosynthesis in semi-arid lands (Cui et al. 2013). However, there are some concerns with the application of RS in arid landscapes, particularly the difficulty of identifying species in arid lands, as it is mostly compromised of small plants such as shrubs and grasses. Therefore, medium and high-resolution satellite imagery are recommended (Maldonado et al. 2007; Munyati & Mboweni 2013). Several studies have used RS and GIS in evaluating restoration projects. These studies focused on monitoring the success of restored areas through assessing vegetation density using RS (Marignani et al. 2008; Klemas 2013). A few other studies used RS for selecting restoration areas mostly for coastal ecosystems (Mollot & Bilby 2008). We argue that RS is a useful tool in generating substantial amount of information when dealing with arid lands.

Over past decades, the problems of land degradation (which leads to a significant reduction of the productivity due to human activities) (Eswaran et al. 2001) in arid regions such as Kuwait have accelerated throughout the world, leading to loss of vegetation cover and top soil fertility, increasing the intensity of desertification (also called desertization, which is the process by which natural or human causes reduce the biological productivity of drylands such as arid and semiarid lands) (Brown 2003; Reynolds et al. 2007; Omar 2014). The military activites in the first Gulf War in 1990-1991 further damaged the degraded desert ecosystem. Accordingly, pursuant to the approval of the environmental claims submitted by the State of Kuwait to the United Nation Compensation Commission (UNCC), four terrestrial protected areas were proposed to restore terrestrial ecosystems damaged by military activities in Kuwait. Several, independent studies were also undertaken in Kuwait using remote sensing to quantify and assess the environmental damages in Kuwait (El- Gamily 2007). However, none of these studies used RS in restoration planning for damaged ecosystems in Kuwait.

The central objective of this chapter is to utilize remote sensing, field assessment, and GIS spatial data to develop a site history and restoration plans for Umm Nigga, Kuwait, one of the suggested protected areas. Our specific objectives were to (1) identify site variability based on field visits and GIS analysis of geology, soil, and vegetation, (2) quantify the changes in desert vegetation cover in the future protected areas versus the De-Militarized Zone (DMZ) using LANDSAT imagery, for years before and after the first Gulf War, (3) compare vegetative expansion among these areas, as well as, correlate vegetation change with climatic data to determine the major sources and patterns of disturbance. Results of this study will guide decision-makers in defining proper restoration objectives and plans.

Materials Methods

Study Area

Umm Nigga (Fig. 2.1a) is located in the northeastern portion of Kuwait with total area of 246 km^2. Kuwait is located in the desert region in Asia, with an approximate latitude and longitude of 29.3286° N, 48.0034° E, and a total area of 17,820 km^2. It is located in the northeastern edge of the Arabian Peninsula at the top of the Arabian Gulf, sharing borders with Iraq to the North and Saudi Arabia to the West and South. The weather is dry and hot in the summer and warm in the winter with occasional rainfalls (Omar et al. 2000; Alsharhan et al. 2001). Umm Nigga is considered an open rangeland, which is distant from residential areas (around 50 km from Kuwait City). The site is currently used for camping and grazing, with several private farms in the northeastern section only. The site was proposed as a future protected area as a representative of a typical native *halophytic* community vegetation type. The Kuwait Supreme Council for Environment, further extended the requested areas by annexing the entire DMZ, which was created between Kuwait and Iraq by United Nations Security Council Resolution 689. It extends along the Kuwait-Iraq border and the Khawr 'Abd Allah waterway, is about 200 km long, extending 10 km into Iraq and 5 km into Kuwait. Although the DMZ is no longer mandated by the UN Security Council, Kuwait still enforces its portion (Fig. 2.1b).

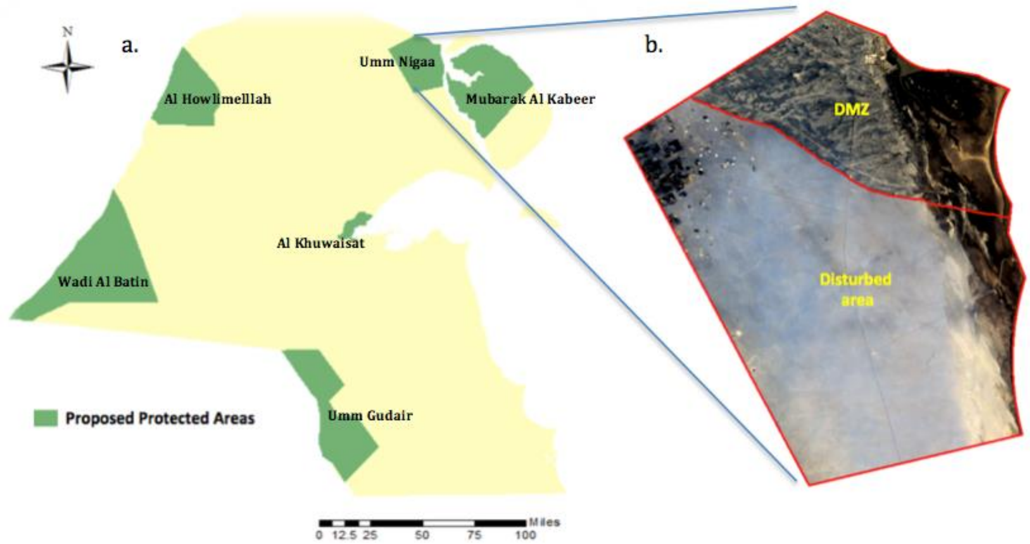

Fig 2.1 (a) Suggested protected area in the State of Kuwait according to the master plan, (b) study area (Umm Nigga), which is divided into De-Militarized Zone (DMZ) and damaged areas.

Historical Site Conditions

Imagery Collection and Processing

Geo-referenced images were obtained from the United States Geological Survey (USGS) for the following years; 1998, 1991, 1993, 1998, 2002, and 2013, which include Landsat 4-5 TM, Landsat 7 ETM+, and Landsat 8 images. Images were selected from months falling within the maximum rainfall season (February-April). The spatial resolution of the images was 30 x 30-m, and the projection was WGS 84 UTM zone 38N (Table 2.1). We could not cover the years from 2003 to 2012, as on May 2003 the Scan Line Corrector (SLC) in the ETM+ instrument failed with Landsat 7. No atmospheric and geometric corrections were necessary for this region due to the low cloud cover, and since multiple images are classified individually, and resulting maps are compared to identify changes (Singh 1989; Foody et al. 1996).

Table 2.1 Details of RS imagery used for the present study

Image Date	Sensor	Spatial Resolution (meters)	Bands	Type
Feb. 1988	Landsat 4-5 TM	30X30	7 (0.45-2.35 µm)	Medium resolution; optical; multispectral
March 1991		30X30		
Feb. 1993		30X30		
April 1998		30X30		
Feb. 2002	Landsat 7 ETM+	30X30	8 (0.45-12.50 µm)	
2013	Landsat 8	30X30	11 (0.43-12.51 µm)	

Image Classification Process

Supervised classification was used in this study using ENVI 5.2 to help in identifying land cover information by selecting regains of interests (ROI) (Jensen 2005) using per-pixel classification logic. Given the fact that the Landsat sensors had different number of bands for each year, all images were also spectrally subset to maintain only the blue (B), green (G), red (R) and near infrared (NIR) bands. Normalized difference vegetation index (NDVI) was also generated and all bands including NDVI were stacked and used in the supervised classification. Each image was divided into five land cover types: soil, bare ground, vegetation, wetlands, and water. Images were classified using Mahalanobis distance in supervised classification methods, as it showed better accuracy assessment compared with other classification methods. Afterwards, the accuracy was assessed for the classified images (Table 2.2) by collecting 50 random ground truth points per class based on the expert's knowledge of the area. Confusion matrix was then created for each image, which provides overall and class-specific accuracy.

Table 2.2 Accuracy assessments for image classification

Year	**Land cover**	**Overall %**	**K-hat**	Producer Accuracy	User's Accuracy
1988	Vegetation	87.43	0.75	83.50	100
	Soil			98	87.5
	Bare ground			100	35.7
	Wetlands			100	98
	Water			100	98.40
1991	Vegetation	89.6	0.87	88	100
	Soil			88	88
	Bare ground			96	85.71
	Wetlands			100	80.65
	Water			76	100
1993	Vegetation	91.2	0.89	80	100
	Soil			98	96
	Bare ground			96	78
	Wetlands			98	90.74
	Water			84	95.45
1998	Vegetation	91.2	0.89	98	90.74
	Soil			92	97.87
	Bare ground			92	76.67
	Wetlands			88	95.65
	Water			86	100
2002	Vegetation	86	0.82	90	100
	Soil			90	81.28
	Bare ground			78	69
	Wetlands			84	91.3
	Water			88	91.76
2013	Vegetation	87.26%	0.828	88.46	100
	Soil			94.59	72.92
	Bare ground			91.18	93.94
	Wetlands			73.53	83.33
	Water			88.46	100

Separation of the Study Area and Selection of Reference Sites

Umm Nigga is comprised of more than one ecosystem. Therefore, we decided to classify the study area into different ecosystems based on ecological aspects, which were collected as GIS-based data sets from the Kuwait Institute of Scientific Research (KISR 1999), including geology (established 1980) (Fig. 2.2a), soil characteristics established 1999 (Fig. 2.2b), and vegetation unit map established by Halwagy (1974) (Fig. 2.2c) with a scale of 1:250,000 We compared the spatial patterning of each of these factors across the future protected area and the adjacent DMZ, with a view of using portions of the DMZ as reference sites, then locations with similar characteristics were digitized. We also used previous studies to describe the digitized units. Finally, the study area was divided into three units based the comparision, with three matching reference sites assigned within the DMZ (Fig. 2.2d). We also conducted field visits observation of the units in order to better outline the ecological description.

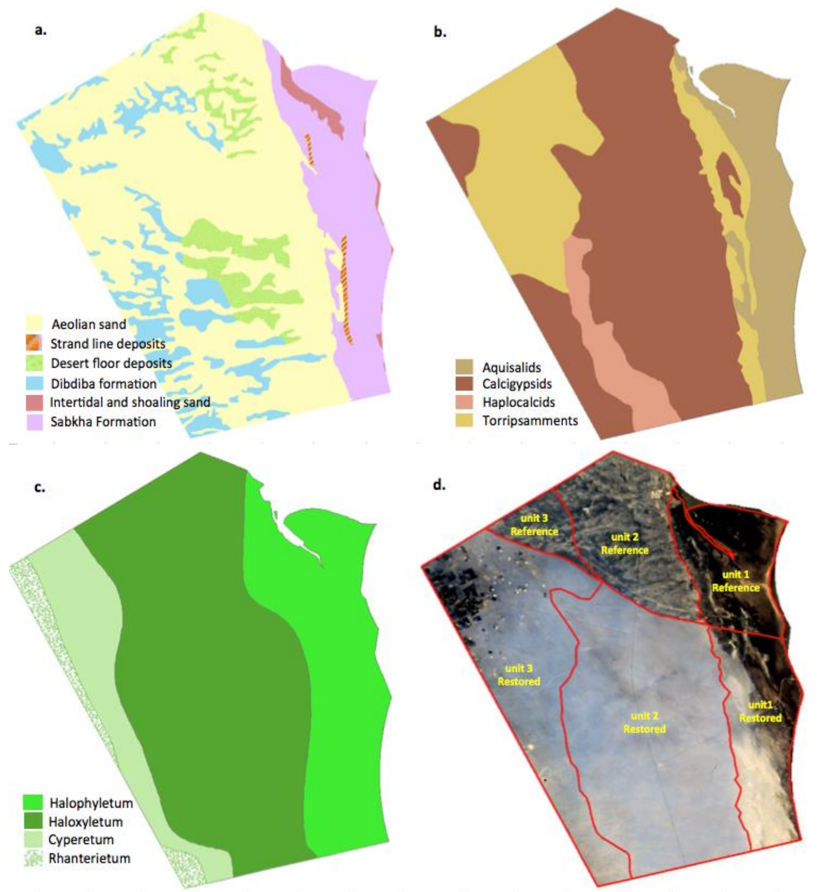

Fig 2.2 (a) Geological units (b) soil groups (c) dominant vegetation unit, and (d) selection of damaged and reference area for each site based on their ecological properties.

Change Detection

We quantified the change over time using standard change detection algorithms in ENVI 5.2, to determine the change in vegetation in the damaged site and DMZ before and after the war. Change detection was also used to compare vegetation cover between reference and damaged areas for each selected unit.

Monthly precipitation and temperature data were also collected from the Kuwait Meteorological Center, Kuwait Airport Station, from 1962- 2013. The annual mean precipitation and annual temperature were calculated to determine the relationship between changes in vegetation cover and climate factors, as vegetation covers may be affected by temperature and precipitation. The climatic analysis covered the first 10 years for period from 1988- 2002, as we did not have enough vegetation data for the period between 2003- 2012. Linear regression analysis was applied to determine the correlation between change in vegetation cover and climate factors. It should be noted that the correlations were based on five points (classified maps) due to data availability. Therefore, this is not a robust outcome, but could give an indication of whether increase in vegetation cover could be correlated with climate factors.

Results

Historical Site Conditions

Changes in Vegetation

Before the first Gulf War 1990-1991, in 1988 the vegetation cover was 2% (Fig. 2.3a). After liberation, the vegetation increased to 6% in 1991, 18% in 1993, and 37% in 1998 (Figs. 2.3b-c-d). However, after 1998, the vegetation decreased to 12% in 2002 (Fig. 2.3e). It then increased again to 23% in 2013 (Fig. 2.3f). The increase in 2013 was due to farming expansion in the private agriculture areas in the northwest of the study area, though the desert itself was still disturbed with a total of 3% vegetation. From the change detection analysis, a large increase in natural vegetation cover occurred in 1998, but it then decreased through 2002, and increased once again in 2013 (Fig. 2.4).

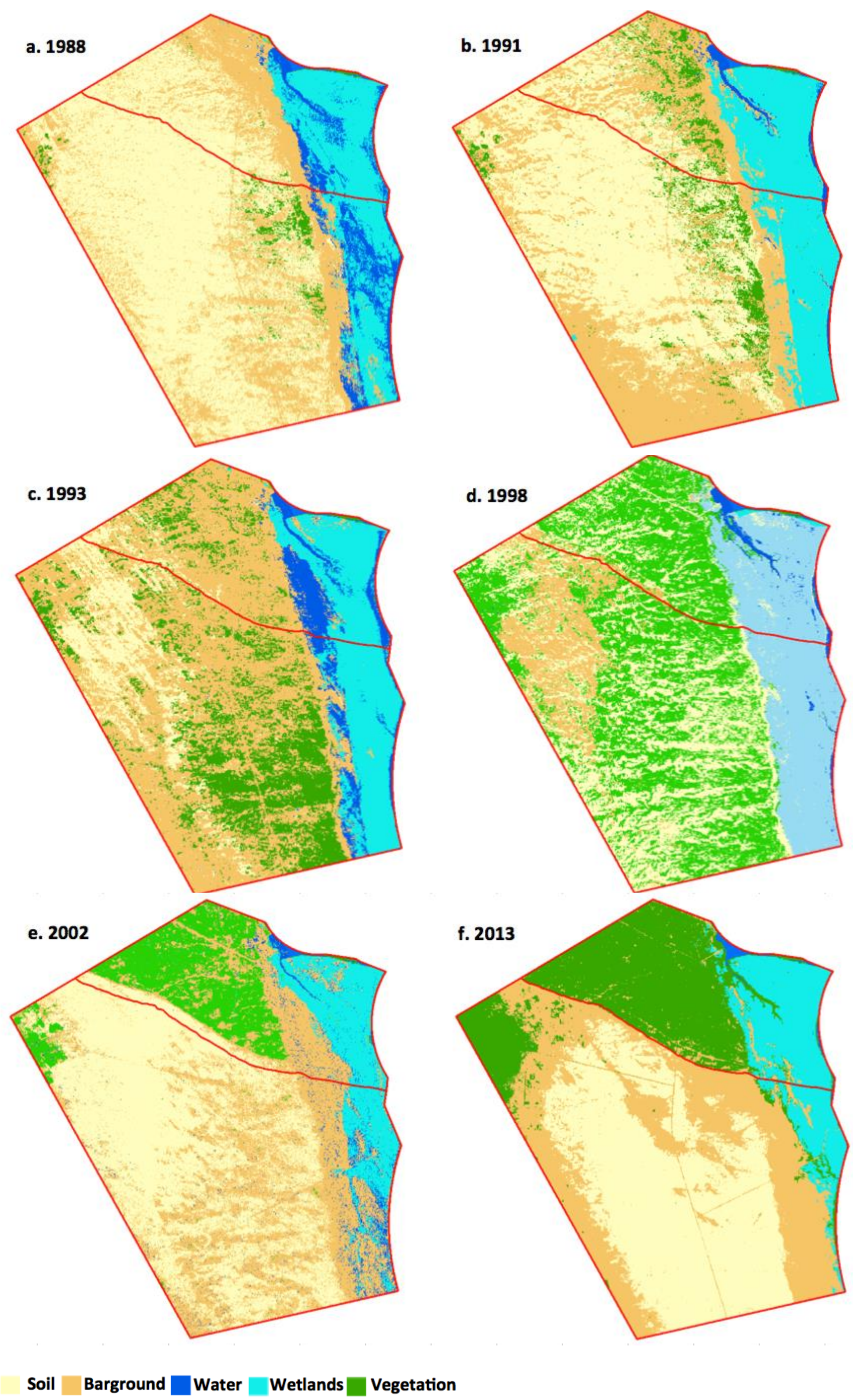

Fig 2.3 Image classification for Umm Nigga including the damaged site (open area) and the De-Militarized Zone (DMZ) (which is fenced and protected by the ministry of interior in Kuwait).

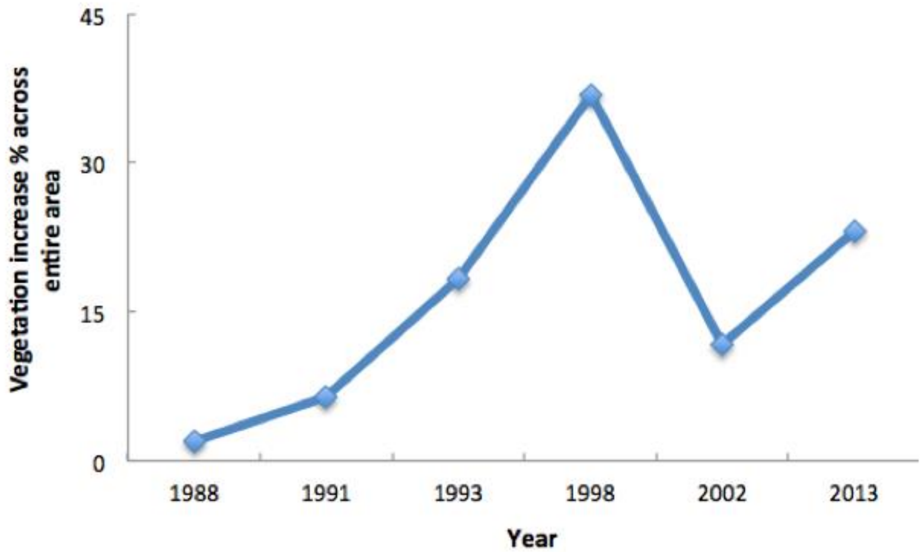

Fig 2.4 Vegetation change detection for Umm Nigga from 1988- 2013.

