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Wetlands Management

Assessing Risk and Sustainable Solutions

Edited by Didem Gökçe

WETLANDS MANAGEMENT - ASSESSING RISK AND SUSTAINABLE SOLUTIONS

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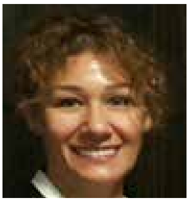
Meet the editor



Dr. Didem Gökçe is an associate professor of biology at Inonu University in Turkey. She graduated with a BSc in Biology at Hacettepe University, Ankara. She obtained her MSc and PhD degrees in Hydrobiology and Limnology at Hacettepe University. Her research focuses on water quality, limnology, and ecosystem ecology. Also, she combines experimental and field studies on qualitative and quantitative populations and community structures of plankton and benthic macroinvertebrate ecology.

Contents

Preface

Wetlands are ecosystems where water is the major controlling factor for the environment and cover a small area of the earth's surface (4–6%). These ecosystems are some of the most productive environments in the world. Major wetland types (marine/coastal wetlands, inland wetlands, and human-made wetlands) are becoming more crucial than ever for the sustenance of life in the world. Wetlands are the only ecosystems for whose conservation an international convention—the Ramsar Convention—was adopted as early as 1971. On the other hand, wetlands are constantly under threat by different effects, especially human activities. More essentially, some publications reveal the situations (political, institutional, cultural, economic, and ecologic) in countries that shape their wetland monitoring and management regulations and conservations.

Wetlands Management—Assessing Risk and Sustainable Solutions is among a number of books that look at the deficiencies in the issue. Particularly, climatic change and industrialization by anthropogenic activities are now accepted as a fact by most wetland ecosystem scientists. The purpose of this book is to help graduate scholars, scientists, and decision-makers utilize a methodology appropriate for a specific problem. Each chapter takes a crucial look at different approaches to the solution and analyzes wetland problems in the laboratory or in the field by collecting data.

The principal objective of this book is to provide unity and coherence in the studies of wetlands. To do so, the book is divided into five main sections:

1. Introduction
2. Water Quality and Diversity in Wetlands
3. Geographic Information Systems (GIS) and Remote Sensing Application
4. Human-made Structures
5. Valuation of Wetlands

The book aims to remedy this deficiency and both its content and authors have been selected with this purpose. Each author has knowledge of research, management, or practice on wetland assessment. Thus, the book concludes with an international view on wetland classification, problems, solutions, conservation, and restoration. I wish to thank all authors from many different regions of the world (Canada, China, India, Malaysia, Pakistan, South Africa, Swaziland, Taiwan, Turkey, United States of America, and Zimbabwe). This book could not have been possible without them.

Finally, I offer special thanks to Author Service Managers Romina Skomeršić, Irina Štefanić, Kristina Jurdana, and Anja Filipović for their help in publishing the book in its present form. I am also grateful for IntechOpen Publishers for their concern, efforts, and encouragement.

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Section 1

Introduction

Chapter 1

Introductory Chapter: Wetland Importance and Management

Didem Gokce

Additional information is available at the end of the chapter

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1. Introduction

Water is an important resource for all living beings. Therefore, the use of water and its supply from sources are very important. Wetlands are an ecosystem from mangrove to subarctic peatlands that have affected human. The earliest civilizations were established near the river, lake, and floodplains [1]. The Mesopotamian civilization is authoritatively accepted to have started around 4000–3500 BC between the Euphrates and Tigris River. The other ancestral civilization, Egypt, commenced in the Nile Valley at around 3200 BC. This represents the importance of the water and wetlands. The fact that people are in these regions is a reflection of how important it is for biotic diversity. Therefore, wetlands are a very critical ecosystem, and some of them are the most productive habitats.

Wetlands occur where the water table is at or near the surface of the land or where the land is covered by water [2]. Wetlands are the only ecosystems for whose conservation an international convention (Ramsar Convention) had been adopted as early as 1971. Ramsar Convention defined wetlands as “areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six meters” [2]. Wetlands have about 6% of the earth although they play an important role in hydrology and include mangroves, peatlands and marshes, rivers and lakes, deltas, floodplains and flooded forests, and even coral reefs. A wetland is a generalized concept including coastal wetlands. It exists in every climatic region, ranging from the polar zones to the arid zones.

Many wetlands are transitional area between aquatic and terrestrial ecosystems. These ecosystems are divided into two groups depending on the quantity of water: permanent and temporal flooded. Since wetlands are distributed in many different habitats on earth, they reflect different responses and behaviors to environmental changes. Therefore, wetland classification is important, and differences can be found. It is basically divided into natural and human-made constructed wetlands. In general, abiotic environmental factors, habitat differences, and biotic factors are considered to have a wide range of classification. Physical, chemical, and sediment quality determine wetland functions and classification. These situations classify its types [3].

Wetland ecosystems rapidly get worse due to various reasons. The environmental quality gradually deteriorates, and biotic diversity decreases in these habitats. It is estimated that more than 50% of specific wetland types in Europe, North America, Australia, and New Zealand were modified or changed during the twentieth century [4, 5]. Coastal wetland ecosystems are under extreme pressure, and it is estimated that about 35% of mangrove have been lost during the last two decades due to increasing agricultural area, deforestation, and freshwater reduction [6, 7].

Monitoring is the long-term regular observation and recording of current and altering situations. In the environmental assessment, these data were utilized to evaluate wetlands based on decision-making and planning processes. Consequently, wetland surveys have possessed a multidisciplinary perspective. The fact that the recognition of wetlands supplies many values for people and is an important case for global conservation has led to an increase in research and management activity.

2. Importance of wetland management

Because of urbanization, economic growth, industrialization, and increasing population, more wastes were discharged into nature. Wetlands carry through some beneficial functions in the protection of whole balance of the nature.

Wetlands are ecologically sensitive systems and provide many significant services to the human population. The evaluation of wetlands with a multidisciplinary perspective in the natural sciences and social sciences provides efficient results. This perspective can give an increased understanding of the processes and problems associated with such strategies. It is clear that wetlands expose noteworthy economic value (depending on the cost-benefit analysis) and they are under severe stress. The reasons for wetland loss and deterioration implicate excessive use, land degradation, urbanization, pollution, climate change, decrease biotic diversity, and invasive species. Since wetlands are complex multifunctional systems, they are likely to be the most beneficial if conserved as integrated ecosystems (within a catchment area) rather than their individual component parts.

Anthropogenic activities (urbanization, water and land uses, land cover changes, industrial activity, pollution, climatic change, etc.) have direct and indirect effects on wetlands. The degradation degree of an ecosystem is depended on temporal variation. Ecosystem recovery level and duration have two main factors. Firstly, anthropogenic pressures can increase or decrease due to the usage grade. Secondly, wetland's carrying capacity is changed due to spatial and temporal variation. For these reasons, positive and negative feedback mechanisms at the wetland are critical control systems. Therefore, the wetland is considered as holistic ecosystem perspective from its basin scale. Odum and Soto–Ortiz [8, 9] concluded that the natural balance is not a steady state and has a homeorhesis. As shown in **Figure 1**, the feedback mechanism occurs to control the wetland ecosystem dynamics. In the natural ecosystems, feedback control processes are repeated between environmental factors and population growth rates in their carrying capacities. However, when the human population intervenes and extremely uses wetlands, this tolerance is destructed, and ecosystems wander off their homeorhesis.

The exponential human population growth reflects why environmental problems appear suddenly [10]. Due to the excess use of wetlands in different ways in time, wetlands have lost

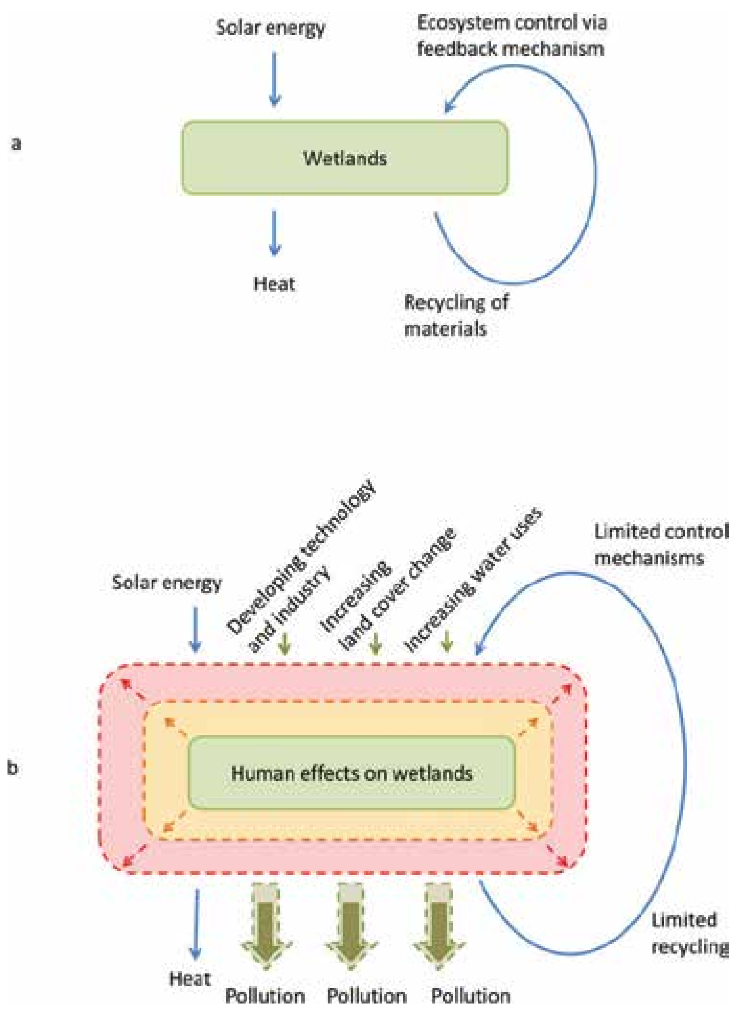

Figure 1. Comparison of natural wetlands (a) and human effects on wetlands (b). Many complex relationships exist in a wetland. Ecosystem feedback control mechanisms play a critical role in the functioning of wetland balance (homeorhesis).