The differences in cover change are drastic when comparing the damaged area versus the DMZ. The native vegetation within the DMZ increased following the same overall pattern seen in the damaged area, up to 1998. But then it continued to increase past 1998 and on into 2002 and 2013 (Fig. 2.5). The vegetation increased from 0 to 64% over the entire period, from 1988 to 2013. In imagery from 1998, 2002, and 2013, the vegetation increase was of a lower rate. For such reasons, the DMZ was recommended as the possible reference area for future restoration of the damaged area.

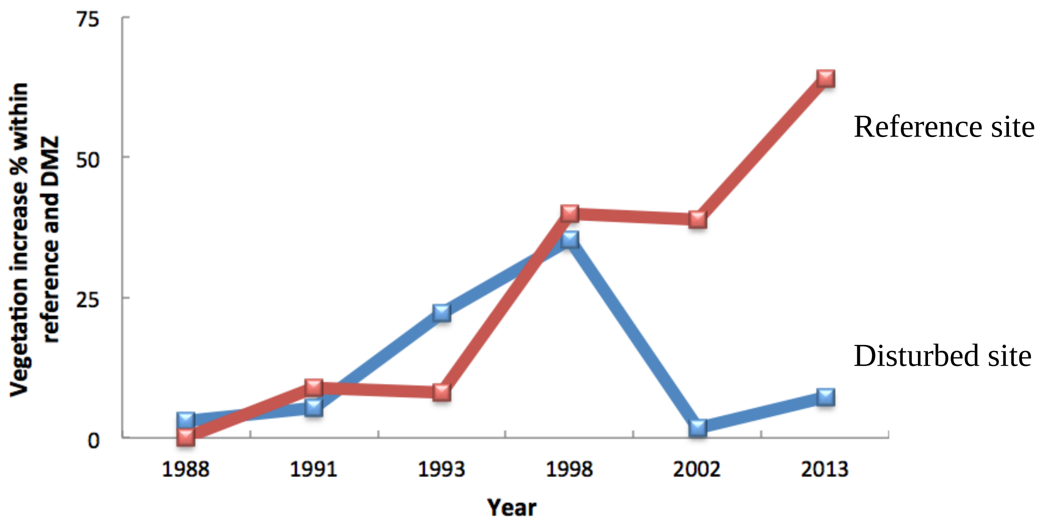

Fig 2.5 Change in vegetation cover at damaged and DMZ areas from 1988- 2013.

Separation of the Study Area and Reference Sites Selection

According to the GIS-based ecological data, vegetation cover, field observation, previous studies and vegetation change detection, the site was separated into three units. Site descriptions for each unit are described in the following paragraphs.

The first land unit is a coastal ecosystem, which is covered with salt marsh and saline depressions, sand dunes, and ridges and terraces. It is composed of *Aquisallid* deep to very deep soils, with very poor drainage. In general, the site is covered with the *Halophyletum* vegetation community unit (Omar 2007). Change detection of satellite imagery illustrates differences between the damaged and reference sites (Fig. 2.6a). From 1988 to 1993 vegetation cover was between 0 and 1% in both the reference and damaged sites. However, it started to increase in both sites after 1993 and continued increasing until 2013, which reached 7% at the damaged site and 15% at the reference site. The percentage vegetation increase was higher at the reference site compared to the damaged site. In the field, the vegetation cover appeared visually similar between the damaged and reference sites (Figs. 2.7a-b). The degree of damage at the two sites was similar, as grazing and camping are not typically conducted to the same degree along the tidal flat due to the muddiness, high salinity, and difficulty of access.

The second land unit is a desert plain ecosystem. It is composed of two soil groups including *Calcigypsids* and *Haplocalcids* (sandy to loamy soils) (KISR 1999). *Haloxyletum* vegetation is dominant in this unit (Al-Sulaimi & Al-Ruwaih 2004; Omar 2007). Vegetation cover was at its lowest in 1988 before the War, with total percentage of 4% for the reference and 0% for the damaged. After the war, and limited access to the northern area of Kuwait for a prolonged period of time, by 1998 the vegetation increased to 53% at the damaged site and 70% at the reference site. Then vegetation dropped again to 1% at the damaged area in 2002, and continued at the same percent of vegetation cover until 2013. In contrast, the vegetation cover continued to increase at the fenced DMZ (reference area) to reach 91% vegetation in 2013 (Fig. 2.6b). Field observation showed that the site in the damaged area is severely disturbed, as evident by the scarce vegetation cover, due to the resumption of grazing, extensive camping and off road driving (Fig. 2.7d). The DMZ exhibited healthy vegetation cover (Fig. 2.7c).

The third land unit is mostly sand and gravel. This unit contains two soil groups, *Torripsamments* and *Calcigypsids* (KISR 1999). The major vegetation unit at the site is *Rhanterietum* (Al-Sulaimi & Al-Ruwaih 2004; Omar 2007). Similar to the second land unit, vegetation was at lowest in 1988, almost 0% for both reference and damaged area. Then it started to increase after the 1990-1991 War and through 1998 to reach 25% at the damaged area and 50% at the reference. However, vegetation then decreased once again at the damaged site to 3% in 2013, while continued increasing at the reference area reaching up to 93% in 2013 (Fig. 2.6c). According to field observations, returning campers and grazing further disturbed the damaged site (Fig. 2.7f). In contrast, high vegetation cover was evident in the fenced DMZ reference area (Fig. 2.7e).

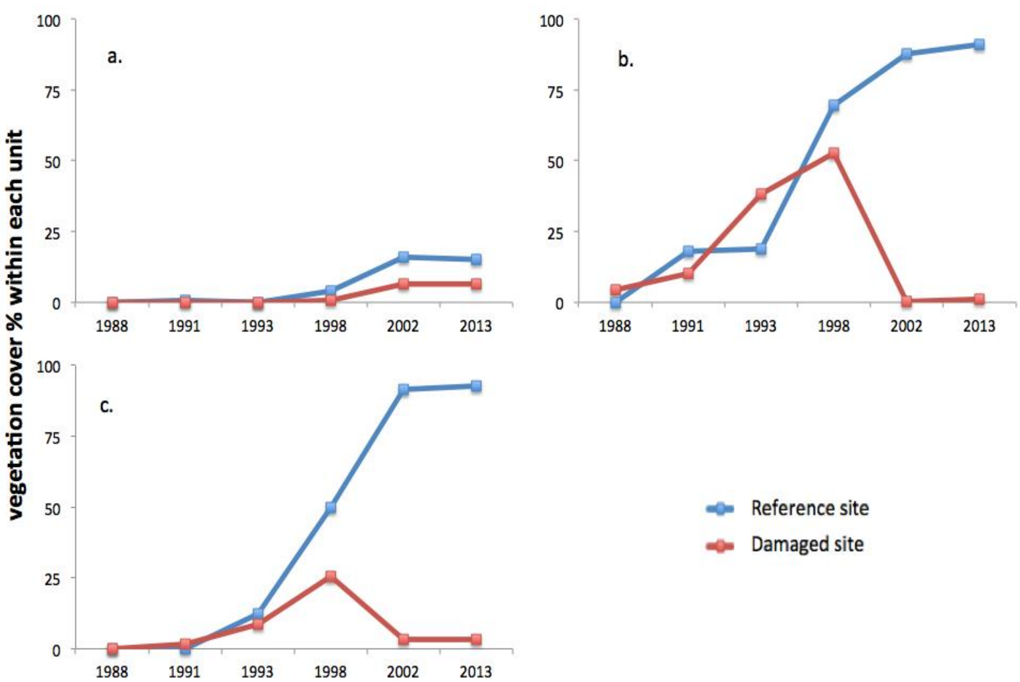

Fig 2.6 Comparison between vegetation cover change for reference and damaged sites in the selected units, (a) unit 1, (b) unit 2, and (c) unit 3.

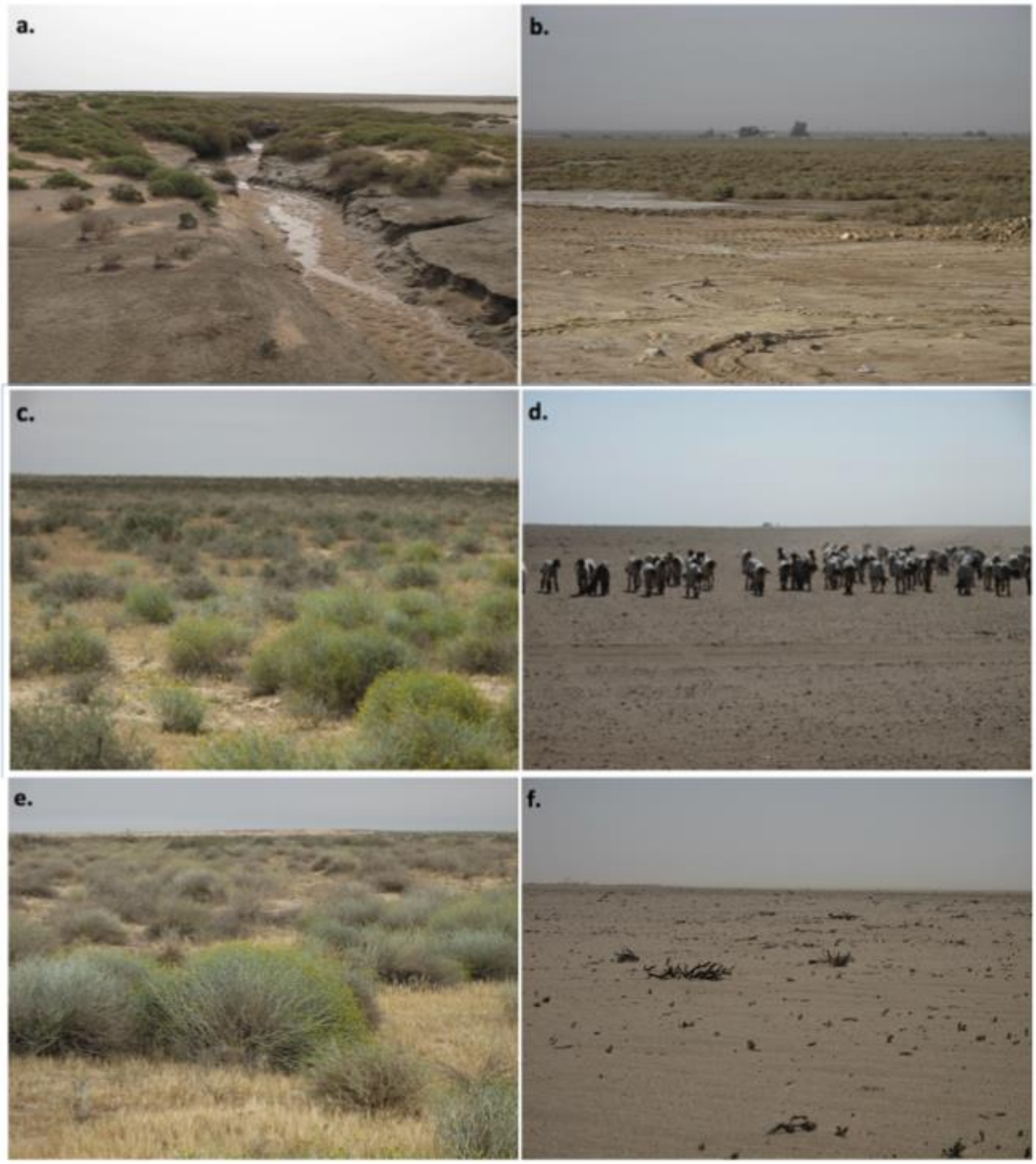

Fig 2.7 Field observation for segmented units, (a) Unit 1 damaged site, (b) Unit 1 reference site, (c) Unit 2 reference site, (d) Unit 2 damaged site, (e) Unit 3 reference site, and (f) Unit 3 damaged site.

Effects of Climatic Factors on Vegetation Cover

The results showed that annual rainfall and vegetation change were moderately correlated in the damaged area (r^2= 0.59, *p* = <1) (Fig. 2.8c). However, there was a high correlation in the DMZ (reference fenced area) (r^2=0.71, *p* = <1) (Fig. 2.8a), as rainfall was the only factor affecting vegetation cover. Annual temperature was also compared with the change in vegetation. According to Al-Fahed et al. (1997) and EPA (2012) there is strong evidence for an increase in the average annual temperatures in Kuwait by ~ 1.6°, over the 48-year period. However, our results showed very low correlation between annual temperature and vegetation change at both sites(r^2= 0.13, *p* = <1) and DMZ (r^2=0.14, *p* = <1) (Figs. 2.8d-b).

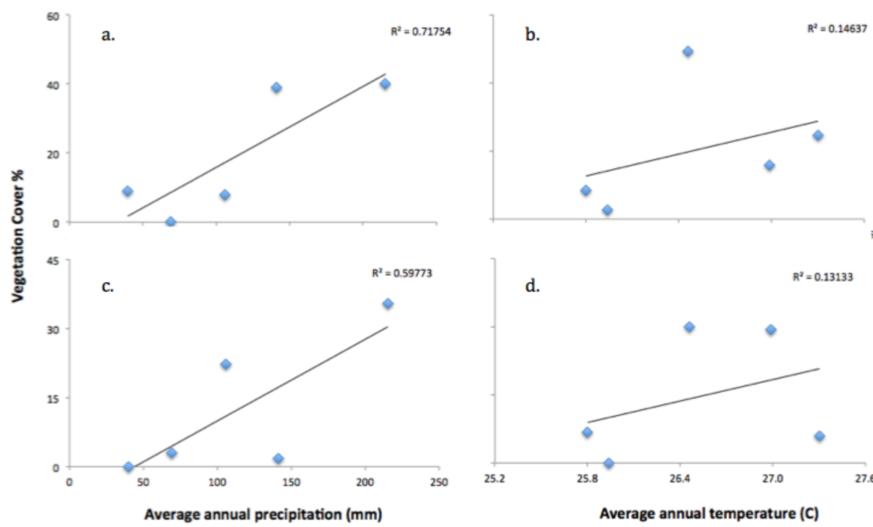

Fig 2.8 Correlation between vegetation cover and climatic factors, (a) annual precipitation and vegetation cover at DMZ, (b) average temperature and vegetation cover at DMZ, (c) annual precipitation and vegetation cover at damaged site, (d) average temperature and vegetation cover at damaged site.

Discussion

Sources and Patterns of Disturbance

Several types of disturbance have occurred at Umm Nigga, including overgrazing, extensive camping, and off-road driving. Military activities took place during 1990-1991, included tanks and armored carriers crossing the desert surface, the destruction of oil and gas wells, and digging and trenching activities. Immediately after liberation, mine clearing operations took place to clear the mines and unexploded ordinance (Omar & Bhat 2008; Devore 2009). As our results show, the vegetation cover was low before the War, and it started to increase thereafter. Following the invasion and liberation, given that the area was heavily mined, the government restricted access to the area as a safety precaution, and this likely resulted in the observed initial increase of plant coverage. Also several de-mining activities took place around the country, hand cleaning by the Pakistani military was used in our study area, which was less disturbing of the soil compared with other operations such as deep plowing with heavy equipment. However, after 1998 the vegetation decreased, coinciding with the conclusion of de-mining operations (Filippino & Paterson 2005; Alsabah et al. 2012). This illustrates that while the war itself was not the sole source of damage in Umm Nigga, the subsequent land management of the area as a result of the war altered the vegetation.

However, grazing activities resumed once again after Umm Nigga was de-mined and declared safe in 1998. From this time on, the vegetation decreased in the unfenced area from 35% in 1998 to 2% in 2002. This rapid decrease in cover occurred over only 4 years. Overgrazing would affect productivity as well as species richness and relative abundance. It might also lead to more severe problems such desertification and loss of soil resources through water and wind erosion (Zaman 1997; Brown 2002; Al-Awadhi et al. 2005; Omar & Bhat 2008). In addition, traditional spring camping practices take place between November and April, which disturbs the vegetation. Such practices lead to soil loss due to the clearance of natural vegetation around the camp. It also causes severe soil compaction, reduction in soil infiltration capacity, and loss of habitat (Misak et al. 2002; Al-Awadhi et al. 2005). According to Omar & Bhat (2008), camping is considered the second most important mechanism of land degredation after overgrazing. Off-road vehicles also reduce vegetation cover and are considered one of the major factors damaging vegetation and the soil surface. It also play an important role in soil compaction (Brown & Schoknecht 2001; Omar & Bhat 2008).

In addition to these factors of disturbance, climatological factors can also affect the vegetation cover. The results suggest a relationship between precipitation and vegetation cover. The analysis showed great variation in annual precipitation during the past 50 years (from 1960 to 2013). Vegetation change detection showed tremendous increase in vegetation in both the damaged area and DMZ (fenced site) during the high rainy seasons between 1993 and 1998; this period also witnessed restricted access to the entire area. In 2002, vegetation percentage decreased at the damaged area, and did not show any changes in the DMZ. This might be related to the drop in average rainfall after 1999. However, after 2002, vegetation did not exhibit any change in the damaged area, and showed tiny increase in the DMZ, which might also be related to low rainfall seasons between 2007-2011 (Figs. 2.9a-b). On the other hand, we believe that slight increase in temperature may not affect the vegetation cover, however, it may affect vegetation trends by replacing the dominant community in the area with new population of the same family or different species (Whisenant 1999).

Vegetation Natural Recovery

As seen in 1988, the vegetation cover was very low and this was before the Gulf War 1990-1991. However, rapid natural recovery occurred at the period between 1991 and 1998. This rapid recovery can be attributed to several factors. First, coarse and sandy soils are usually favorable for plant growth in arid and semiarid lands, as water percolates through the surface layer rapidly. Second, rainfall is central to processes in desert ecosystems. Our results illustrate that an increase in vegetation can be correlated to the high rainfall years (Brown & Al-Mazrooei 2003). Third, seeds can also accumulate and be retained in deep sandy substrates. Many desert plants build up substantial inter-annual seedbanks (Brown & Al-Mazrooei 2001).The study site is located in the pathway of the prevailing northwest wind, which brings continuous deposition of sand, seed and pollen (Al-Dousari et al. 2013). It is worth noting that the vegetation continued to increase at a slower rate at the DMZ after 1998. This may be related to the carrying capacity of the system (Lohmann et al. 2012).