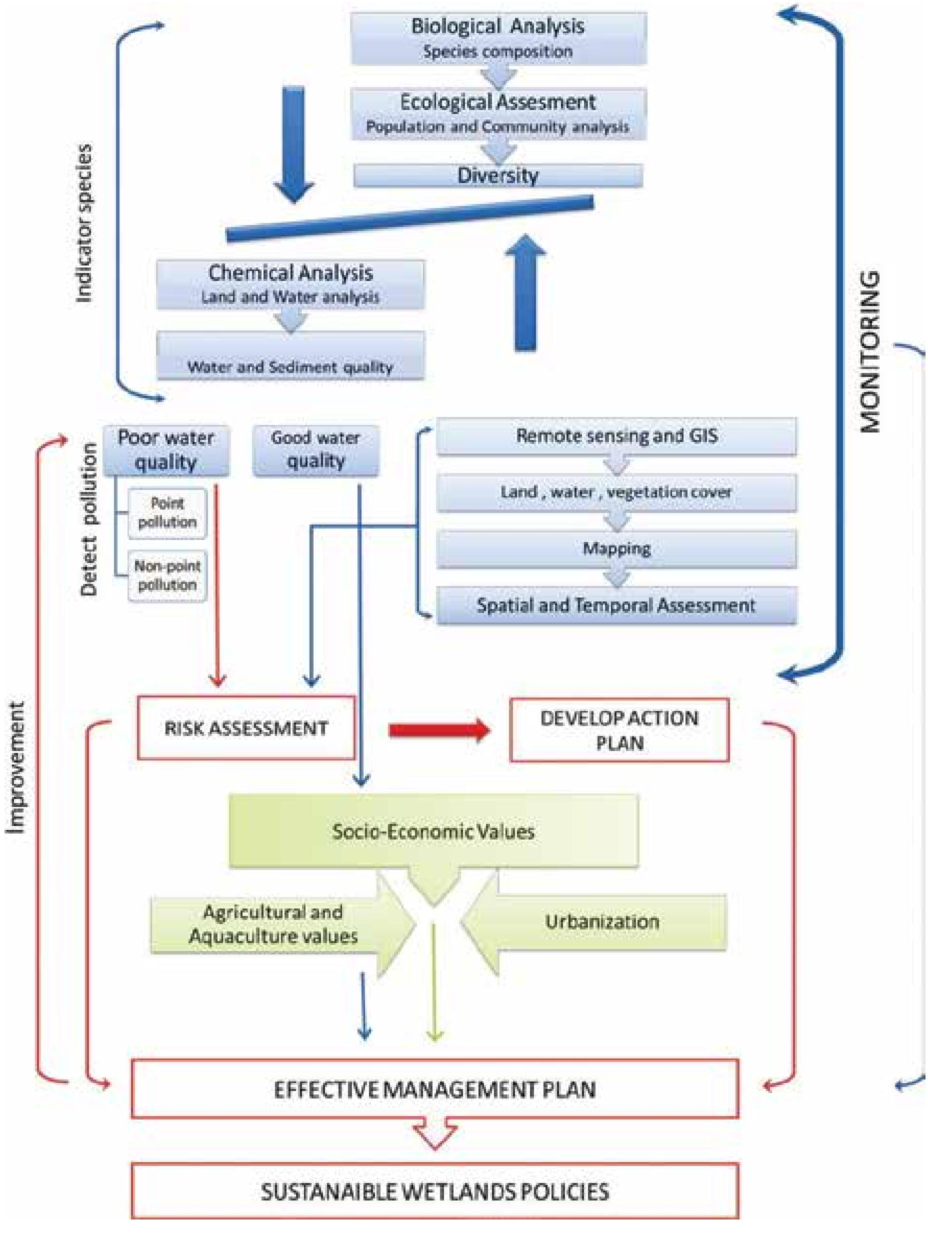

Figure 2. The summary of an effective wetland management process.

completely their natural feedback process. The exponential growth in natural resource utilization and the pollution from industrialization can reach the limits of ecosystems to provide the resource [9–11]. Eighty-seven percent of the wetlands in the world have been lost since 1700. Wetlands have been damaged by anthropogenic sources three times faster than natural forests. Therefore, there are direct and indirect negative impacts on biodiversity and carbon sequestration. Eighty-one percent of inland wetland species and 36% of coastal and marine species have been influenced since 1970 [2].

Wetlands in and around city provide significant services such as water supply and climate regulation [12]. However, the value of these ecosystems remains largely unrecognized by policy- and decision-makers [13].

We need to accept natural balance and geochemical cycles as a wetland ecosystem modulator. Assessment of water quality is classified based on physical, chemical, and biological parameters. While physicochemical characteristics are defined as snapshot industrial pollution, any change in water quality has a controlling effect on integrated community structure. The fauna and flora compositions not only reflect the certain situation of ecosystems but also the previous situation of habitat quality. Bioindicator species that occur according to environmental quality factors are the more reliable assessment for long-term ecological effects in wetland quality. Moreover, biological quality and monitoring give strong evidence for ecosystem problem (**Figure 2**). Water monitoring and assessment develop based on biology, hydrology, and water chemistry. In addition, nowadays, geographic information system and remote sensing data detect any change in the wetland area, vegetation cover, and the water level in spatial and temporal variation and supply crucial information about habitat variations [14–17].

Managing wetland ecosystems gives a substantial contribution to biodiversity conservation and restoration. Also, it may be actualized with a holistically multidisciplinary strategy. The variation of management strategy may be caused by a more different urban wetland area with various levels of success.

It needs decision-makers who are involved in different management strategies to cause restoration and improvement of an ecosystem due to globally ecological and regional economic values of wetlands. Therefore, integrated decision-making process and wetland perspective provide a sustainable ecosystem management and utilization of wetland resources.

Consequently, an effective management plan provides a crucial basis for maintaining the bio-ecological characteristics of a wetland, a dynamic ecosystem, and allowing to use resources economically.

3. Wetlands in the future

Wetlands that may be accepted as ecosystems on edge because of their importance for the future have gained a crucial role to climatic change. Wetland management policies and simulations of their ability to absorb major quantities of carbon from the atmosphere as more than five times from tropical forest show an important solution in future climate [6, 12, 18–21]. It seems clear that wetlands are balanced due to mechanism of geochemical cycles (natural control-feedback mechanism).

As a result of the floods increasing based on climate change, the decrease in drinking water and the increasing human population, the future tasks of wetlands on the negative effects of urbanization are increasing for sustainable urban. It is estimated that at least 64 of the global wetlands have disappeared since 1900 due to cities and exponential human population growth. For this reason, the main mechanism of pollution removal from domestic and industrial wastewater in constructed wetlands will have much importance in their fixation and precipitation capacities [22]. Furthermore, constructed wetland systems would be good alternative technologies in the future, which have wastewater treatment standards as compared to conventional methods [23, 24].

We would also like to stress the great potential that such investigations have in the understanding and protection of these fragile, but extremely important, coastal ecosystems and encourage their incorporation into future wetland management tools.

Wetland degradation usually impacts environmental quality and can lead to major changes in the community composition. Therefore, a recent paradigm that alters within wetland science toward integration of social, all environmental, and life sciences is further appealing to the historical linkage between wetland and special kinds of science today.

Modern wetland science has become a multidisciplinary, interdisciplinary, and sometimes transdisciplinary study that melds the social with the life sciences to understand wetlands as social-ecological systems.

4. Conclusions

The wetland ecosystems have vital values and functions in the world. Human (controlling factor), as an ecosystem stakeholder, benefit from this. Rapidly developing technology enables us to better understand the planet we live in. Due to technological development and increasing human population, all ecosystems are inevitably deteriorated by domestic, agricultural, and industrial pollution; climate change; reducing biodiversity; invasive species; and change of land use. Sustainability includes a greater and more explicitly long-term situation and target than environmental quality increment. Sustainable environmental management depends mainly on ecosystem stability, ecologic tolerance, and biotic diversity. Sustainable environmental management plans need to be implemented and controlled.

Therefore, the role of decision-maker authority is important. The fact that the research institutes and the sciences in different disciplines form a consortium and maintain their management plans with a holistic approach has a critical value in this respect. Due to the different kinds of wetlands and the case study of multidisciplinary approaches in the world, the book *Wetlands Management: Assessing Risk and Sustainable Solutions* can be considered as an important source.

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Section 2

Water Quality and Diversity in Wetlands

Chapter 2

Isotopic Signatures ($\delta^{13}C$ and $\delta^{15}N$) and Characteristics of Two Wetland Soils in Lesotho, Southern Africa

Olaleye Adesola Olutayo

Additional information is available at the end of the chapter

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Abstract

There is sparse data on comparative analysis of soil indicators and isotopic signatures to monitor the health of wetland ecosystems in Lesotho. This study used (i) soil indicators (i.e. soil organic carbon (SOC), soil organic carbon density, and silt:clay ratio) and (ii) isotopic signatures ($\delta^{13}C$ and $\delta^{15}N$) to monitor environmental change aquatic ecosystems of Lesotho. Transects of 2000 m were chosen in two agro-ecological zones (AEZ) (Lowlands and Mountains) of Lesotho and sub-divided into upper (US), middle (MS) and toe slopes (TS). Soil samplings were made horizon-wise (1.20 m deep) in triplicates, labeled and shipped to the laboratory in plastic bags. Aquatic vegetation samples were randomly collected along these transects for stable isotopes. All samples analyzed using standard procedures. Results showed that wetlands located in the Lowlands (Ha-Matela) AEZ were much more degraded and heavily impacted. This indicated by low silt/clay ratios, low SOC contents and SOC density and less negative $\delta^{13}C$ compared to that of Mountains AEZ (Butha Buthe). Thus, these indicators can be used to predict degradation of wetlands. However, the severity of degradation, can be easily predicted the $\delta^{13}C$ values and $\delta^{13}N$ served as a robust indicator of wetland eutrophication. These results showed that soil indicators used as well as stable isotopes signatures used (i.e. $\delta^{13}C$ and $\delta^{13}N$) may be used as monitoring tools for wetland management and restoration.