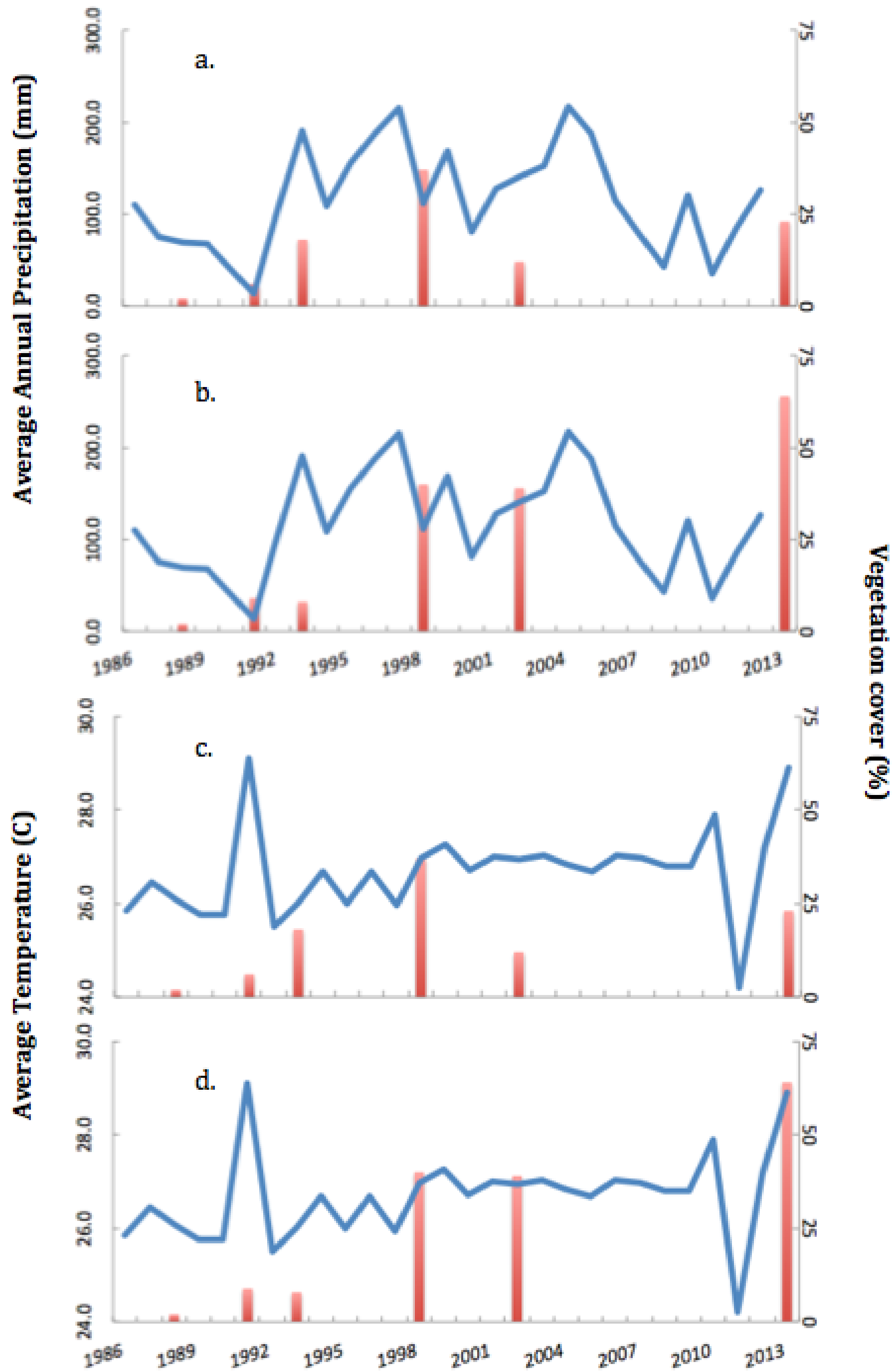

Fig 2.9 Climatic data and vegetation cover in Umm Nigga, (a) average temperature and DMZ, (b) average temperature and damaged site, (c) annual precipitation and DMZ, (d) annual precipitation and damaged site.

The Restoration Plan

Kuwait proposed to re-vegetate areas within approximately 3,500 km^2 of its damaged ecosystems by military activities (UNCC 2002). Umm Nigga is one of the selected areas that will fall under this restoration plan. It was recommended to protect the location by fencing it in order to prevent human disturbance. It was also proposed to develop a local facility with the capability to produce the large numbers of seeds and plants required to re-vegetate 30% of the damaged area. Due to possible episodes of drought that might affect the rate of re-vegetation, inadequate rainfall might not support the recovery and establishment of new-planted vegetation, especially in degraded areas such as Umm Nigga. Therefore, Omar (2014) recommended a drip irrigation system for the re-vegetation areas.

Our study shows that natural recovery occurred after the war from 1991 to 1998, when human activities were limited. However, the site was degraded again after it was cleared in 1998 due to human activities, but the DMZ continued increasing until 2013. Therefore, planting or irrigation may not be necessary given that fencing alone appears to release the ecosystem from the major disturbances. The area may only be fenced, protected and monitored for the first few years. If the site does not show any development after the first few years, then planting can take place as a secondary option. We also contend that using a drip irrigation system may not be an effective method, as native plants are adapted to survive under such arid climate conditions. Moreover in a comparative sense, it is likely that the DMZ was even more heavily compacted before its recovery and following the war, and imagery suggests that it was equally damaged before 1988, likely due to overgrazing and camping. Moreover, such practices require high costs of setting up the system and extensive maintenance (Bainbridge 2002), and leave non-degradable polymers in the soil. In addition, due to the high temperature and evaporation rates, the irrigation of arid lands can cause soil salinization (Misak et al. 2002).

Conclusions

Our work demonstrates the power of utilizing remote sensing to determine the history of a desert site for a relatively large area, when there are no other histories available for reference. NDVI also helped in detecting areas covered with live green vegetation. The study illustrates that understanding the history of the location can make the restoration plan much more effective, as well as finding proper reference sites. It also provided information regarding the level of the problem by knowing if it is ongoing, recent, or historic. More work will be necessary to address the extent of damage through field reconnaissance (soil and plant analyses), assessing water and wind driven erosion rates, and the importance of precipitation or climatic changes on the recovery. Our results indicate that human activities (camping and overgrazing) are the most likely reason for the decrease in the natural vegetation, and that fencing alone may provide an adequate plan for a quick restoration of large areas. However, fencing the location only will not stop the sources of ecosystem disturbance in Kuwait. It is crucial that the country develops a national land management strategy and action plans to manage all land use, including grazing and spring camping in open ecosystems. Increasing public awareness of the problem will also help in controlling negative impacts caused by such activities. The methodology that we used can also be applied for other damaged ecosystems in arid and semiarid regions.

CHAPTER III

COMPARING BETWEEN SPATIAL EMPIRICAL MODELS TO ESTIMATE SOIL EROSION IN ARID ECOSYSTEMS

Overview

The central objective of this project was to utilize GIS and remote sensing to compare soil erosion models (by water) including MPSIAC, EMP, and RUSLE, and to determine their applicability for arid regions such as Kuwait. The northern portion of Umm Nigga, containing both coastal and desert ecosystems, falls within the boundaries of the De-Militarized Zone (DMZ) adjacent to Iraq, and has been fenced off to restrict public access since 1994. Results showed that the MPSIAC and EMP models were similar in spatial distribution of erosion, though the MPSIAC had more variability. However, RUSLE presented unrealistic results. We then identified the amount of soil loss between coastal and desert areas, and fenced and unfenced sites for each model. In the MPSIAC and EMP models, soil loss was different between fenced and unfenced sites at the desert areas, which was higher at the unfenced due to the low vegetation cover. The overall results implied that vegetation cover played an important role in reducing soil erosion, and that fencing is much more important in the desert ecosystems to protect against human activities such as overgrazing. We conclude that the MPSIAC model is best for predicting soil erosion for arid regions such as Kuwait.

Introduction

Soil erosion is a major issue in most arid and semi-arid regions, greatly affecting soil quality and productivity, as most of these soils are generally shallow in depth. Soil erosion is also considered one of the principle mechanisms of desertification processes at national and regional levels (Martín-Fernández & Martínez-Núñez 2011; Kairis et al. 2013). The consequences of desertification involve vegetation and soil loss, reduction in soil fertility and biodiversity, and reduction in rainfall infiltration rates (Vásquez-Méndez et al. 2011). According to the United Nations Convention to Combat Desertification (UNEP 1994), desertification was defined as “land degradation in arid, semi-arid and dry sub-humid areas, resulting from various factors, including climatic variations and human activities”. However, erosion is difficult to estimate and expensive to measure, especially when dealing with large landscapes. Therefore, it is important to use erosion indicators and modeling to estimate potential soil loss (Rostagno & Degorgue 2011).

Many empirical models are proposed to predict soil erosion by water and associated sediment yield. Most of these models are not well tested and require several parameters, but they are used due to their simplicity (Mahmoodabadi 2011). These models are also limited to the specific site of origin since they were designed according to the correlation of multiple parameters performed using site-specific empirical data. Therefore, many researchers have tried to overcome these limitations by producing numerical models of erosion. Often, these models are classified as semi-quantitative models due to the combination of descriptive and quantitative procedures, which result in a quantitative or qualitative estimate for soil erosion and sedimentation (Mohamadiha et al. 2011). Geographical Information Systems (GIS) and remote sensing (RS) technologies can be innovative tools in the estimation of soil erosion. GIS and RS modeling are widely used for the preparation of variables required to estimate soil erosion, and they have the capability to analyze a large amount of data for arid and semi-arid landscapes (Amini et al. 2010; Ahmad & Verma 2013; Taheri et al. 2013).

The Universal Soil Loss Equation Method (USLE) is the most widely used empirical model in soil water erosion investigations due to its simplicity, though it was designed for agriculture practices (Harmon & Doe 2001). It is used for planning soil conservation measures, especially in developing countries (Breiby 2006; Csáfordi et al. 2012; Kamaludin et al. 2013; Meusburger et al. 2013). The major disadvantage of empirical models is that they are applicable only for the data base from which they have been derived. Therefore, in 1987 the USLE was modified to into the Revised Universal Soil Loss Equation (RUSLE) (Renard et al. 1997). The Erosion Potential Method (EPM) model is also an empirical model, which was developed for Yugoslavia for estimating the quantity and quality of soil erosion and sedimentation (Amiri 2010). This model was tested in several agriculture locations in Iran, as the output results of this model were compatible with field observations (Amini et al. 2010; Daneshvar & Bagherzadeh 2012). However, the Modified Pacific South West Inter Agency Committee (MPSIAC) model was designed for arid and semi-arid lands in the United States (Adib et al. 2012; Ilanloo 2012; Belete 2013). The original MPSIAC model was developed by Johnson & Gebhardt (1982). The newer enhanced MPSIAC model is more quantitative than earlier versions and its scoring is more realistic (Najm et al. 2013). This model has been used in several locations in Iran, and these studies also illustrate that the output was compatible with field observation (Rostamizad & Khanbabaei 2012; Shahzeidi et al. 2012; Taheri et al. 2013).

A few studies have compared the EMP and MPSIAC models, though in agricultural areas in Iran. The MPSIAC model showed more appropriate results when compared with EMP (Baqerzadeh-Karimi 1993; Bayat 1999; Taheri et al. 2013). Mahmoodabadi (2011) concluded that the MPSIAC model showed a maximum value for erosion and they stated that it needed modification. Eisazadeh et al. (2012) compared the MPSIAC and USLE, with results showing that both models had reasonable results, though the MPSIAC was the superior model. One issue with all these studies is that they were conducted for agricultural areas in Iran; none of them were tested or compared for native desert ecosystems. Therefore, our central objective of this chapter was to test and compare the MPSIAC, EMP, and USLE models for arid natural ecosystems. We used Umm Nigga, Kuwait as an example landscape, and discuss soil loss as a function of land degradation, management, and restoration for this region.

Materials and Methods

Study Area

Kuwait is located in Asia, has a total area of 17,820 km^2, and a latitude and longitude of 29.3286° N, 48.0034° E. Umm Nigga is situated on the northern edge of Kuwait with a total area of 246 km^2. The study area is distant from residential areas at approximately 50 km from Kuwait City. It is considered an open rangeland, which is used for camping and grazing, with several private farms in the northeastern section. Currently, restoration is being planned for the site and it has been selected as a future protected area. The De-Militarized Zone (DMZ) lies immediately north, and was created as a buffer between Kuwait and Iraq by the United Nations Security Council Resolution 689. The restoration area includes this area, but was further extended by the Kuwait Supreme Council for Environment through annexation. The DMZ extends along the Kuwait-Iraq border and the Khawr 'Abd Allah waterway; it is approximately 200 kilometers long, extending 10 kilometers into Iraq and 5 kilometers into Kuwait. It was also illustrated in chapter 2 that Umm Nigga contains a coastal ecosystem type, and two desert ecosystem types. The coastal area is covered with sabkha, salt marshes and saline depressions, sand dunes, and ridges and terraces. It is also covered with the *Halophyletum* vegetation community unit. The other two desert ecosystems are composed of four soil groups including *Calcigypsids* and *Haplocalcids* (sandy to loamy soils), and *Torripsamments* and *Calcigypsids* (mostly sand and gravel). *Haloxyletum* and *Rhanterietum* are the major vegetation units in the desert ecosystems.

Potential Soil Loss Estimation Using MPSIAC Model

The MPSIAC model requires nine variables: surface geology, topography, land cover, soil characteristics, climate (rainfall), runoff, land use, present erosion, and channel erosion. The channel erosion factor was excluded from our work, as the study area does not include any channels. GIS and remote sensing products were used to generate each variable. GIS data layers were collected from Kuwait University (KU) and Kuwait Institute of Scientific Research (KISR), these layers include geological map, soil survey, vegetation communities, elevation points, and contour lines. Geo-referenced Landsat imagery was also obtained from the United States Geological Survey (USGS). Then, each variable was generated and calculated individually based on the MPSIAC equations (Table 3.1). Finally, the soil erosion and sedimentation layer was estimated using the following equations:

$$\mathrm{Qs} = 38.77e^{0.0353R} \qquad \text{(Eq. 1)}$$

Where: Qs= total sediment yield in $m^3/km^2/yr$. e = 2.718, R is the sum of the effective factors. Following Bagherzadeh & Daneshvar (2013), the sediment delivery ratio (SDR) is obtained from the following equation:

$$SDR = (46.7 \times \frac{A}{2.58})^{-0.2071} \qquad \text{(Eq. 2)}$$

Where SDR is the sediment delivery ratio and A is the sub-basin surface area. SDR is defined as the ratio of sediment yield to total soil losses. The equation can be expressed in non-dimensional terms as:

$$SDR = \frac{SY}{T} \qquad \text{(Eq. 3)}$$

Where SY is the sediment yield ($m^3/km^2/yr$), and T is the total eroded soil loss ($m^3/km^2/yr$).

Table 3.1 Effective factors on soil erosion for the MPSIAC model

Effective factors	Equations	Parameters
Surface geology	Y1=X1	X1=geological erosion index
Soil	Y2=16.67X2	X2=soil erodibility factor
Climate	Y3=0.2X3	X3=6-hour rainfall with a 2-year return period
Runoff	Y4=0.006R+10Qp	Q_p= annual specific Debi (m^3/skm^2) R=annual runoff height (mm^3)
Topography	Y5=0.33X5	X5=percentage of the average basin slope
Vegetation	Y6=0.2X6	X6=percentage of land without vegetation
Land use	Y7=20-0.2X7	X7=percentage of vegetation cover
Surface erosion	Y8=0.25X8	X8=total surface soil factor scoring in BLM*
Channel erosion	Y9=1.67X9	X9=gully scoring in BLM*

Data Collection and Preparation

Surface Geology (y1)

The surface geology was obtained from the geological map of Kuwait and other previous studies. The study area was covered with Aeolian sand, Desert floor deposits, Dibdibah formation, Intertidal and shoaling sand, silt and Mud Sabkha deposits, and Strand line deposits (Al-Sulaimi & Al-Ruwaih 2004). Aeolian sands are mostly sandy with high infiltration rates and low runoff. The desert floor deposits were generated from slopes, and Dibdibah formation is a white fine-grained cherty limestone and sand and gravel, with high infiltration rates. In contrast, the Intertidal and shoaling sand, silt and Mud Sabkha deposits, and Strand line deposits are muddy and contains clay soils with low infiltration rates and high runoff rates (Abdal et al. 2002; Al-Sulaimi & Al-Ruwaih 2004). Based on these characteristics, each unit within the geological layer was ranked between 0 (less sensitivity to erosion) to 10 (high sensitivity to erosion) following the MPSIAC model (Fig. 3.1a).

Soil Factor (y2)

The soil factor was calculated using data from the soil survey of Kuwait (KISR 1999). The soil erodibility map was generated according to the RUSLE equation for soil erodibility (K factor) (Gitas et al. 2009; Benzer 2010; Dumas & Printemps 2010):

$$K = 2.8 \times 10^{-7} \times M^{1.4} (1.2 - a) + 4.3 \times 10^{-3} (\mathrm{b} - 2) + 3.3(c - 3) \qquad \text{(Eq. 4)}$$

Where M is the size of soil particles $(\%Silt + \%Very\ fine\ sand) \times (100 - \%Clay)$, *a* is the percentage of organic matter, *b* is the code number defining the soil structure (very fine granular = 1, fine granular = 2, coarse granular = 3, lattice or massive = 4), and *c* is the soil drainage class (fast = 1, fast to moderately fast = 2, moderately fast= 3, moderately fast to slow = 4, slow = 5, very slow = 6). Subsequently, the K factor was used to calculate the soil factor using MPSIAC model equation. The final score for the soil factor layer ranged from 1.5 to 7.05 (Fig. 3.1b).

Climate (y3)

The commonly used index of rainfall aggressiveness, which is significantly correlated with soil erosion, is the ratio of the highest mean monthly precipitation and the mean annual precipitation (Morgan 1976). Recent work suggests that elevation may also influence erosivity (Daly et al. 1994). Precipitation data were collected from meteorological records from the Kuwait Meteorological Center, Kuwait Airport Station.

The climatic factor rating was estimated based on 20 years (1990-2010). The same rating value was given to the entire location since there is no variation with rainfall around the study area.

Runoff (y4)

Surface runoff is a major factor influencing soil erosion. This factor was generated using the Soil Conservation Service Curve Number Equation (SCS-CNE) model, which was developed in the mid-1950s (Mockus 1964; Beven 2011). This model is widely used as a simple method for predicting direct runoff volume for a given rainfall event. This model requires rainfall data and soil data including potential maximum retention, soil moisture retention, and infiltration rates. An empirical relationship estimates initial abstraction and runoff as a function of soil type and land use. The rainfall-runoff relationship was calculated using the following equations:

$$Q = (P - I_a)^2 \ / \ (P - I_a) + S \qquad \text{(Eq. 5)}$$

Where, Q = runoff (in)

P = rainfall (in)

S = potential maximum retention after runoff begins (in) and

I_a = initial abstraction (in)

Initial abstraction (I_a) includes water retained in surface depressions, water intercepted by vegetation, evaporation, and infiltration. It is also correlated with soil and cover parameters, and was found to be 20% of the potential maximum retention (S) (Ghadiri & Rose 1992). By assuming that the initial abstraction is equal to 20% of potential maximum retention (I_a □□0.2S), the above equation can be simplified to:

$$Q = (P - 0.2S)^2 \ / \ (P - 0.2S) \quad \text{(Eq. 6)}$$

Where S is related to the soil and cover conditions through the Curve Number (CN). CN has a range of 0 to 100, and S is derived from CN by:

$$S = 1000/CN - 10 \quad \text{(Eq.7)}$$

The runoff curve number (CN) parameter values correspond to various soil, land cover, and land management conditions and can be selected from model tables. However, it is preferable to estimate the CN value from measured rainfall and runoff data if available (Soulis & Valiantzas 2012). Here, the CN value was estimated using soil survey of Kuwait (KISR 1999), and the Natural Resources Conservation Service (NRCS) curve number, which divides soils into four hydrologic soil groups (HSGs) based on infiltration rates (Ghadiri & Rose 1992). Soil infiltration data were used to estimate HSGs, which were combined with the land cover factor (y7) to estimate CN. Then, potential maximum retention (S) was calculated from the CN value using Eq. 6, and the potential maximum retention was used in Eq. 7 to estimate runoff (Q). The scoring values ranged from 0.66 to 3.43 (Fig. 3.1c).

Topography (y5)

The topography factor was generated using elevation contour lines and spot elevation points to create a raster Digital Elevation Model (DEM). Percentage slopes were derived from the DEM using GIS, and were used in the MPSIAC equation to compute the scoring value, which ranged from 0 to 0.56 (Fig. 3.1d).