Keywords: Lesotho, organic carbon, stable isotopes, South Africa, wetlands soils

1. Introduction

The kingdom of Lesotho is a small landlocked country in South Africa with a population of about 1.8 million [1] and occupies a total land area of 30,350 km^2 [2] and has four distinct agro-ecological zones (AEZ) based on the geology and climate (**Table 1**) [3] and has 10 districts (**Figure 1**).

Wetlands are among the Earth's most productive ecosystems. The significance of wetlands lie in their roles in the hydrological cycle, for flood and biomass production, as refuge for wildlife, biogeochemical functions, as nutrient and pollution filters for water quality improvement among others [4]. Globally, large percentage of these lands have been lost due to drainage and land clearance as consequence of agricultural, urban and industrial development activities [5–8]. According to Barbier et al., [9], the features of wetlands system can be grouped into *components, attributes* and *functions*. The *components* are the biotic and non-biotic features such as soil, water, plants and animals, while the *attributes* relate to the variability and diversity of these components e.g. diversity of species. However, the interactions between the components are expressions of the *functions* of the system such as nutrient cycling, water flow/exchange dynamics between the atmosphere (rainfall), the surface water and the shallow groundwater system. However, influence of agricultural land-use activity and hydrological modifications (affecting a biotic factor) are said to affects the attributes and functions of wetlands ecosystem [12–15].

Agriculture and wetlands has not had a very harmonious relationship in the past and agricultural activities have been affecting ground and surface water quality adversely from both point and non-point sources [10–12]. In Lesotho, wetlands are called *mekhuabo,* which apart from serving as refuge for wildlife, are primarily utilize to sustain agricultural activities at the local communities. These ecosystems support more than 300,000 households through agriculture and livestock watering. The wetlands ranged from several square meters to several square kilometers and occur in all the AEZs [13–15]. They can be categorized under three broad categories: palustrine, lacustrine and riverine [13, 16]. The palustrine wetlands are the dominant type and these include mires (bogs and fens), most of which are found at high altitude, at valley heads and at the upper reaches of rivers [8–12]. The lacustrine on the other hand occupies land area of ≥0.41 ha and comprises of artificial impoundments for water supply and soil conservation works (e.g. Katse and Mohale dams). The riverine wetlands are found along the river systems and these are generally small and often localized. In the recent years, there have been threats to wetlands across all the four AEZs [3, 22]. Threats to wetlands in Lesotho are attributable to over grazing, livestock watering; weed infestation, agricultural runoff and eutrophication, land reclamation for agricultural uses, and sedimentation of wetland beds [16, 23].

Agro-ecological zones	**Area (km^2)**	**Altitude (m)**	**Topography**	**Mean annual rainfall (mm)**	**Mean annual temperature (°C)**
Lowland	5200	<1800	Flat to gentle	600–900	−11 to 38
Senqu river valley	2753	1000–2000	Steep sloping	450–600	−5 to 36
Foot-hills	4588	1800–2000	Steep rolling	900–1000	−8 to 30
Mountains	18,047	2000–3484	Very steep bare rock and gentle rolling valleys	1000–1300	−8 to 30

§Source: State of the Environment in Lesotho [9, 10].

Table 1. Agro-ecological characteristics of Lesotho§.

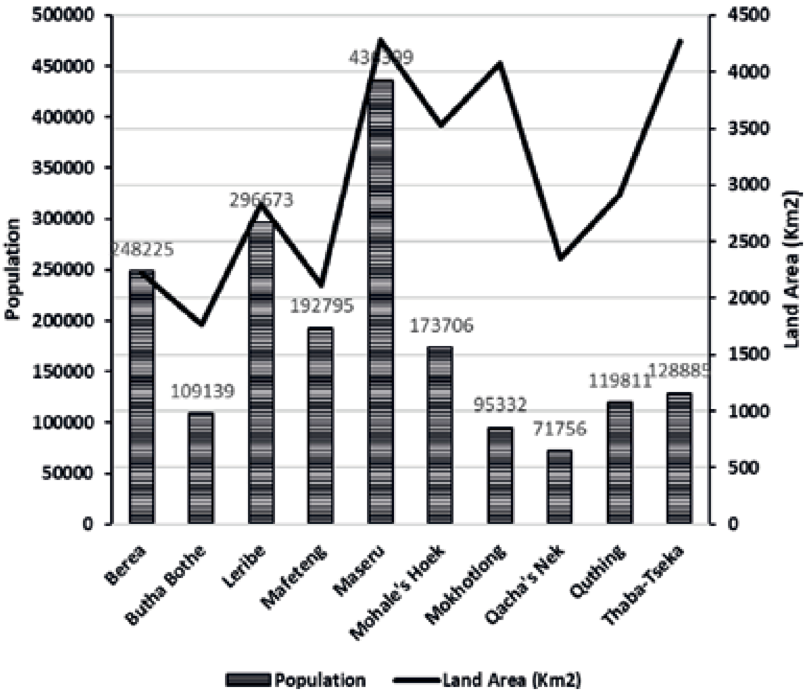

Figure 1. Population and land area in 10 districts of Lesotho.