Vegetation Cover and Land Use (y 6 & y7)

Geo-referenced Landsat 8 imagery was obtained from the United States Geological Survey (USGS) for the year 2013 to create a land use and vegetation cover layer. The spatial resolution of the imagery was 30 x 30 m, and the projection was WGS 84 UTM Zone 38N. No atmospheric and geometric corrections were necessary for this region due to the low cloud cover. Supervised classification was used in this study using ENVI 5.2 to identify the land cover; methods are described in detail in chapter 2. The imagery was divided into five land cover types: soil, bare ground, vegetation, wetlands, and water. Then, vegetation cover and land use layers were combined in one layer and were scored as ranks using the MPSIAC equation. The final ranking ranged from 0.046 (for locations covered with vegetation) to 20 (for bare ground locations) (Fig. 3.1e).

Surface Erosion (y8)

The surface erosion was estimated based on the surface soil erosion types using Bureau of Land Management (BLM) method (Ypsilantis 2011). The study area includes sheet erosion (as occurs when rain falls on bare or sparsely covered soil), and some rill erosion (as occurs on slopes and streams). Each erosion type was rated from 0 (low sensitivity) to 15 (high sensitivity) based on their level of degree of sensitivity. The layers that were taken into consideration to create surface erosion are streams, land cover, and slopes. Finally, all three factors were combined to establish the surface erosion layer using the MPSIAC equation. The final ranking layer ranged from 1.25 (low sensitivity to erosion) to 6.25 (high sensitivity to erosion) (Fig. 3.1f).

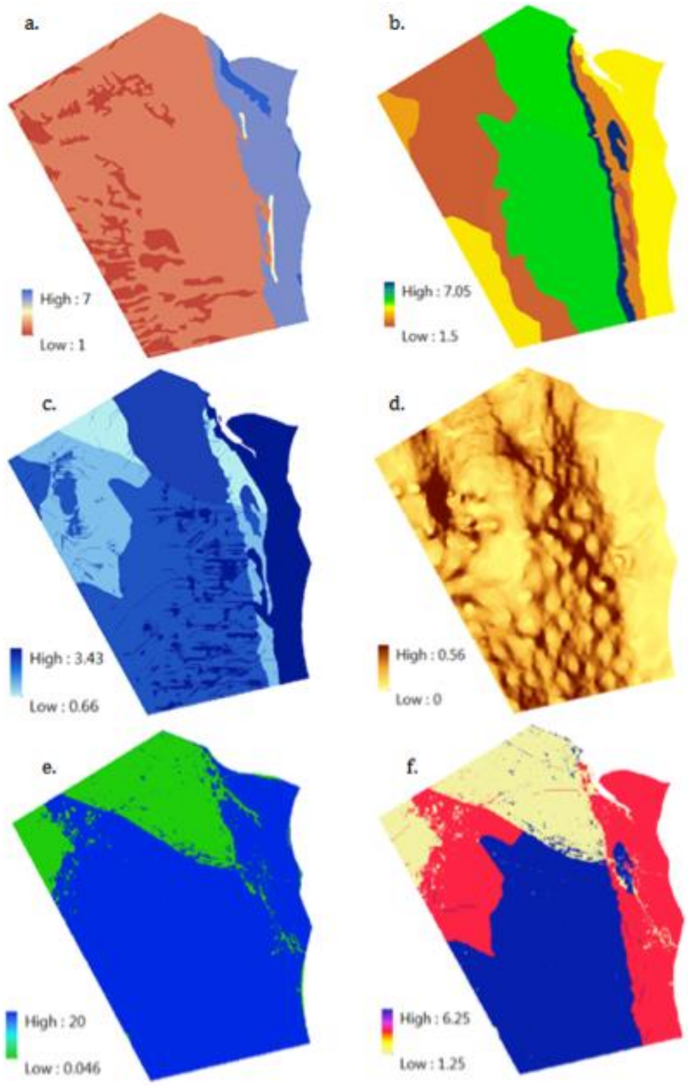

Fig 3.1 MPSIAC model variables: (a) Surface geology (y1), (b) Soils (y2), (c) Runoff (y4), (d) Topography (y5), (e) Vegetation and land use (y6 & y7), (f) Surface erosion (y8).

MPSIAC Model versus RUSLE and EMP Models

The MPSIAC model was compared with the EMP and RUSLE models. The majority of the data layers for the MPSIAC model were also used in this step, but the coefficient for the variables were rated and scored according to their respective EMP or RUSLE equations.

EMP Model

Soil erosion in the Erosion Potential Method (EMP) model is based on the following four factors:

Y: The coefficient of rock and soil erosion, ranging from 0.25- 2

Xa: The land use coefficient, ranging from 0.05- 1

Ψ: The coefficient for present erosion type, ranging from 0.1- 1

I: Average land slope in percentage

The necessary layers and data for these factors were: geology and soil types, land use, slope, and erosion type. The required data and GIS layers were the same as in the MPSIAC model, but they were rated based on the EMP coefficient rating (Fig. 3.2). The Erosion Potential Method (EPM) calculates the coefficient of erosion and sediment yield (Z) of an area using the following equation:

$$Z = Y \times Xa\ (\Psi + I^{0.5}) \qquad \text{(Eq. 8)}$$

In which Y is the coefficient of rock and soil erosion, Xa is the land use coefficient, Ψ is the coefficient for the present erosion type, and I is the average land slope in terms of percentage.

Then, the volume of soil erosion was calculated using the following equation:

$$W_{sp} = T \times H \times \pi \times Z^{1.5} \quad \text{(Eq. 9)}$$

In which, W_{SP} is the volume of soil erosion (m^3/ km^2/ yr), H is annual rainfall (mm), Z is erosion intensity and T is coefficient of temperature which is calculated as shown below:

$$T = (t/10 + 0.1)^{0.5} \quad \text{(Eq. 10)}$$

Where t is the mean annual temperature (°C).

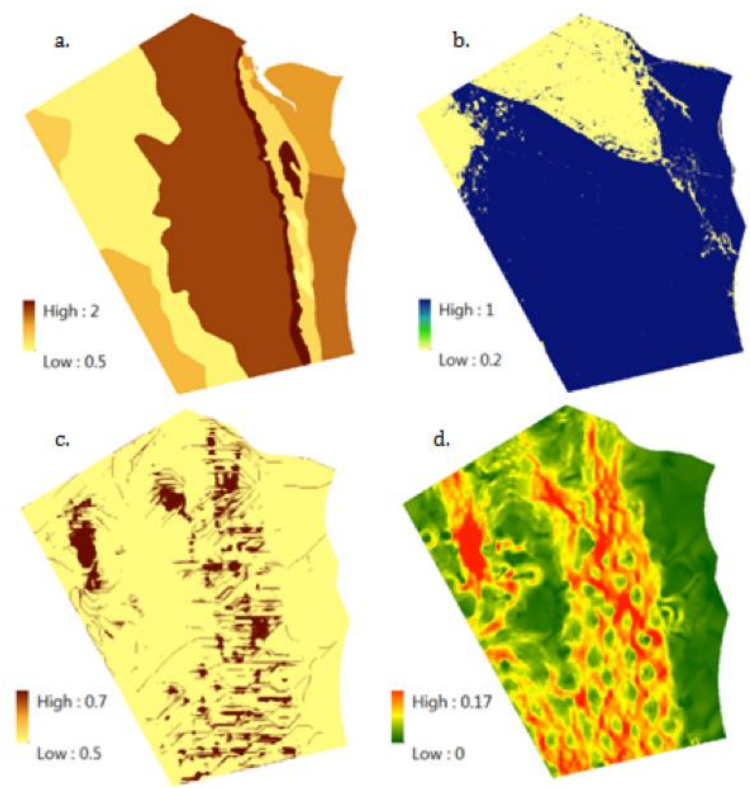

Fig 3.2 EMP model variables: (a) Coefficient of rock and soil erosion (Y), (b) Land use coefficient (Xa), (c) Coefficient for present erosion type (Ψ), (d) Average- land slope in percentage (I).

RUSLE Model

The RUSLE is the most common used model, as it is considered the most simplistic model for estimating soil erosion from water. This model covers five variables as shown in the following equation:

$$A = R \times K \times LS \times C \times P \qquad \text{(Eq. 11)}$$

Where,

A = predicted soil loss (tons/ acre/ year)

R = rainfall and runoff factor

K = soil erodibility factor

LS = slope factor (length and steepness)

C = crop and cover management factor

P = conservation practice factor

The same vegetation cover and soil erodibility layers (Fig. 3.3a-b) that were generated for the MPSIAC model were used. An annual rainfall of 129 mm was defined for the entire location. However, the P factor was discounted to 1 because there were no conservation practices in the study area. The slope length and steepness (LS) factor was generated (Fig. 3.3c), using the DEM and the following equation:

$$LS = (\text{Flow accumulation} * \text{Cell value} /22.1)^{m} (0.065 + 0.045\, s + 0.0065\, s^{2}) \qquad \text{(Eq. 12)}$$

Where LS is slope length and steepness and s, is the slope percentage, and m is a variable plot exponent adjustable to match terrain and soil variants.

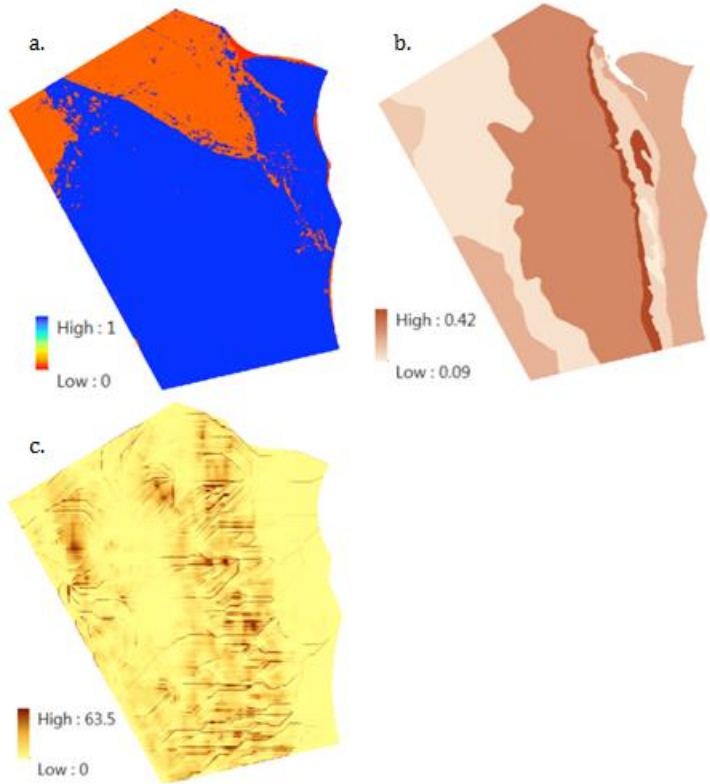

Fig 3.3 RUSLE model variables: (a) Land cover (C), (b) Soil erodibility factor (K), (c) Slope length and steepness (LS).

Model Comparison and Testing

To compare the models, results of potential soil loss maps were classified into five ranked classes (which ranged from very low to very high) using GIS. Maps were converted to grid files and analyzed using FRAGSTATS 4.2 to compute a set of class matrices. Total Area CA (how much of the class is comprised of a particular patch type), Percentage of Landscape (PLAND) (quantifies the proportional abundance of each patch type in the class), Patch Number (NP), Patch Density (PD), and Patch Area Distribution (PAD) were selected to provide information on class area and number. The shape index (measures the complexity of patch shape compared to a standard shape) was also used to measure the shape complexity for each class. Then the aggregation index (AI) (the percentage of like adjacencies between cells of the same patch type) was used to analyze patch connectivity within the classes.

Sensitivity analysis was also conducted to evaluate the soil erosion models response to changes in input. Sensitivity analysis is a technique for evaluation and calibration models, which, helps to understand the influence of input data on output. For this study we used the sensitivity analysis method that was designed by Lane & Ferreira (1980). Input data variables were increased by 20% with the aim of calculating Qs and variation of erosion. Sensitivity for the main factors for each model were calculated using the following equation:

$$SI = ((Qs - Qsa)/Qsa)/((P - Pa)/Pa) \qquad \text{(Eq.13)}$$

Where: SI is parameter sensitivity indices, Pa= is initial first parameter, Qsa = is calculated sediment using Pa, P = is increased or decreased input data, and Qs = Is calculated sediment using P. Sensitivity index was calculated using Excel.

In addition, a simulation was conducted to assess the effects of increasing vegetation cover on soil loss, as might result from a change in management practices as part of a major restoration effort (for example, fencing to prevent overgrazing or camping). As discussed in chapter 2, the vegetation cover was relatively high at the unfenced area in 1998 (37% in unfenced areas), but then decreased to 3% by 2013, due to overgrazing and spring camping by people; these management practices accelerated vegetation loss after land mines were removed from the area. To simulate the potential vegetation cover after restoration, satellite imagery for the year 1998 was classified and the model was re-run using this as input for y6 and y7 in Eq. 1 and 2.

Results

Potential Soil Loss

MPSIAC Model

A soil erosion risk map was generated based on the attributes of the nine variables and the given scores by the MPSIAC model (Fig. 3.4a). Modeled soil loss varied from 129 to 1184 m3/km^2/yr, which was categorized into five classes ranging from very low to very high. Approximately 24% of the total area ranged between low-very low potential soil loss; of that 18 % of the surface was moderately and 58% was high-very high. The estimated soil loss varied between coastal and desert areas (Table 3.2) and was higher at the desert area. At the coastal area, the potential soil loss was high, and the amount of erosion was similar between the DMZ (fenced) and unfenced sites. The average soil loss was 570 m^3/km^2/yr for the fenced and 523 m^3/km^2/yr for unfenced (Fig. 3.5a). The high soil erosion levels at the coastal area were likely due to natural geomorphic changes, as opposed to grazing and camping, as these activities are not typically conducted along the tidal flat due to the muddiness, high salinity, and difficulty of access. The erosion rate was still higher at the desert areas, and it varied substantially between unfenced (high to very high, avg. = 703 m^3/km^2/yr) and fenced DMZ sites (very low to low, avg. = 313 m^3/km^2/yr) (Fig. 3.6a). Vegetation cover greatly influenced the modeled erosion for the desert area unfenced (3% vegetated surface in unfenced versus 88% in fenced).

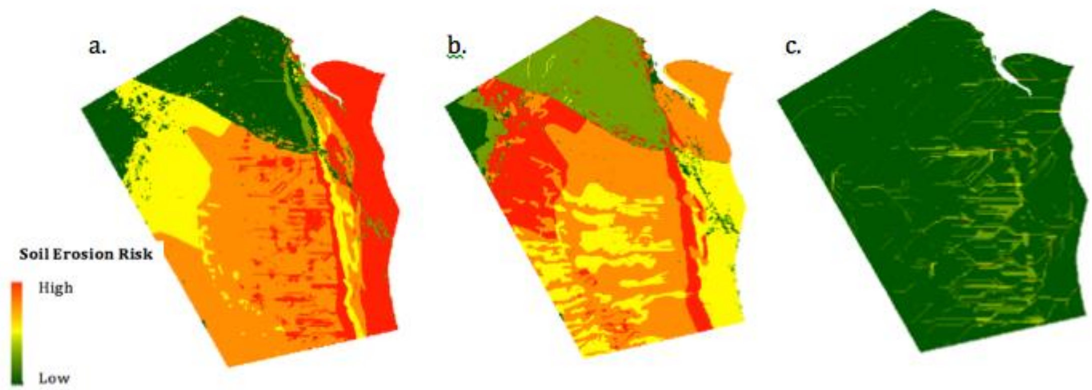

Fig 3.4 Potential soil loss map (a) MPSIAC model, (b) EMP model, (c) RUSLE model.

Table 3.2 Average annual coefficient for effective factors and soil loss for MPSIAC model

Effective factors	Geology	Soil	Climate	Runoff	Topography	Land use	Erosion Type	Total score	Soil loss $m^3/km^2/yr$
Coastal	3.77	2.76	4.72	3.06	0.03	19.99	2.9	27.2	470
Coastal (fenced)	2.9	2.82	4.72	2.62	0.3	17.25	3.1	36.1	423
Terrestrial	1.4	2.06	4.72	2.61	0.14	18.21	4.2	31.6	703
Terrestrial (fenced)	1.8	3.3	4.72	2.44	0.12	0.046	1.25	13.8	310

Sensitivity analysis indices showed that the vegetation cover (y6 and y7) was the most influential factor to the final output with the highest sensitivity index (0.569). Surface geology (y1) came next with an index of 0.281, soil factor (y2) had a sensitivity index of 0.143 and runoff (y4) had 0.093. However, sheet erosion was the most common erosion type (y8) in the study area, and it was less influential on the output (0.021) as compared with other possible erosion types such as gully erosion. Topography (y5) only slightly affected the output (0.0003), as the study area is mostly flat (Fig 3.7). The simulation showed that by increasing vegetation cover from 3 to 37% at the unfenced area, soil loss could decrease from 703 $m^3/km^2/yr$ to 478 $m^3/km^2/yr$ (Fig. 3.6).

EMP Model

The calculated soil loss for the EMP model varied from 9 to 1,252 m3/km2/yr, and was also classified from very low to very high (Fig 3.4b). Approximately 65% of the total area ranged between low- very low potential soil losses, 12% of the surface was moderately, and 19% was high-very high. Also similar to MPSIAC, the EMP-based erosion was high at the desert area, with large differences between fenced (223 m3/km2/yr) and unfenced sites (1051 m3/km2/yr) (Table 3.3). The potential soil loss at the coastal area was almost similar between the fenced and unfenced sites (Fig. 3.6b). At the coastal area, the degree of soil loss ranged from moderate to high. The average soil loss was 682 m3/km2/yr at the unfenced and 587 m3/km2/yr at the fenced site. Vegetation cover highly influenced the modeled erosion at the fenced and at some parts of the unfenced area, as well as, soil types also influenced the model as areas with clay soils were less affected compared with sandy soils. It was also seen that erosion rates were not high at the coastal area, which was mostly considered moderate.

Sensitivity analysis (Fig 3.7b) showed that land use and vegetation cover (Xa) highly influenced the EMP model output. Also similar, the simulated increase in vegetation decreased the rates of soil erosion at the fenced site. Geology and soils (Y) came next; slopes (I) did not influence the model output as the study area is considered a flat area with slops ranges from 0-5%.

Table 3.3 Average coefficient for effective factors and soil loss for EMP model

Effective factor	The coefficient of rock and soil (Y)	The land use coefficient (Xa)	The coefficient for present erosion type (Ψ)	Average- land slope in percentage (I)	Z	Soil loss $m^3/km^2/yr$
Coastal	1.4	1	0.5	0.1	1.07	682
Coastal Fenced	1.4	0.8	0.5	0.9	0.9	587
Terrestrial	1.2	0.9	0.56	0.42	1.31	1051
Terrestrial Fenced	1.3	0.2	0.56	3.37	0.4	223

RUSLE Model

The RUSLE showed different results compared with MPSIAC and EMP (Fig 3.4c). Around 94% of the total area ranged from very low- low erosion rate and 6% ranged from moderate to very high. This model did not show any variation between the classes, and 94% of the total areas were concentrated in the low erosion zone. The degree of soil loss was also similar between the fenced (4.38 ton/acres/yr) and unfenced area (5.31 ton/acres/yr) at the coastal site (Fig. 3.6c), but some differences were seen at the desert fenced versus unfenced sites, which had an average of 98.2 ton/acres/yr for the fenced and 21.66 ton/acres/yr for the unfenced area (Table 3.4). The results also showed that each of the four variables had similar influence on the output (Fig. 3.7c).

Table 3.4 Average coefficient for effective factors and soil loss for RUSLE model

Effective factors	K factor	LS factor	C factor	R factor	Soil loss ton/acre/yr
Coastal	0.17	0.49	1	60	5.31
Coastal Fenced	0.16	0.40	0.89	60	4.38
Terrestrial	0.18	7.2	0.9	60	98.2
Terrestrial Fenced	0.18	9.25	0.3	60	21.66

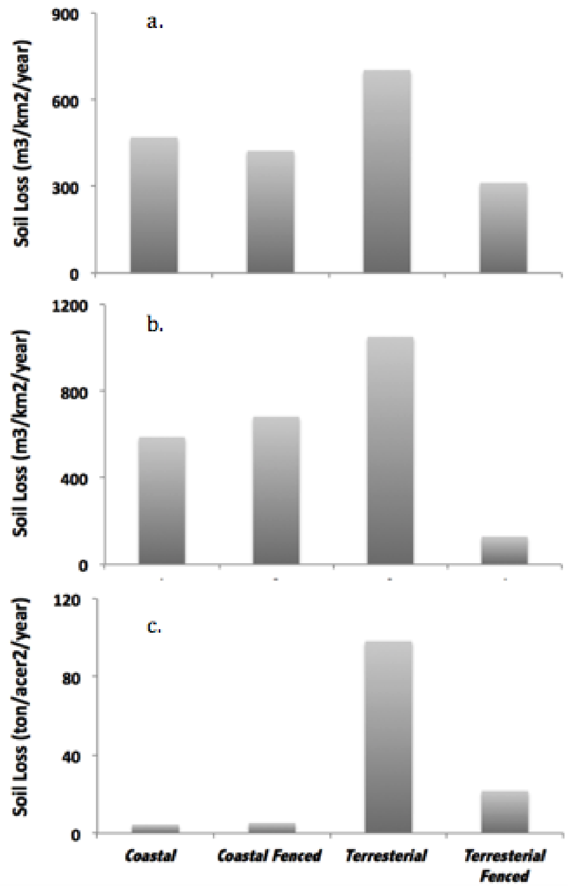

Fig 3.5 Comparison between soil loss for coastal and desert areas (a) MPSIAC, (b) EMP, (c) RUSLE.

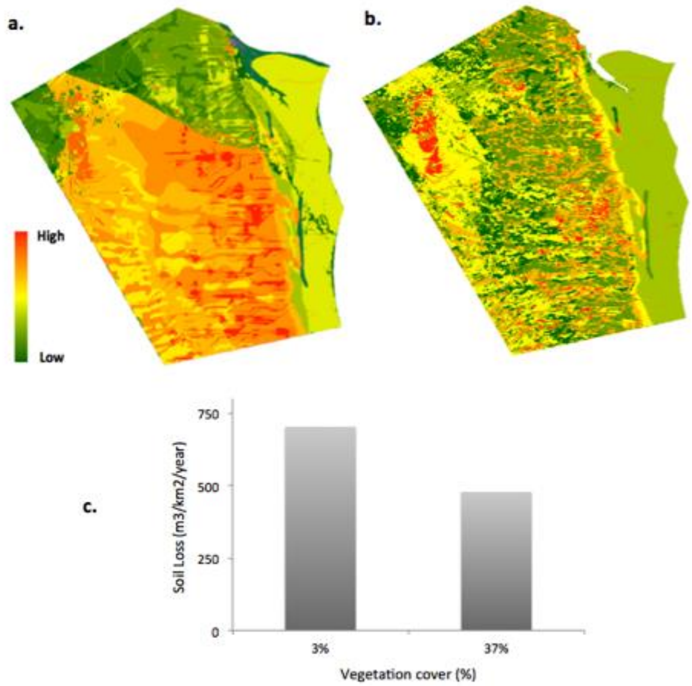

Fig 3.6 Vegetation cover simulation. (a) Soil erosion map with 3% vegetation cover at the unfenced area, (b) Soil erosion map after increasing vegetation cover at the unfenced area to 37%, (c) Potential soil loss decreased from 703 $m^3/km^2/yr$ to 478 $m^3/km^2/yr$ with increase in vegetation cover.

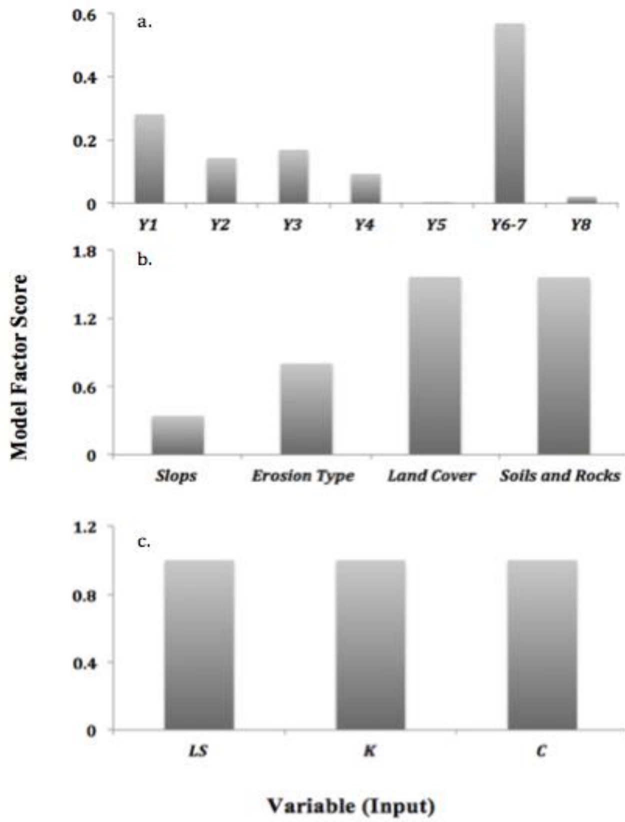

Fig 3.7 Results of sensitivity analysis for input variables, (a) MPSIAC, (b) EMP, (c) RUSLE. Higher values indicate a higher sensitivity to an input parameter.

FRAGSTATS Class Metrics Analysis for Empirical Models

Results of FRAGSTATS class metrics showed variation between the three models. The total area for each class was somewhat consistent for the MPSIAC model as 39% of the total area was considered as high potential soil loss, 3% were considered low, but the remaining classes were almost similar at around 20%. The total area for the EMP and RUSLE classes differed more greatly in general, with around 65% of total area of the EMP model was considered very low to low, and more than 90 % of the total area at the RUSLE was considered very low to low (Fig 3.8a).

The MPSIAC model also had the highest patch number and density, and was relatively consistent among the five erosion levels. However, patch number and density varied more greatly between classes in the EMP and RUSLE. The EMP model had a higher patch density among all classes as compared with the RUSLE, except at the moderate erosion level, which was higher for the USLE (Figs 3.8b-c).

The results also showed that the erosion classes in the MPASIC model were more evenly distributed within the five classes, since the patch area distribution was fairly consistent among the five classes. However, the classes within the EMP and RUSLE were mostly concentrated at the low erosion level (Fig 3.8d). The MPSIAC and EMP model had similar shape index values among, the five classes though with a slightly higher value at the very high level of erosion, which illustrates that all classes had the same shape complexity. However, the shape index varied much more strongly with RUSLE across the classes, though showing the same generally increasing pattern among the classes (Fig 3.8e). Overall, the FRAGSTATS results illustrate that MPSIAC model produced output with more evenly distributed classes of erosion, yet within these classes there were more individual patches and a greater density of them, suggesting that the MPSIAC results were more finely-detailed as compared with the other models.

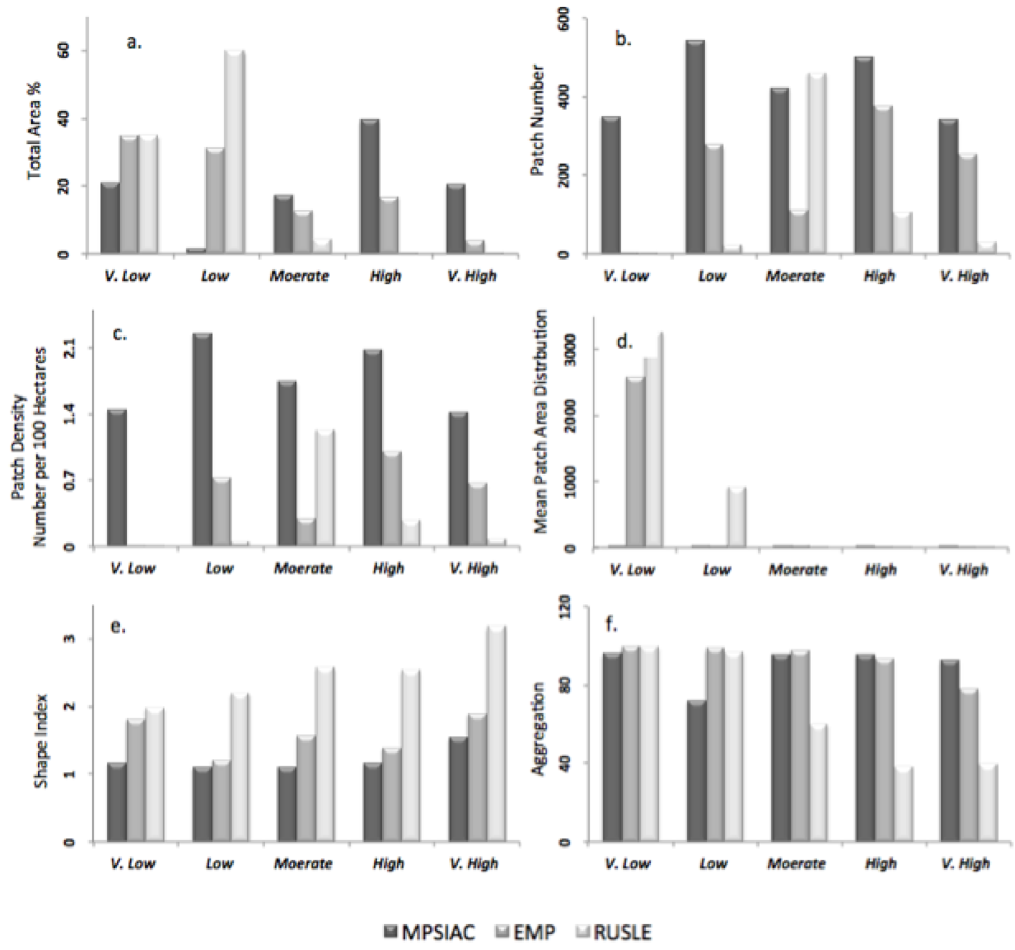

Fig 3.8 Class matrix analysis. (a) Total area, (b) Patch number, (c) Patch density, (d) Mean patch area distribution, (e) Shape index, (f) Aggregation percentage.

Discussion

Response of Soil Erosion Models

The MPASIC and EMP models produced somewhat similar results, but the MPSIAC model presented more logical and well-resolved spatial results. The soil factor was more effective in the MPSIAC model; it showed that erosion rates were higher at the coastal area, which is indeed the case as it is covered with clay soils with low infiltration rate and high runoff rates. Also in the MPSIAC, the desert areas with low vegetation fell within the high erosion risk class. In contrast, the EMP model produced a low erosion rate in some parts of the desert area, especially those that were covered with sandy to loam soils with a high runoff rate, which could be unrealistic, especially with the absence of vegetation cover. The MPSIAC model also presented better spatial detail, with a higher patch number and density, and higher evenness across all classes for the various FRAGSTATS metrics, when compared with EMP and RUSLE. One reason for this result is likely that the higher number of input variables covers a higher number of independent erosional processes. For these reasons, MPSIAC model should be considered the superior model to assess and map soil erosion in arid regions such as Umm Nigga. Moreover, the results calculated by the MPSIAC model are in better accordance with those of the studies of Renard et al. (1997) and Rahmani et al. (2004).

The RUSLE presented unrealistic results, as all four factors had the same sensitivity, but moreover the locations with high erosion risks were most strongly correlated with slope length and steepness – an unrealistic result since the study area is a flat open landscape. This model was designed for agricultural areas in USA and so this conclusion should not be surprising. For these reasons, the RUSLE was deemed an unsuitable model for the Umm Nigga study area, and likely for other arid lands such as those found in Kuwait.

Why Do the Model Responses Differ?

Our study showed that the MPSIAC model is likely the superior model, when compared with EMP and RUSLE models. However, this may not always be the case, as it depends on the region and condition in which the model was developed. Empirical models are based on the determination of the significant relationship between model input and model output. The realistic response of the MPSIAC model in our study is likely due to the fact that model was designed for arid and semiarid lands in the United States (Bagherzadeh & Daneshvar 2013). However, this does not mean that RUSLE is always unrealistic, as it showed reasonable results when applied for forest regions with high slop percentage (Terranova et al. 2009; Csáfordi et al. 2012) and agriculture areas (Angima et al. 2003; Fu et al. 2006; Meusburger et al. 2013).

Differences among model outputs may also be due to the erosional processes dominant at different spatial and temporal scales, with each representing a somewhat different mix of erosional processes. Models that were designed for different regions differ in the mix of erosional process they aim to model. In addition, most empirical models lump a number of processes together and describe them as a signal mathematical or logical relationship, for example the older USLE. The advantages of such models are their simplicity in term of data requirements and computation (Harmon & Doe 2001). However, the individual modeled processes cannot be disaggregated or changed, which is a problem if the model was designed for a different location or spatial scale. Moreover, empirical relationships are often calibrated for a particular dataset that is only valid for the dataset in which they were derived from (Ghadiri & Rose 1992; Hudson

1993). The ideal approach is for each country or region to have their own model, for example the Soil Loss Estimation Model for Southern Africa (SLEMSA) and the European Soil Erosion Model (EUROSEM) (Hudson 1993).

Factor models are considered empirical models in that the variables are represented by a quantified factor and are combined together by adding them up or multiplying them together (Hudson 1993). The MPSIAC and RUSLE could be considered as factor models since the scoring of each factor is created based on equations, then the scores are used in the final equation to predict the amount of soil loss. However, the EMP model depends on tables when selecting a coefficient score, and then these scores are added in an equation to calculate the amount of soil loss. Moreover, the MPASIC model contains the highest number of factors influencing the erosion processes.

Can Native Vegetation Control Soil Erosion?

Our results indicated that vegetation cover plays an important role in controlling soil erosion. In the MPSIAC model, the erosion was most sensitive to this factor, as were demonstrated by the difference between the fenced DMZ and unfenced portion of the desert areas. The fenced area ranged between low to very low soil loss, as compared with high to very high at the unfenced area. Previous studies have similarly concluded that vegetation is a major driver for the MPSIAC model (BehnamA et al. 2011; Ilanloo 2012). For this reason, it is likely important to restore the vegetation in the desert unfenced areas. Somewhat conversely, the high amount of erosion that occurred in the coastal areas could be considered natural and thus re-vegetation is not a relevant or likely outcome.

Limitations

Judgments on how well the models perform are usually made by comparing the output with observed data from the field (Harmon & Doe 2001). Since we did not measure soil erosion in the field or lab, we are unable to judge the accuracy of the potential soil loss values for each model. Direct field measurements of surface soil erosion will be required to confirm the results of our model evaluation work. With this verification in mind, it might become necessary to modify or calibrate the MPSIAC model in order to get more accurate results. Lal (1994) discussed the critical nature of continuous simulation modeling in predicting erosion reliably, stating that long-term continuous simulation may be needed in order to quantify erosional responses within 10% of field values.

For Kuwait, it will be important to provide further calibration between winter storms and summer storm conditions. Also, calibration for bare soils may also not be applicable for mature crop stands (Harmon & Doe 2001). Models cannot fully represent all details in the natural world, but simultaneously it is not possible to use field samples only to quantify and map soil erosion across a large area, by making assumptions that the landscape is homogenous between each sample. Therefore, models are a critical tool in estimating soil erosion.

Conclusion

The MPSIAC model was the superior model for our study site, and when combined with the findings of other authors, suggests that arid regions should avoid use of the EMP and RUSLE when possible. The MPSIAC produced the most even and detailed results, likely because of the greater number of modeled factors that represent the various mechanisms that affect soil erosion. For all of the models, vegetation (ideally native plants) played an important role in decreasing the amount of soil erosion and controlling desertification. Thus, we suggest restoring the unfenced areas at Umm Nigga, Kuwait, by restricted grazing and access, and by protecting native plant species. Practices that limit vegetation loss could potentially lower soil erosion by 32%, as shown by our results. Moreover, the output maps generated by this study could be used to select suitable locations for re-vegetation efforts as based on the rated erosion rates or compounding factors mapped by each independent input factor. In summary, the MPSIAC spatial model is a useful predictive tool for estimating soil erosion across large-extent, arid landscapes.

CHAPTER IV

WILL AUTOGENIC SUCCESSION BE SUFFICIENT TO RECOVER FROM VEGETATION COVER LOSS OR WILL SOIL CONDITION NEED TO BE ADDRESSED IN THE ARID LANDS OF KUWAIT

Overview

Intervention is often required for the restoration of damaged arid ecosystems, particularly when the base environmental conditions are no longer suitable for autogenic recovery. Umm Nigga, in the northeastern portion of Kuwait, was damaged by overgrazing and destructive camping, following de-mining operations that occurred shortly after the Iraq War in 1990/1991. For Umm Nigga, it is unclear whether its restoration will require remediation of the soil conditions, or whether autogenic succession can occur once the area is fenced and released from this pressure. Thus, the central objective of this chapter was to design a conceptual strategy for selecting the required restoration actions. Our specific objectives were to assess the soil condition at the site and determine suitable locations for re-vegetation using GIS. We collected soil samples within each ecosystem and the vegetation was assessed using Braun-Blanquet cover-abundance scale. We also used GIS models to select locations for planting native species by seeds and seedlings. Our results showed that the vegetation in the coastal portions of the ecosystem was not damaged. However, in the desert ecosystem locations, phosphorus, potassium, and organic matter were higher in the reference area soils, and correlated with the higher vegetation cover. We conclude that soil remediation and planting/seeding are likely not necessary to restore the damaged sites in any ecosystem type at Umm Nigga, given that each still contains sufficient concentrations of nutrients to support native desert plants that are adapted to these harsh conditions. We also conclude that the introduction of fencing will likely release the ecosystem from the grazing disturbance and allow autogenic recovery. With these sites as a model, a conceptual framework is presented for arid ecosystem assessment and restoration planning.

Introduction

Ecosystem restoration is now globally recognized as a key component in conservation programs and essential to long-term sustainability in arid and semi-arid lands (Aronson & Alexander 2013). Overgrazing, and the resulting desertification and soil compaction, can dramatically influence these ecosystems (Perrow & Davy 2002). Intervention is often required for the restoration of damaged ecosystems in arid lands as their recovery via natural processes may take centuries (Bainbridge 2007). Autogenic succession can be limited in arid ecosystems, and the base environmental conditions must often be remediated before biota can survive due to the low amount of nutrients, which are concentrated in the topsoil and can thus be washed away easily. Soil damage alters stability, hydrological (Price 2011; Le Maitre et al. 2014; Melesse & Abtew 2015), and biological (Gobat et al. 2004; Hillel 2007; Plaster 2013) processes in arid regions. Thus, soil quality is a major factor that must be addressed before deciding how, when, and to what extent restoration activities must occur (Whisenant 1999; Bainbridge 2007).