Current indicators of wetland monitoring often examine nutrient loadings such as soil and water total phosphorus (TP) and total nitrogen (TN) concentrations, species composition, biomass and primary production. These indicators often show the changes that have taken place on the impacted systems [24–26], but these have the shortcoming of identifying early ecosystems disturbance. However, the need for early and timely identification of systems of ecosystems disturbance is critical these days [27, 28]. Stable isotopes of carbon and nitrogen in organic matter offers an alternative means to detect early signs of environmental changes in aquatic ecosystems [29–32]. The ratios of $^{13}C/^{12}C$ and $^{15}N/^{14}N$ (defined as $\delta^{13}C$ and $\delta^{15}N$) has been used to provide insight into the sources, sinks and cycling of carbon and nitrogen in aquatic ecosystems as these biota interact with its physical and chemical environments [33–36]. This study aimed at comparing the characteristics of wetland soils in the Lowland and Mountains AEZ in terms of soil characteristics, and compare the isotopic signatures ($\delta^{13}C$ and $\delta^{15}N$) in these wetlands thereby understand the responses and mechanisms controlling the isotope variation in these wetlands. The ultimate goal is to identify causes of mismanagement and suggests plausible management options for a sustained and continuous use of these fragile lands for ecosystems services and agriculture.

2. Methods

The study was conducted on two wetlands located separately in two AEZs of Lesotho namely the Mountains (Butha-Buthe) and the Lowlands (Ha-Matela) (**Figure 2**). Butha-Buthe:

The wetland in Butha-Buthe is a palustrine wetland [13] and it is situated in the Mountain AEZ. It is located at an altitude/elevation of between 3181 and 3202 m above sea level (asl) and at points Latitude 28° 53.821/Longitude 28° 47.993 E. The site falls within the Afroalpine Grassland zone characterized by grasses-*Festuca caprina, Merxmuellera disticha* and *Pentaschistis oreodoxa;* shrubs and woody plants—*Chrysocoma ciliate, Erica dominans* and *Euryops evansii;* and other flowering plants—*Kniphofia caulescens, Helichrysum trilineatum, Dierama robustum, Zaluzianskaya ovate* and *Dianthus basuticus var. grandiflorus* [13]. Ha Matela: Ha Matela wetland is a Riverine wetland situated in the Foothills AEZ at an elevation of 1820 m above sea level, at points; Latitude: −29°38.3333/Longitude: 27°76.6667. It is characterized as the Afromontane Grassland zone. Dominant grasses includes: *Themeda triandra, Festuca caprina, Merxmuellera macowanii* and *Eragrostis curvula;* trees and shrubs: *Salix mucronata, Rhus erosa, Rhus pyroides, Leucosidea sericea, Myrsine Africana, Rhoicissus tridentate, Buddleja loricata* and *Chrysocoma ciliate* and flowering plants: *Gladiolus (several species), Kniphofia (several species), Helichrysum (many species), Agapanthus campanulatus subsp. Patens, Dierama robustum, Euphorbia clavarioides* and *Aloe polyphyll*. The geology of Lesotho is called formation [37] with sedimentary and volcanic clastics. Wetlands in these two agro-ecological zones: the Mountains and Lowlands (**Table 1**) were characterized as low, medium or high impacted wetlands based on local (i) land-use characteristics and (ii) intensity of anthropogenic pressures such as mining, smelting and discharge of industrial pollutant into the wetlands [38]. According to [38], the low impacted wetlands has little (i.e. <5%) or no agricultural activity within 150 m of the wetland boundary. Secondly, wetlands that were classified as highly impacted had agricultural activities; within 10 m of wetland boundary (i.e. < 33% of the wetland area is impacted). The medium impacted wetlands had agricultural activities between 5 and 32% of the wetland boundary. Wetlands in the Lowlands AEZ (i.e. Ha-Matela) were classified as being highly impacted, while that in the Mountains (i.e. Butha Buthe) had little impacts after [38]. About 2000 m transects were chosen and divided into upper (US), middle (MS) and toe slope (TS). Profile pits (1.20 m) were dug