There are several restoration activities, which include improving management strategies, repairing soil properties, repairing hydrological and nutrient cycling, controlling soil erosion, and re-vegetation (Whisenant 1999; Bainbridge 2007). Some studies argue that if natural recovery can occur, it would be the best to go with natural recovery, which is cheaper and more likely to succeed than intervention (Holl & Aide 2011). Others argue that in some cases, natural recovery is not practical and planting is necessary as natural recovery may lead to the appearance of new plants that were not existed in the past (Reinecke et al. 2008). As there is no single approach that is universally applicable, selecting a suitable approach requires a good assessment of the disturbed site (Cooke & Johnson 2002). Assessing the disturbed ecosystem can include identifying relevant hydrological processes (infiltration and runoff), and nutrient cycling to develop an ecological description that can then be used as a focal point for restoration efforts (Whisenant 1999; Tongway & Ludwig 2011).

Additionally, it is necessary to fully understand the history of the location, which helps in understanding what we are seeing today within the context of the past. A site history can guide our choice of assessment tools, and inform our views in determining the best restoration strategy. It also reduces the cost of field assessment or sampling, and determine the key sources of disturbance (Bainbridge 2007). Understanding the history can also help in identifying appropriate reference sites. There are several examples of successful landscape-scale restoration projects that utilized an assessment of the to select an appropriate restoration plan (Van Andel & Aronson 2012). For example, re-vegetation projects using seeds can fail due to seeding unsuitable sites, seeding at the wrong time, or inadequate site preparation (Whisenant 1999). Such failure in projects could be avoided if the history and site condition were well assessed beforehand.

In Kuwait, the loss of soil fertility and desertification (Omar 2014; Brown 2003) can be linked in part to the military activites of the first Gulf War in 1990/1991. These activities damaged the soil via oil spills (Abuelgasim et al. 1999; El- Gamily 2007), surface compaction, reduction in soil infiltration capacity, and loss of habitat (Misak et al. 2002; Al- Awadhi et al. 2005). Accordingly, the State of Kuwait established four terrestrial protected areas, using compensation funds as designated by the United Nations Compensation Commission (UNCC). Scientists at the Public Authority of Agriculture and Fisheries (PAAF) in Kuwait subsequently collected native seeds from other locations for use in the re-vegetation of these sites. Today, plans are being developed for planting vegetation and the use of irrigation (UNCC 2002) across very large areas, but it is not known if soil remediation activities will be first required. Umm Nigga, which is one of the designated protected areas in Kuwait, was selected as a study area to design a concept strategy and identify a restoration plan. Our specific objectives were to assess the soil condition at the site, to determine suitable locations for re-vegetation using GIS, and help guide decision-makers to define a restoration plan.

Materials and Methods

Study Area and Experimental Design

Kuwait is located in Asia, and has a total area of 17,820 km^2. Umm Nigga is situated on the northern edge of Kuwait with a total area of 246 km^2. It is somewhat rural, generally unpopulated, and approximately 50 km from Kuwait City. It is considered an open rangeland, which is used for intensive camping and grazing, with several private farms in the northeastern section. Currently, restoration is being planned for the site and it has been selected as a future protected area. The De-Militarized Zone (DMZ) lies immediately north between Kuwait and Iraq, and has been fenced since 1994 (Fig 4.1a).

The study area is considered a large landscape, which covers more than one ecosystem. In chapter 2, we divided the study area into three units according to ecological aspects such as geology, soil characteristics, and vegetation communities using a Geographic Information System. The first unit is considered a coastal ecosystem, covered with sabkha, salt marshes and saline depressions, sand dunes, and ridges and terraces. The major vegetation community is *Halophyletum*, and soils are mostly clay soils. The other two units are considered desert ecosystems, unit 2 (desert 1) is mostly covered with *Calcigypsids* and *Haplocalcids* soil groups (sandy to loamy soils), and *Haloxyletum* is the dominant vegetation in the community. The third unit (desert 2) is also considered a desert ecosystem, and is primarily covered with *Torripsamments* and *Calcigypsids* soil groups (mostly sand and gravel), and *Rhanterietum* is the dominant vegetation in the community. The DMZ was selected as a reference site such that it matched the damaged site in terms of the presence of the same three units (Fig 4.1b).

The study area was relatively large (283 km^2, with 222 km^2 of that total distributed in the damaged site and 60 km^2 in the reference site). Six plots (80 acres each) were selected in order to compare soil properties and vegetation. Three plots were selected at the damaged sites, with three matching plots at the reference sites (control plots). Each plot was placed randomly within the spatial extent of each of the six units, damaged/reference combinations (Fig 4.1b). Then, 15 points were selected within each plot using an accepted systematic sampling plan (Carter 1993; Doran & Jones 1996), composed of three parallel lines spaced by 200 m, with five points per line spaced by 100 m. (Fig 4.1c).

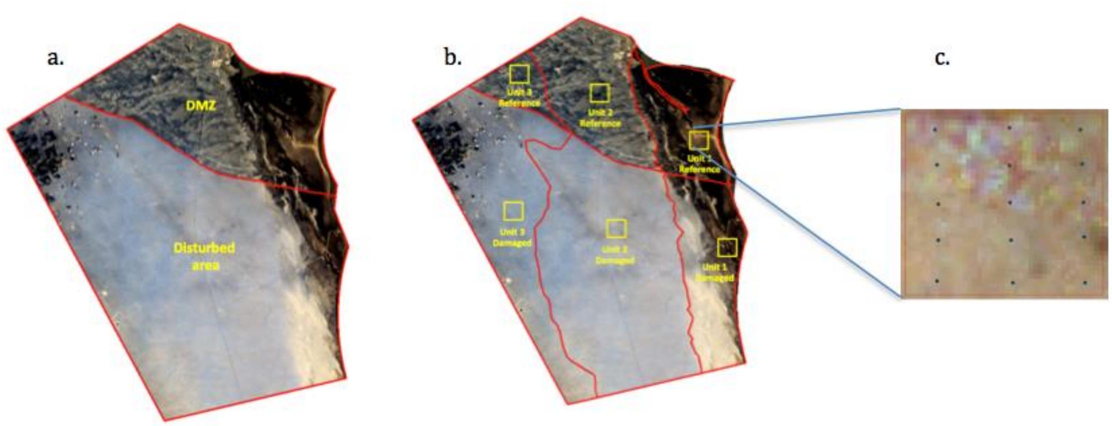

Fig 4.1 (a) The study area including DMZ and disturbed site, (b) distribution of the six plots within the three major units, including disturbed and reference sites within each, unit 1 (coastal ecosystem), unit 2 (desert ecosystem 1), and unit 3 (desert ecosystem 2), (c) plot-level sampling points.

Soil Sampling and Laboratory Analysis

At each of the 15 points within a plot, we excavated the soil by shovel at the surface (0-10 cm of depth) and sub-surface (5-35 cm of depth), placing the samples into glass jars. There were a total of 30 samples collected at each plot. Soils were sampled in both damaged and reference sites after the rainfall season in March 2013.

Laboratory analysis was conducted by Kuwait University, measuring: 1) Electrical Conductivity (EC), 2) pH, 3) fertility including phosphorus (P), potassium (K), Sodium (Na), Iron (Fe), and magnesium (Mg), 4) Total organic carbon (TOC), and 5) grain size (sorted by sand, silt, and clay). All chemical tests were conducted in accordance with US EPA 6010 standards. The percentage of organic matter was determined from the total organic carbon (TOC) by multiplying by 1.9. Grain size analysis was conducted in accordance with ASTM D422-63 (reapproved 2007), the "Standard Test Method for Particle-Size Analysis of Soils".

Statistical Analysis

The following hypotheses were addressed with the soil data set, using statistical tests within RStudio programming (3.12, RStudio, Boston, USA):

Hypothesis 1: Soil properties differ between the three units (as each unit represents different ecosystem). To address this hypothesis, we used single factor analysis of variance (ANOVA) to test the differences between the three units based on their soil properties. Three post-hoc comparisons were made: between unit 1 (coastal ecosystem) and unit 2 (desert ecosystem), unit 1 (coastal ecosystem) and unit 3 (the second desert ecosystem), and between the two desert ecosystems (unit 2 and unit 3). Each unit contained 2 plots as replicates, with the 15 sample points as subsamples. Tests were conducted for both surface and depth, and were applied for each soil property individually.

Hypothesis 2: Soil properties differ between the reference (DMZ, fenced) and damaged (unfenced) sites. To address this hypothesis, we used a two-factor (ANOVA) with the factors as reference vs. disturbed sites, and topsoil (0-5 inch) vs. depth (5-15 inch). Tukey post-hoc contrasts were used to further illuminate specific combinations of difference: (a) between disturbed and reference sites for the coastal ecosystem (1 rep each), (b) between disturbed and reference at the desert ecosystem such that the two desert ecosystems (unit 2 and unit 3) were combined together and treated within the same grouping, as the results of Hypothesis 1 showed that they were not significantly different (2 reps each). The 15 sample points within each reference vs. disturbed and topsoil vs. depth combination were considered subsamples. Hypothesis 2 ANOVA tests were applied for each soil property individually.

Vegetation Assessment

To assess vegetation, we used the Braun-Blanquet cover-abundance scale (Wikum & Shanholtzer 1978), which is well-suited to explain graphically species-environment relationships (Van der Maarel 1975; Wikum & Shanholtzer 1978; Podani & Díaz 2006). This method uses a scale table (Table 4.1). Vegetation assessment was done within a 10-meter radius at the 15 soil sample locations within each plot.

Table 4.1 Braun-Blanquet Cover Scale

Braun-Blanquet Cover Scale	
Rating	**Description**
+	Sparsely, or very sparsely present; cover very small.
1	Plentiful, but of small cover value.
2	Very numerous, or cover 5-20%.
3	Any number of individuals; cover 25-50%.
4	Any number of individuals; cover 50-75%.
5	Cover greater than 75%.

Selecting Locations for Re-vegetation

Re-vegetation using seeds and seedlings have been suggested as a possible project action at Umm Nigga, and have been used at other locations in Kuwait. However, many seeding/seedling projects fail due to unsuitable locations, seeding at the wrong time, poor quality seeds, or too few seeds (Lippitt et al. 1994; Whisenant 1999; Dorner 2002; Pfaff et al. 2002). An important first step is to determine suitable locations for seeding and seedlings to maximize their survival.

Six unique factors of site suitability for seeding/seedlings were considered and combined using GIS: land use, previous damage, soil characteristic, slope, soil erosion, and runoff. The major land use activities that existed in the study area were agricultural activities and roads. It is important to avoid planting near these activities, as soil near agricultural areas in this region have high salinity due to the use of irrigation. It is also important to select locations that are non-adjacent to roads since the movement of vehicles can affect seed germination and seedling growth (particularly in Kuwait, it is not uncommon for people to leave the road surface and venture). We considered previously damaged locations with low vegetation cover as suitable sites, but excluded undamaged sites since the goal is to restore damaged sites. Soil characteristics are an important factor in this region; particularly soil texture, salinity, infiltration, and drainage (Allen 1995; Bainbridge 2007; Hillel 2007) were considered. We also considered the slope (percentage), as high slopes increase runoff and erosion, as well as affecting the distribution of the seeds and affecting seedling growth.

Data Collection and Preparation

GIS-based data sets were collected from the Kuwait Institute of Scientific Research (KISR) and Kuwait University including a land use layer, a soil survey, plant community map, and a Digital Elevation model (DEM), produced at a scale of 1:250,000. The land use layer was used to determine agricultural areas and roads. For the plant community maps to identify previous damage, we created a classified image, as detailed in chapter 2. These maps were made from geo-referenced Landsat 8 imagery, taken following the rainfall season March 2014, and classified using ENVI 5.2 (Exelis Visual Information Solution, McLean, VA). The soil survey was used to determine soil characteristics based on texture, salinity, infiltration and drainage. The DEM was used to generate the slope. For runoff and soil erosion, we utilized a layer of estimated potential soil loss, created using an empirical GIS model in chapter 3.

GIS Processing

The six criteria were applied using GIS to determine suitable locations for re-vegetation (Fig 4.2). Several geo-processing steps were applied for each criterion using ModelBuilder in ArcGIS (10.3, ESRI, California, USA). Each layer was given a value of 0 for unsuitable areas, and 1 for suitable areas. The six suitability layers were combined and re-ranked to select the final suitable locations for re-vegetation. Finally, we used our field vegetation assessment (see Vegetation Assessment section) and Kuwait vegetation map, which was established by Halwagy & Halwagy (1974). The specific geo-processing steps are as follows.

Land use

Roads and adjacent areas were selected as unsuitable locations for re-vegetation. They were given a buffer distance of 300m, then, the layer was converted to raster and reclassified into 0 and 1. Agricultural areas were given a buffer distance of 2 km, converted to raster, and classified into 0 and 1. Damaged locations were considered suitable (1). Locations with high vegetation cover and reference areas were considered unsuitable locations (0).

Soil characteristics

Clay soils and soils with high salinity and low infiltration rates were considered unsuitable (0) for the re-vegetation program. However, sandy to loam soils were considered suitable (1) for re-vegetation.

Slopes

Slopes greater than 2% were considered unsuitable (0), since seeds and seedlings may be negatively affected by water runoff and soil erosion.

Runoff and soil erosion

Locations with moderate, high, and very high erosion rates were considered unsuitable (0), however, locations with very low and low erosion rates were considered suitable locations (1). High runoff locations were also considered unsuitable for re-vegetation (0).

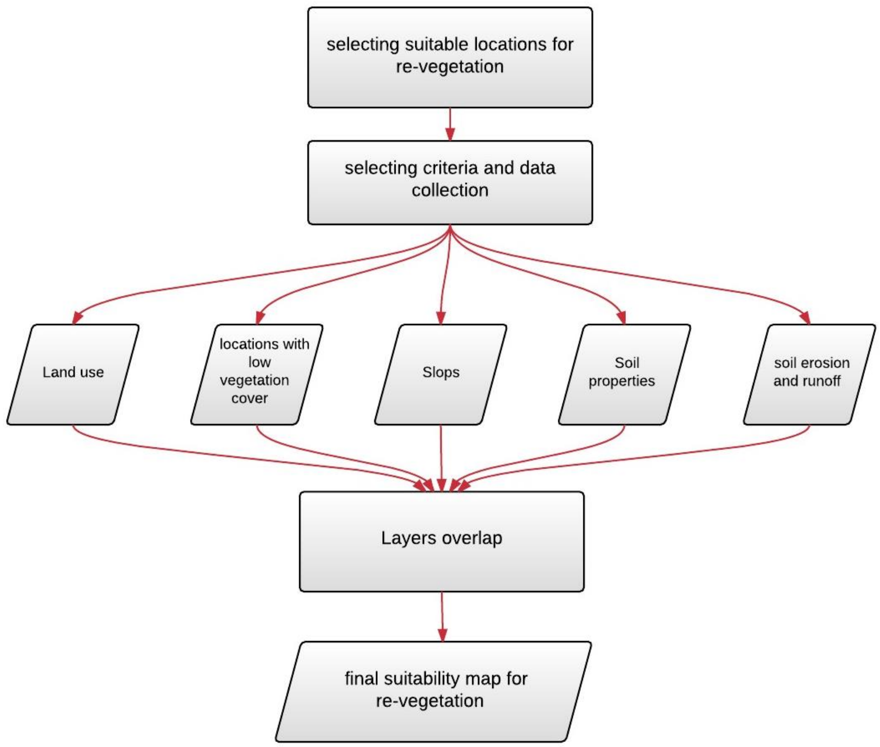

Fig 4.2 Flow chart for the major processes for the selection of suitable locations for re-vegetation.

Results

Hypothesis 1: Soil Properties Differ Between the Three Units (As Each Unit Represents Different Ecosystem)

The coastal ecosystem (unit 1) was significantly different in soil properties when compared with the two desert units (Table 4.2). The coastal ecosystem (unit 1) was significantly different than the first desert ecosystem (unit 2) with p values < 0.05 for all chemical properties, except Fe (p = 0.7). They also differed in soil particles including sand and clay percentage (P < 0.0001), but they were not significantly different with silt percentage (p = 0.05). The coastal ecosystem (unit 1) also differed in all soil properties when compared with the second desert ecosystem (unit 3) (for p <0.05). However, the two desert ecosystems (unit 2 and unit 3) were not significantly different for most of the chemical properties except pH (p = 0.04) and Mg (p = 0.005). For grain size analysis, they were not significantly different in sand (p = 0.4) nor clay (p = 0.8) percentage, but they differed in silt percentage (p = 0.005). Based on the lack of significant difference, we decided to combine the two desert units and consider them as a single ecosystem for the Hypothesis 2 tests.

Table 4.2 Statistical results for hypotheses 1 and 2

Component	**Hypothesis 1:** *Soil properties differ between the three units (as each unit represents different ecosystem)*						**Hypothesis 2:** Soil properties differ between the reference (DMZ, fenced) and damaged (unfenced) sites			
	Coastal vs Desert 1		Coastal vs Desert 1		Desert 1 vs Desert 2		Reference vs Disturbed at Coastal area		Reference vs Disturbed at Desert area	
	F value	P value	F value	P value	F value	P v	F value	P value	F value	P value
pH	12.9	**0.0006**	32.86	**0**	4.05	**0.04**	4.5	**0.03**	0.58	0.4
EC	198.3	**0**	438.7	**0**	0.54	0.4	8.5	**0.005**	0.06	0.7
Na	71.96	**0**	115.6	**0**	0.12	0.7	3.9	0.5	0.21	0.6
Mg	51.57	**0**	13.5	**0.0005**	13.46	**0.0005**	32.8	**0**	3.9	0.05
Ca	64.14	**0**	8.7	**0.004**	0.05	0.8	128.3	**0**	40.32	**0**
Fe	0.09	0.7	15.6	**0.0002**	2.96	0.09	1.51	0.2	0.06	0.7
K	14.29	**0.0003**	8.94	**0.004**	0.09	0.7	1.11	0.2	1.3	0.2
P	40.8	**0**	32.8	**0**	3.61	0.06	0.028	0.8	7.5	**0.007**
OM	4.2	**0.04**	16.3	**0.0001**	0.87	0.8	12.925	**0**	10.1	**0.002**
Grain size										
Sand %	15.6	**0**	40.05	**0**	0.496	0.4	63.2	**0**	1.55	0.2
Clay %	20.77	**0**	59.8	**0**	0.058	0.8	1.37	0.2	1.25	0.2
Silt %	3.931	0.05	7.57	**0.007**	8.5	**0.005**	82.29	**0**	11.6	**0.001**

Hypothesis 2: Soil Properties Differ Between the Reference (DMZ, Fenced) and Damaged (Unfenced) Sites

Coastal Ecosystem

Several soil properties varied between the reference and damaged sites at the coastal ecosystem (Table 4.2). Electrical conductivity (EC) and pH differed at $p = 0.03$ and $p = 0.005$, respectively, both were higher at the damaged site (Figs 4.3a-b). Some soil nutrients also differed including Mg and Ca ($p < 0.001$), with Mg higher at the damaged site and Ca higher at the reference site (Fig 4.4a). Organic matter greatly varied ($p = 0.0001$), and was higher at reference site (Fig 4.3c). This difference in organic matter could be correlated with the differences in soil texture, as the percentage of sand and clay were significantly different ($p < 0.001$), with the disturbed site sandier and the reference site more clayey (Fig 4.5a). Generally, the reference sites had a sandy loam and the damaged sites had loamy sand.

The topsoil (0-5 inches of depth) and deeper soil (5-15 inches of depth) were not significantly different for most soil properties between reference and disturbed sites, except for P and organic matter ($p < 0.001$). They were higher at the reference area. However, when contrasting between topsoil and depth for each site individually, no significant differences were found at the damaged site. However, at the reference sites, there were differences in P and organic matter (Fig 4.6c and Fig 4.7a-b), with both higher at greater soil depth.

Desert Ecosystem

There were few differences in soil properties between the reference and damaged sites for the desert ecosystem units (Table 4.2). Electrical conductivity (EC) and pH ($p > 0.05$) were not significantly different. Soil nutrients were not significantly different except for P and Ca ($p < 0.001$). They were higher at the reference site (Fig 4.4b). Organic matter content was also higher at the reference site (Fig 4.3c). This was expected since P and organic matter are correlated with the present of vegetation, which is higher at the reference sites. The concentration of Ca was high at all sites compared with other metals, which could be due to the low rainfall. The soil grain sizes were not significantly different in sand or clay percentages, however, they did vary in silt percentage ($p < 0.001$) (Fig 4.5b). Reference and disturbed sites within the desert ecosystem units had the same soil texture, loamy sand.

The topsoil and deeper soil differed between the reference and damaged sites in the desert units for pH, Na, Mg, and Fe (all $p < 0.05$). They also differed in sand percentage. However, when contrasting between the topsoil and deeper soil within each of the reference and damaged sites individually (across the desert units), the topsoil soil and deeper soil were not significantly different at the damaged sites, though they were higher at the reference sites in pH (Fig 4.6e), Mg, and Fe (all $p < 0.5$) (Fig 4.7b); they were higher in the topsoil.

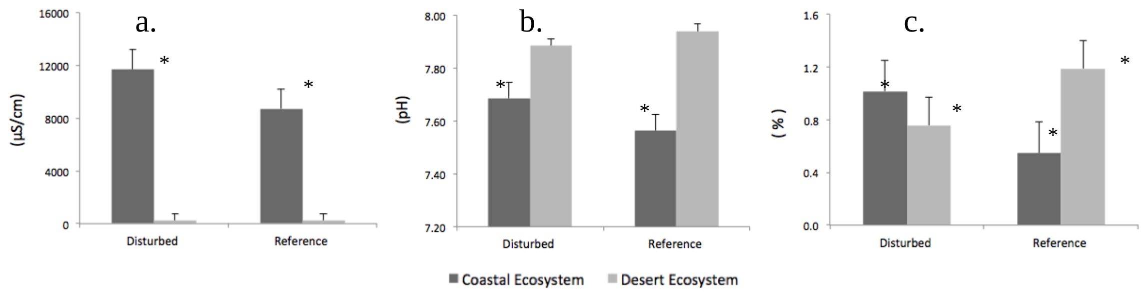

Fig 4.3 (a) EC, (b) pH, and (c) organic matter content at damaged and reference sites in the coastal and desert units. Values followed by (*)statistically differ with P value < 0.05.

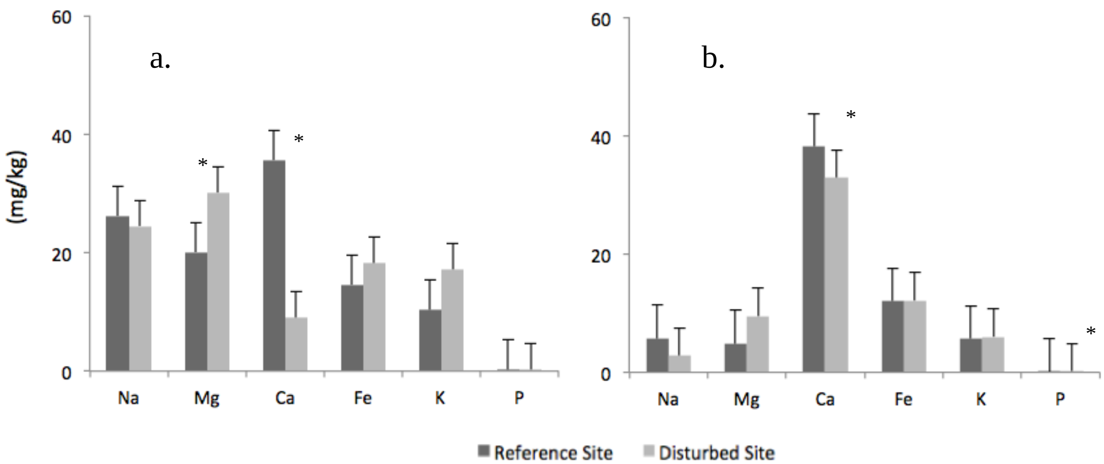

Fig 4.4 (a) Amount of nutrients in the reference and damaged sites at the coastal unit (b) Amount of nutrients between reference and damaged sites at the desert units. Values followed by (*) statistically differ with P value < 0.05.

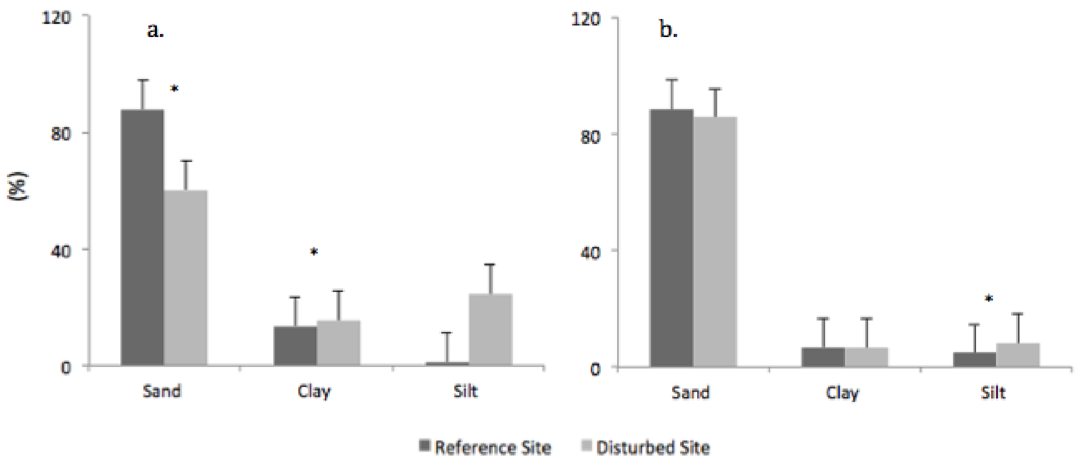

Fig 4.5 (a) Amount of soil particles in the reference and damaged sites at the coastal unit (b) Amount of soil particles in the reference and damaged site at the desert units. Values followed by (*) statistically differ with P value < 0.05.

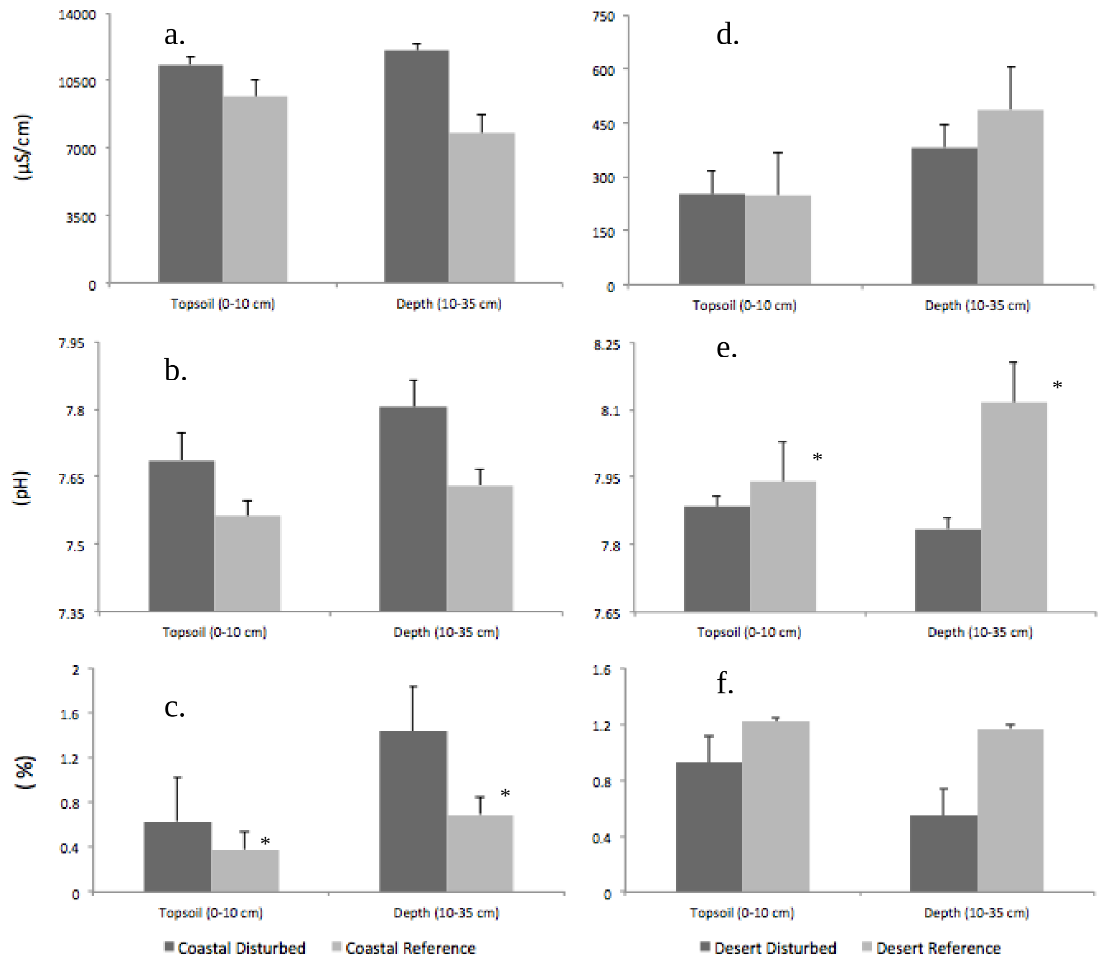

Fig 4.6 (a) Soil EC, (b) soil pH, and (c) organic matter for the coastal unit, for both the topsoil and soil at depth. (d) Soil EC, (e) soil pH, and (f) organic matter for the desert units, for both the topsoil and soil at depth. Values followed by (*) statistically differ with P value < 0.05.

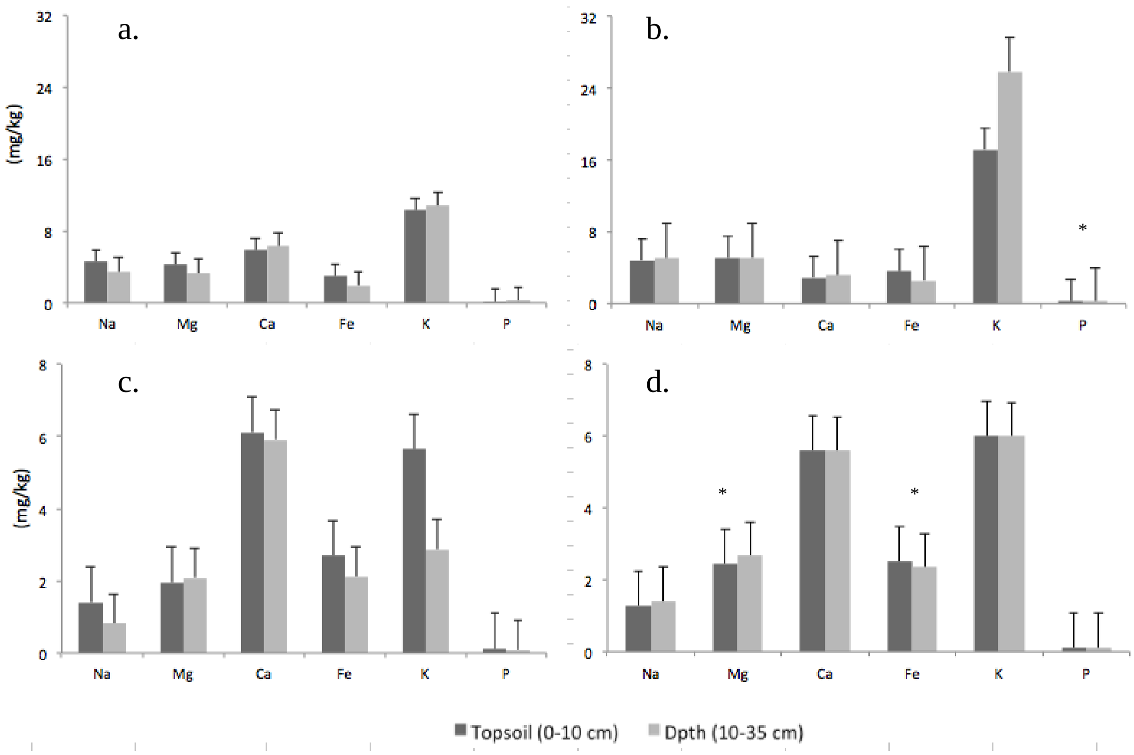

Fig 4.7 Amount of nutrients in the topsoil (0-5 cm) and soil depth (5-15 cm) at the (a) coastal damaged site (b) coastal reference site, (c) desert damaged sites (d) desert reference sites. Values followed by (*) statistically differ with P value < 0.05.

Vegetation Assessment

The reference and damaged sites at the coastal unit have the same species richness (Table 4.3). Only one species was found at both sites, *Halocnemum strobilaceum*, with a cover of 25-50%. Differences were found in species richness between the reference and damage sites for the desert units. The reference site was more rich in species. Twenty species were found at the first plot in the reference site, with *Haloxylon salicornicum* cover greater than 75%, and *Rhanterium epapposum* with cover of 50-75%. The remaining species had very small cover value. At the second reference plot in the desert units, 19 different species were found, where most species are the same as in the first plot. The dominant species at this site was *Rhanterium epapposum* with a cover greater than 75%, and *Haloxylon salicornicum* with a cover of 50-75%. The remaining species were very low in cover. Conversely, the damaged sites in the desert units had very low species cover in both plots, and were primarily annuals.

Suitable Locations for Re-vegetation

The GIS suitability analysis showed that 25% of the total area was suitable for re-vegetation (Fig 4.8). The west part of the study area had a greater area that was suitable for re-vegetation. These locations only cover desert ecosystems. While our results help determine the suitable locations for re-vegetation, it is still important to identify the species that can be planted in these spots. The vegetation assessment from the reference areas and vegetation community map (as historical data for the GIS model) both showed that *Haloxylon salicornicum* and *Rhanterium epapposum* were the best native species for planting.

Table 4.3 Plant assessment at each plot

Coastal Ecosystem		
Disturbed Site		
Species	**Braun-Blanquet Cover Scale**	**Description**
Halocnemum strobilaceum (Pall.) M.Bieb.	3	Any number of individuals; cover 25-50%.
Reference Sites		
Species	**Braun-Blanquet Cover Scale**	**Description**
Halocnemum strobilaceum (Pall.) M.Bieb.	3	Any number of individuals; cover 25-50%.
Desert ecosystem (1)		
Disturbed Site		
Species	**Braun-Blanquet Cover Scale**	**Description**
Astragalus schimperi Boiss.	1	Plentiful, but of small cover value.
Arnebia decumbens (Vent.) Coss. & Kralik	1	Plentiful, but of small cover value.
Gymnarrhena micrantha Desf.	1	Plentiful, but of small cover value.
Podaxis sp.? (white mushroom)	+	Sparsely, or very sparsely present; cover very small
Reference Site		
Species	**Braun-Blanquet Cover Scale**	**Description**
Haloxylon salicornicum (Moq.) Bunge ex Boiss.	5	Cover greater than 75%.
Rhanterium epapposum Oliv.	4	Any number of individuals; cover 50-75%.
Senecio glaucus L.	2	Very numerous, or cover 5-20%.
Koelpinia linearis Pall.	1	Plentiful, but of small cover value.
Centaurea pseudosinaica Czerep.	+	Sparsely, or very sparsely present; cover very small.
Gypsophila capillaris (Forssk.) C.Chr.	2	Very numerous, or cover 5-20%.
Pennisetum divisum (Forssk. ex J.F.Gmel.) Henrard	2	Very numerous, or cover 5-20%.
Plantago boissieri Hausskn. & Bornm.	2	Very numerous, or cover 5-20%.
Anisosciadium lanatum Boiss.	1	Plentiful, but of small cover value.
Plantago ovata Phil.	1	Plentiful, but of small cover value.
Salvia aegyptiaca L.	1	Plentiful, but of small cover value.
Helianthemum lippii (L.) Dum.Cours.	1	Plentiful, but of small cover value.
Carduus pycnocephalus L.	1	Plentiful, but of small cover value.
Schismus barbatus (L.) Thell.	2	Very numerous, or cover 5-20%.
Launaea mucronata (Forssk.) Muschl.	1	Plentiful, but of small cover value.
Atractylis carduus (Forssk.) C.Chr.	1	Plentiful, but of small cover value.
Heliotropium bacciferum Forssk.	1	Plentiful, but of small cover value.
Scabiosa olivieri Coult.	1	Plentiful, but of small cover value.
Scabiosa palaestina L.	+ (rare plant)	Sparsely, or very sparsely present; cover very small.
Desert ecosystem (2)		
Disturbed Site		
Species	**Braun-Blanquet Cover Scale**	**Description**
Haloxylon salicornicum (Moq.)	+	Sparsely, or very sparsely present; cover very small
Gynandriris sisyrinchium (L.) Parl.	1	Plentiful, but of small cover value.
Reference Site		
Species	**Braun-Blanquet Cover Scale**	**Description**
Rhanterium epapposum Oliv.	5	Cover greater than 75%.
Haloxylon salicornicum (Moq.) Bunge ex Boiss.	4	Any number of individuals; cover 50-75%.
Gypsophila capillaris (Forssk.) C.Chr.	2	Very numerous, or cover 5-20%.
Plantago boissieri Hausskn. & Bornm.	2	Very numerous, or cover 5-20%.
Allium vineale L.	1	Plentiful, but of small cover value.
Launaea mucronata (Forssk.) Muschl.	2	Very numerous, or cover 5-20%.
Plantago ovata Phil.	1	Plentiful, but of small cover value.
Atractylis carduus (Forssk.) C.Chr.	2	Very numerous, or cover 5-20%.
Centaurea sinaica DC.	1	Plentiful, but of small cover value.
Anisosciadium lanatum Boiss.	1	Plentiful, but of small cover value.
Senecio glaucus L.	2	Very numerous, or cover 5-20%.
Stipa capensis Thunb.	2	Very numerous, or cover 5-20%.
Schismus barbatus (L.) Thell.	2	Very numerous, or cover 5-20%.
Rumex vesicarius L.	1	Plentiful, but of small cover value.
Brassica tournefortii Gouan	2	Very numerous, or cover 5-20%.
Heliotropium bacciferum Forssk.	1	Plentiful, but of small cover value.
Asphodelus tenuifolius Cav.	1	Plentiful, but of small cover value.
convolvulus oxyphyllus	+	Sparsely, or very sparsely present; cover very small.

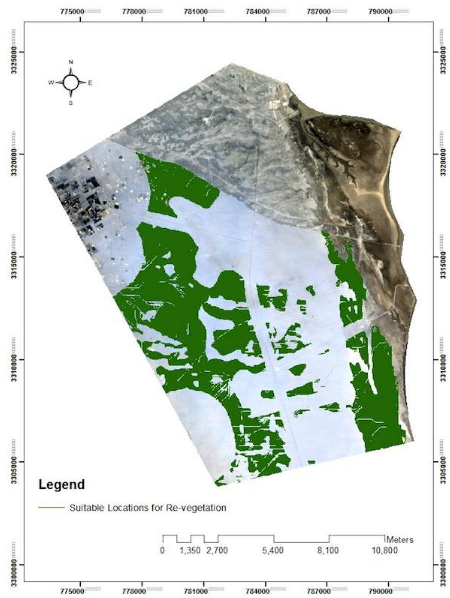

Fig 4.8 Suitable locations for re-vegetation efforts.