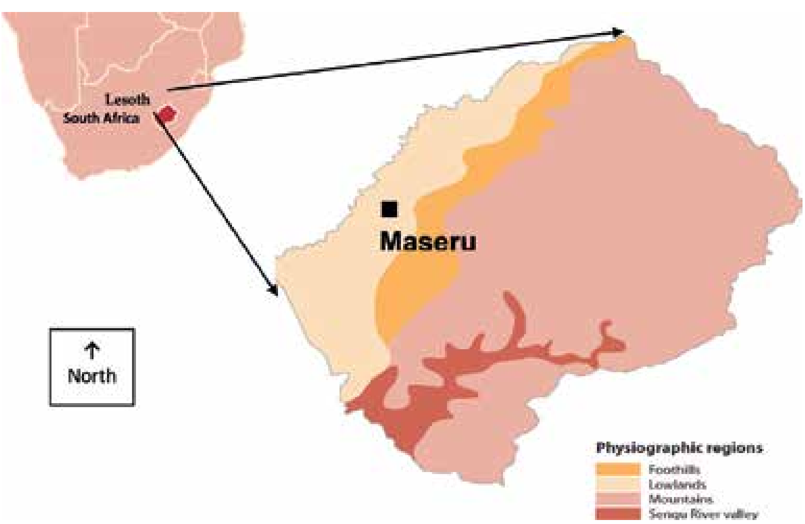

Figure 2. The location of Lesotho within South Africa and its four agro-ecological zones.

to reveal the natural soil horizons. Samplings were made in triplicates using the natural soil horizons. Soil samples were placed inside labeled plastic bags and shipped to the laboratory. Soils collected were analyzed after the standard methods:pH water (1:2 soil-water ratio) and pH-KCl (1:1 soil-water ratio), particle size analysis [39], total N [40] and available P (Bray-1-P) [41], the organic carbon (OC) [42], and the SOC pool [43] and the equation:

$$\text{C-pool} = d \times BD \times organic\ \ carbon \tag{1}$$

where C-pool (kgC m^{-2}), d: soil layer thickness (m), BD: bulk density (kg m^{-3}), organic carbon (g g^{-1}). The base cations (Ca, Mg, Na and K) were by extracting soils with 1 N NH_4OAc (pH 7) and these were determined atomic absorption spectrophotometer (Perkin Elmer, 2007 AAS model WinLab) and flame photometers. Plant samples for isotopic signatures (i.e. $\delta^{13}C$ and $\delta^{15}N$) in these wetlands were randomly collected in duplicates from the US, MS and TS sections of the toposequence/topography across years (2008–2010). These were labeled, air-dried, and shipped to the Soil and Water Management and Crop Nutrition Laboratory, of the International Atomic Energy Agency (IAEA), Seibersdorf, Austria. The results are reported in standard δ notation as $\delta^{13}C$, $\delta^{15}N$, %C and %N values in reference to the international standards Vienna Pee Dee Belemnite (V-PDB) and air N_2 respectively. Analytical precision was ±2‰ for both $\delta^{13}C$ and $\delta^{15}N$ based on repeated analyses of laboratory standards. All data collected were subjected to analysis of variance (ANOVA) using the general linear model procedure (PROC GLM) of Statistical Analysis Systems (SAS) [44]. Means were separated using Duncan multiple range test (DMRT) at 5%.

3. Results and discussion

Generally, most of the wetlands across all agro-ecological zones of Lesotho are either used for livestock watering, grazing and agriculture and drinking water. In a related study on comparative assessments of wetlands in West and Southern Africa, it was found that most of the rural population used the wetlands largely for grazing and watering (**Figure 3**). It is evident from this result that approximately, 21% respectively of the population considered wetlands being important for irrigation and livestock grazing and watering. Similar observations were made by researchers from Southern Africa [45, 46], Taznania [47] and Kenya [45]. These authors found that wetlands constituted an important area of the livelihoods of the rural people. Hence, one of the major constraints to the sustainable use of wetlands in Lesotho and Africa in general is the lack of information on the diverse benefits that can be obtained from wetlands if properly managed. Hence, this information is needed by the government planners, natural resource managers and local communities.

A close observation of the soil physico-chemical properties of these wetlands is shown in **Table 2**. Results showed that the particle size distribution (i.e. texture) of the wetland soils at Butha Buthe was dominated by sand size texture compared to that at Ha-Matela. At the latter site, the particle size distribution had almost equal proportions of sand, silt and clay sized particles (**Table 2**). Both wetland soils generally had acidic soil pH (i.e. 4.69–5.44),

Transects	Position	Sand (%)	Clay (%)	Silt (%)	pHw	pHKCl	AVP (mg kg^{-1})	SOM (%)	Ca (cmol kg^{-1})	K (cmol kg^{-1})	Na (cmol kg^{-1})	CEC (cmol kg^{-1})
	Butha Buthe (Mountains AEZ)											
1	US	67.16	12.04	20.81	5.44	4.69	1.40	4.49	17.51	0.05	3.17	5.23
1	MS	63.99	10.17	25.81	5.26	4.60	2.71	5.25	15.00	1.49	5.38	2.07
1	TS	62.95	7.32	29.53	5.33	4.66	2.23	4.58	15.31	0.05	2.89	0.05
2	US	66.65	5.42	27.93	5.00	4.47	3.29	8.38	10.44	0.04	4.57	4.82
2	MS	78.21	5.08	16.71	4.69	4.31	2.42	8.00	11.28	0.04	2.35	5.97
2	TS	70.84	7.40	21.72	4.95	4.49	3.04	7.74	17.06	0.05	4.69	4.92
	Ha-Matela (Lowlands AEZ)											
1	US	24.66	33.33	42.83	5.42	4.46	4.11	4.54	0.63	0.49	0.09	0.18
1	MS	40.78	23.28	36.00	5.02	4.39	3.21	3.74	0.28	0.34	0.09	0.17
1	TS	34.12	36.81	30.58	5.12	4.37	2.83	3.35	0.38	0.47	0.10	0.17
2	US	38.37	36.97	25.33	5.69	4.72	3.30	2.94	1.28	0.37	1.15	0.17
2	MS	39.23	33.91	27.08	5.06	4.56	2.56	4.32	1.03	0.41	0.53	0.17
2	TS	44.97	29.30	26.56	5.25	4.64	3.13	3.90	1.06	0.38	0.25	0.17