Discussion

Site Condition

There appears to be no evidence that the coastal ecosystem was damaged. Both the damaged and reference sites in this unit had a low amount of vegetation cover and species richness. This is in concordance with the evidence from the soil properties analysis whereby the soil chemical and physical properties between the two sites were generally similar. This is likely because grazing and camping are not typically conducted in these the tidal flat areas due to the muddiness, high salinity, and difficulty of access. Moreover, remote sensing analysis in chapter 2 showed that the coastal ecosystem had a continuously low vegetation cover since 1988, suggesting that this is the natural state. It is possible the little variation in soil chemical properties found among these sites may be related to the differences in soil texture. Most soil nutrients were lower at the reference site, which could be related to the higher sand percentage, as sand affects nutrient leaching and water holding capacity, particularly in desert regions (Verboom & Pate 2006; Bainbridge 2007; Tefera et al. 2007). Overall, the low amount of vegetation in the coastal ecosystem unit is most likely related to the natural condition of tidal flooding, rather than any disturbance. Therefore, we recommend excluding this ecosystem from the restoration plan.

However, the unfenced areas and sites within the two desert ecosystem units are highly disturbed due to human activities. A large quantity of sheep and camels have been noted during our field visits to the location, as well as trash and other waste related to spring camping by people (Fig 4.9d, f). Camping in Kuwait disturbs very large portions of the landscape, as temporary tent cities are erected and four-wheel driving is a prime recreational activity. In addition, only a few annual plant species with low coverage were found in the damaged sites in the desert units, as compared with the relatively high richness and cover found in the reference sites. Similarly, as discussed in chapter 2, we found that vegetation cover was low at the damaged sites (3%) and high at the reference sites (73%). Still, our results identified few significant differences among most of soil properties between the damaged and reference sites in the desert units, except for Na, P, and organic matter, which were higher at the reference sites, likely correlated with the greater vegetation cover. In addition, the soil texture was similar across the desert unit sites, all were covered with loamy sand. Thus, it is likely that the soil itself does not need remediation for vegetation to grow, but rather the grazing and camping pressure are the cause for apparent differences in vegetation cover damage. This result suggests that fencing and restricting access to the area could result in autogenic recovery, without unnecessary monetary expenditures on soil remediation, planting/seeding, and irrigation.

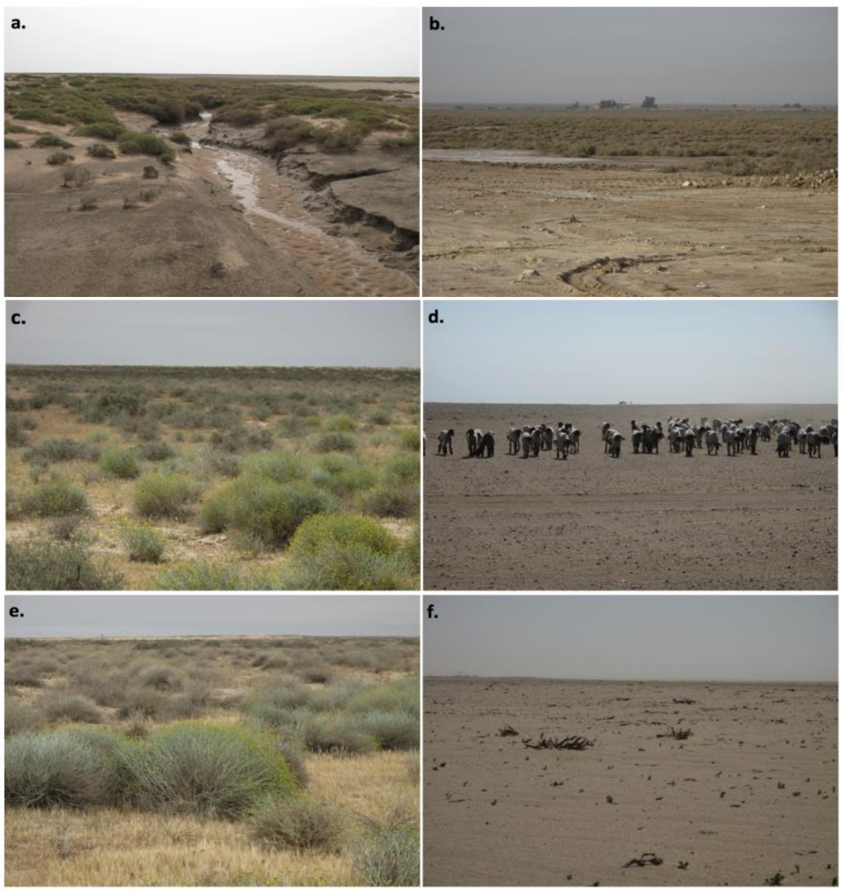

Fig 4.9 (a) Coastal ecosystem (unit 1) damaged site, (b) Coastal ecosystem (unit 1) reference site, (c) Desert 1(unit 2) reference site, (d) Desert 1(unit 2) damaged site, (e) Desert 2 (unit 3) reference site, and (f) Desert 2 (unit 3) damaged site.

Restoration Plan

Ecosystems have different processes that contribute to proper functioning; therefore, no single assessment or approach is the best for all restoration management objectives. Using the results herein and the results of chapter 2, we next designed a conceptual framework for restoration planning in arid ecosystems such as Kuwait (Fig. 4.10). Any restoration approach should start with setting the objectives, which need to be developed according to the availability of the resources. Then, one needs to understand the history and condition of the site.

Often, the first step in restoration is to assess the disturbed ecosystem including identifying surface soil conditions, relevant hydrological processes (infiltration and runoff), and nutrient cycling at the site (Whisenant 1999; Tongway & Ludwig 2011). However we found that for our site, understanding the history of the location in terms of the type, nature, location, and intensity of disturbances should be considered first, as it helps in understanding what we are seeing today, guides our choice of assessment tools, and informs our views on the best restoration strategy (Bainbridge 2007). After that history is determined, it then becomes logical to expend resources to assess the current condition of the soils and ecological processes.

These two steps can help in selecting suitable alternatives for any restoration program. A first question to ask is whether the condition of the soil is suitable for natural vegetation recovery, or will soil remediation be required? If soil remediation is necessary, the appropriate remediation approach needs to be selected first. However, if the soil condition is appropriate, the second question is “Will natural recovery occur if the stressor that is damaging or has damaged the site is removed?" If the answer is yes, then removing the stress could be the best and cheapest approach to restoring the location. If natural autogenic recovery will not occur, there are likely other factors affecting seeding establishment that need to be considered, such as a low amount of precipitation or a low number of seeds. If necessary, re-vegetation using seeds or seedlings and irrigation may help increase recovery. It is then important to monitor the site in order to evaluate the results and determine whether the objectives are achieved, or whether re-assessment needs to be conducted.

Following this framework, two plans were selected for restoring the Umm Nigga site. Plan A was to remove the stress by fencing the location and seek autogenic recovery. Plan B was a secondary option, which we only recommend if recovery is slow. In that case, after few years then we can plant native species and/or irrigate. Plan A is clearly a cheaper and less impacting option.

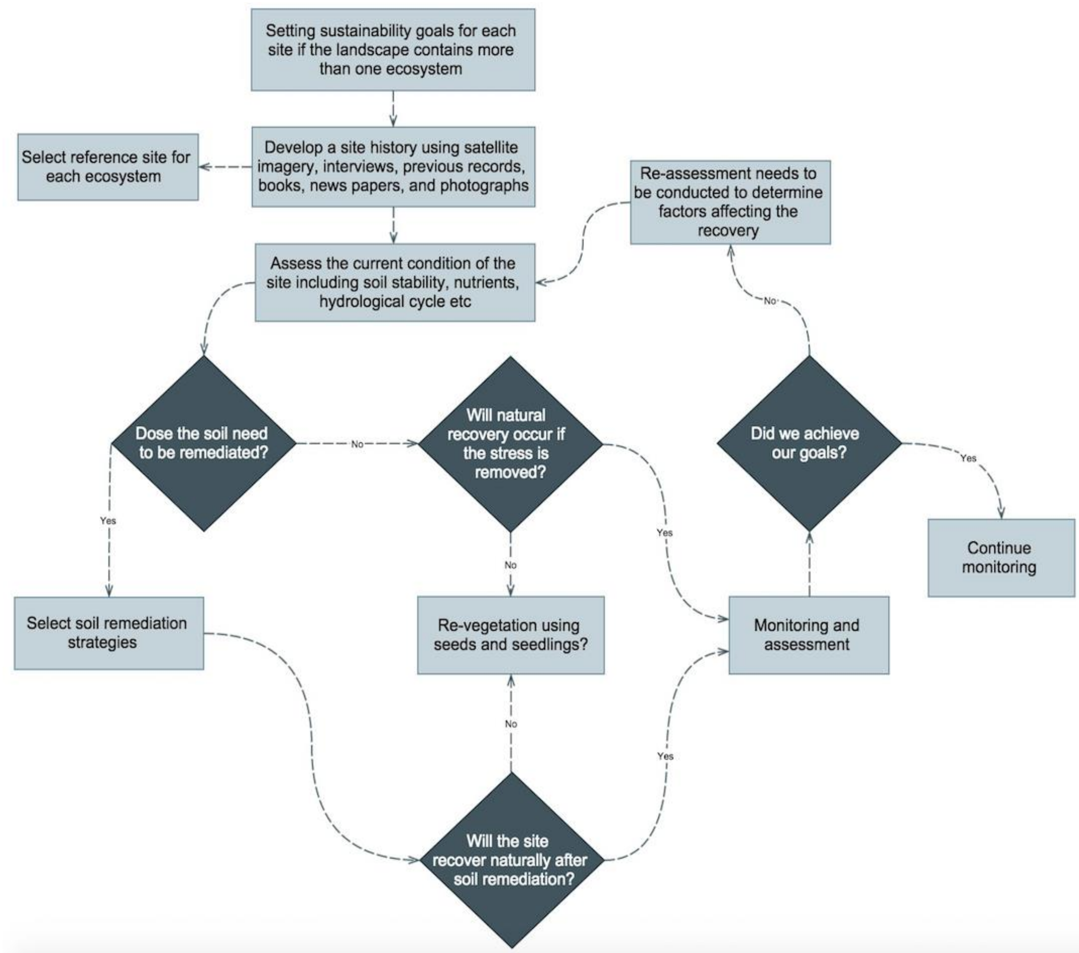

Fig 4.10 Conceptual framework for restoration planning on arid landscapes, such as Umm Nigga, Kuwait.

Plan A

For Umm Nigga, we contend that planting or irrigation may not be necessary to restore the location for several reasons. First, the vegetation cover was naturally low at the coastal ecosystem; therefore, this area does not require any remediation or re-vegetation. The desert ecosystem was heavily disturbed due to human activities such as overgrazing and camping. Still, most soil properties were not significantly different between the reference and disturbed sites in these desert units. The damaged sites appear to have fair concentration of soil nutrients compared with the reference sites. Soil nutrients, particularly P and organic matter, were low at both types of sites. Soil nutrients are generally low in desert ecosystems, with limited organic matter and very low levels of N and P. They are sensitive to erosion as most soil nutrients are concentrated in the surface soils of the desert. Soil pH is also high at most desert ecosystems, which also plays an important role in limiting soil nutrients (Lewis et al. 2006). Native plants can often survive and grow under the same conditions we found at the sites, as evidenced by the reference site. In addition, a few annual seedlings were present at the damaged site, which indicates that early succession may be occur before these plants can be grazed. Therefore, fencing the site and the stopping disturbance activities is likely to release the ecosystem, allowing early succession processes to begin.

Second, several examples in Kuwait demonstrate that natural recovery occurred within few years. We used remote sensing in chapter 2 to describe the site history of Umm Nigga, with the results showing that the location was disturbed before the First Gulf War in 1990, and that natural recovery occurred after the war from 1991- 1998, when human activities were limited due to landmines and unexploded ordinance. However, the site became degraded again due to human activities and grazing, after it was cleared of mines in 1998, while the fenced reference site (DMZ) continued increasing in vegetation cover until 2013. Similarly, Brown & Al-Mazrooei (2003) showed that rapid natural recovery occurred after 4 years in Sabrya, Kuwait after grazing stress was removed by fencing the location, because the soil was still adequate.

Third, coarse, sandy substrates are usually favorable for rapid plant growth due to the fact that water percolates through the surface layers quite rapidly. In deep soils, seeds can accumulate and be retained as many desert plants build up substantial inter-annual seedbanks, lasting for several years (Brown & Al-Mazrooei 2003). Abdullah (2015) extracted DNA from soil samples collected from Umm Nigga (the same study area), for both disturbed and referenced sites. Those results showed that there was more DNA recovered (≈ 9.09 ng/µl) at the reference site as compared to the disturbed site (≈ 1.52 ng/µl), but that the disturbed site was not devoid of such material and that living organisms are present though at a lower rate compared to the fenced reference area.

Implementing large-scale projects, which include re-seeding, planting, and irrigation for damaged arid areas can be very expensive, and often create environmental problems. Thus, in situations where the abiotic function of disturbed site has not been irreversibly damaged, then limited and appropriate management may be the best solution for vegetation recovery (Papanastasis 2009). By fencing the area, vegetative can likely recover naturally through autogenic succession, or self-repair of the formerly disturbed ecosystem. Successional trajectory can potentially change as well (Whisenant 1999).

Vallentine (1989) suggested that if at least 15% of the existing species are surviving, then the damaged site could be restored through management practices alone. Since the fenced reference site shows that the ecosystem likely has the ability to recover naturally, then removing the stress and allowing natural recovery, without planting or seeding, will be much cheaper. It will also leave less of a human imprint on long-term species composition, and thus be less likely to alter the future successional trajectory (Murcia 1997; Celentano et al. 2011; Van Andel & Aronson 2012).

Plan B

In the event that the Umm Nigga site does not recover sufficiently after few years from fencing and removing the stress, then seeding, planting, and irrigation could be used as a secondary option. This option requires several considerations, however, which need to be examined before planting. First, one needs to determine the suitable locations for planting seedlings and seed germination, and ensure the breakage of seed dormancy and the availability of nutrients and water (Whisenant 1999). Our results showed that about 25% of the location is suitable for such planting. Still, it will be important to follow up on our results and evaluate the suitable sites in the field.

Once the sites are finalized, the first question that would then need to be asked is “What species are appropriate for these sites?” We have provided answers to this question through the vegetation assessment at the reference site, which helped determine the plants that should be considered. The historical data produced by our vegetation community map can also help in refining the planting goals, and assessing the soils also helped in suggesting what can, and what cannot be grown. Our results showed that

Haloxylon salicornicum and *Rhanterium epapposum* could be planted in this stage for the desert ecosystem units. It is also important to understand the proper collection, processing, storage, and germination techniques for each plant species (Bainbridge 2007), as well as selecting the species that are suited for each part of the landscape.

Thus, if in deciding to go with planting rather than natural recovery, it will be important to choose between strategies that can modify the site for the chosen species, or strategies that rely on plant tolerance for the existing conditions (Whisenant 1999). It will also be important to select plants that can improve the available resources. Well-adapted plants should also maintain the damaged hydrologic and nutrient cycling processes (Jones et al. 1996; Jones et al. 1997). Restoration plantings of native species can be more challenging and costly than planting crops (Van Andel & Aronson 2012), so if taking this route more research may be needed to have successful results.

Conclusion

Our work demonstrates the importance of having a good site assessment in order to design a successful restoration program. The coastal ecosystem units did not appear to need restoration. However, the desert ecosystem units were degraded and contained low to no vegetation cover, and had low amounts of organic matter and P in the soil, both of which were likely related to absence of the vegetation. In contrast, the reference sites within the desert ecosystem units had a relatively high quantity of vegetation cover, organic matter, and P. Given that the many other nutrients and physical properties of the soil were similar among the damaged and reference sites in the desert units, it is likely that fencing the area will be enough to release the ecosystem from grazing pressure. Based on what has been seen within the reference sites within the DMZ since the First Gulf War, the native desert plants can adapt and survive under such fenced conditions. Re-vegetation by seeding, planting seedlings, or providing irrigation could be considered as a second option, but only if fencing alone cannot restore the ecosystem. We have thus developed a conceptual framework for restoration work to proceed in Umm Nigga, Kuwait, based on the required criteria for a successful autogenic recovery to occur. This framework may prove useful for designing restoration programs in other arid ecosystems as well.

CHAPTER V

CONCLUSION AND RECOMMENDATIONS

Our work demonstrates the power of using GIS and remote sensing in restoration planning. Remote sensing helped in determining the history of a desert site for a relatively large area, when there are no other histories available for reference. The study illustrates that understanding the history of the location can make the restoration plan much more effective, as well as help in finding proper reference sites. It also provided information regarding the level of the problem by knowing if it is ongoing, recent, or historic. It was found from the site history that vegetation cover was low at the coastal area since 1988, which might be due to natural geomorphic changes, as opposed to grazing and camping, as these activities were not typically conducted along the tidal flat due to the muddiness, high salinity, and difficulty of access. However, vegetation cover was very low (2%) at the desert ecosystem in 1988 and increased after the war to reach 37% in 1998, but then decreased again in the unfenced site but continued increasing in DMZ. Our results also document that the effect of overgrazing and camping is the most likely reason for the decrease in the natural vegetation, and that fencing alone may provide an adequate plan for a quick restoration of large areas.

Utilizing GIS modeling also helped in estimating soil erosion at large landscapes. It was illustrated from the results that the MPSIAC model was the superior model for mapping soil erosion compared with EMP and RUSLE. It also produced the most even and detailed results, likely because of the greater number of modeled factors that represent the various mechanisms that affect soil erosion. Vegetation cover also played an important role in decreasing the amount of soil erosion and controlling desertification. Thus, it is necessary to restore the unfenced areas, as practices that limit vegetation loss could potentially lower soil erosion by 32%. This model could also be used for arid sites in the region. Direct field measurements of surface soil erosion are highly recommended in the future to confirm the results of our model evaluation work.

It was illustrated from the soil sampling and vegetation assessment that the coastal area was not disturbed since there were no differences between vegetation cover at the disturbed and reference sites, and the same vegetation type and percent cover was also determined at both sites. However, the desert ecosystem was quite damaged and restoration is required as the vegetation cover was very low at the disturbed (unfenced) site, and very high at the reference site. However, there were no significant differences in soil condition between damaged and reference site. Therefore, we have suggested two options to restore the disturbed ecosystem. First, we believe that the effect of camping and overgrazing is the most likely reason for the decrease in the natural vegetation, and that fencing alone may provide an adequate plan for a quick restoration of these large areas. Second, if the succession is slow after few years, then re-vegetation could take place as a secondary option. It was also illustrated from the GIS modeling for selecting suitable locations for re-vegetation, that 25% of the desert ecosystem was considered suitable for planting using seeds and seedlings. However, choosing re-vegetation strategy requires more research and assessment for the type and quality of seeds that could be collected and planted, and to evaluate the suitable sites for planting in order to maximize their survival.

Thus, it is critical to keep in mind that fencing the location alone will not stop the original sources of ecosystem disturbance in Kuwait, because the disturbance simply will be moved to other open areas, putting more pressure on those lands. Therefore, it is crucial that the country develops a national land management strategy and action plan to manage all land use in natural ecosystems, including grazing and spring camping. Increasing public awareness of the problem will also help in to control negative impacts caused by such activities. It is also necessary to consider the importance of precipitation or climatic changes on the recovery, as it will influence the successional trajectory of the site. The methodology and conceptual framework for restoration planning that was designed in this research is also be applicable for other damaged ecosystems in arid and semiarid regions.

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