US, upper slope; MS, mid-slope; TS, toe slope; AVP, available P; SOM, soil organic matter; CEC, cation exchange capacity; pHw, pH in water.

Table 2. Physical and chemical properties of wetlands in Butha Buthe and Ha-Matela, Lesotho.

low available P ranging between 1.40 and 3.29 mg kg^{-1} (Butha Buthe) and between 2.94 and 4.54 mg kg^{-1} (Ha-Matela). Some researchers had associated phosphorus mineralization in wetland soils was associated negatively with acidic soil pH and coarser soil texture [49–51]. The soil organic matter across both wetland types was relatively high. Higher exchangeable Ca (10.44–17.51 cmol kg^{-1}) was noted in Butha Buthe wetlands as opposed to very low contents observed in the Ha-Matela soils (i.e. 0.28 cmol kg^{-1}). Wetland soils in Butha Buthe had higher bulk density (BD) (i.e. 1.24–1.55 g cm^3) compared to Ha-Matela wetlands (i.e. 1.32–1.38 g cm^3). The higher BD in the former compared to the latter might be attributed to higher sand contents (**Figure 4**). The ratio of silt and clay—called silt:clay ratio—is an index of soil age and the ease of erodibility [52]. Lower ratio of between 0.43 and 1.99 (Ha-Matela) compared to 1.10 and 9.89 (Butha Buthe) is an indication that wetland soils in the former site are older and would be easily eroded compared to the latter (**Figure 5**). This was in agreement with the findings of some researchers that lower silt/clay ratio is an indication of high degree of erosion [52, 53]. Higher SOC contents were observed in the Butha Buthe wetlands

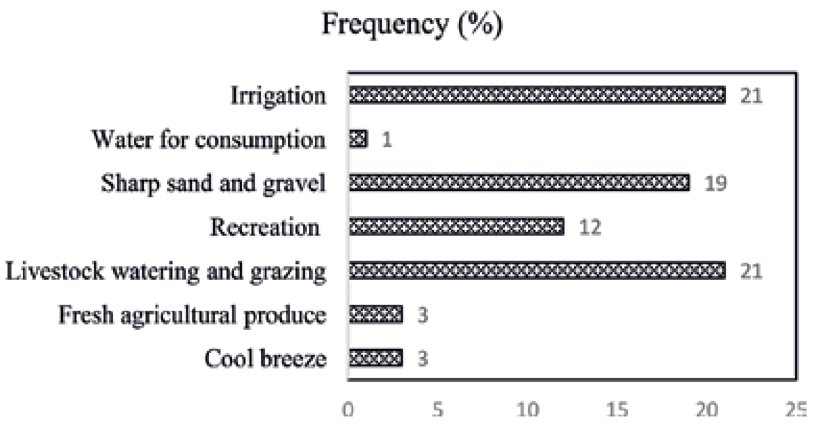

Figure 3. Utilization of wetlands in the Lowlands AEZ of Lesotho.

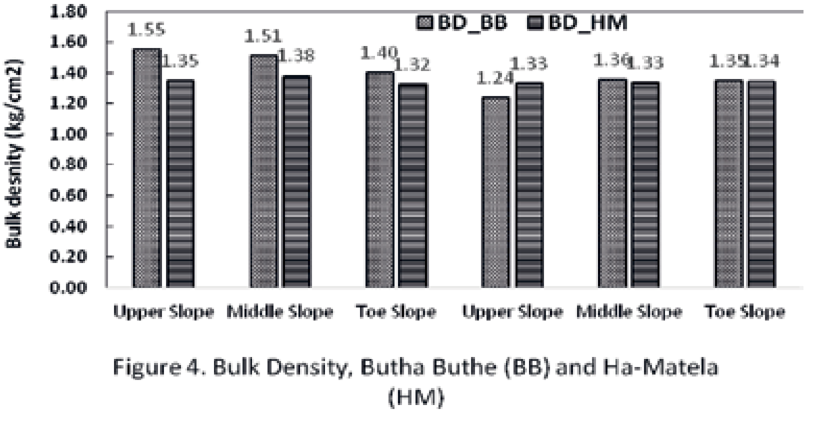

Figure 4. Bulk density, Butha Buthe (BB) and Ha-Matela (HM).

compared to the Ha-Matela wetlands (**Figure 6**). The high SOC is related to the balance of input from net primary production and microbial decomposition and the decomposition rates in wetlands are generally low due to low availability of oxygen and low temperatures [54].

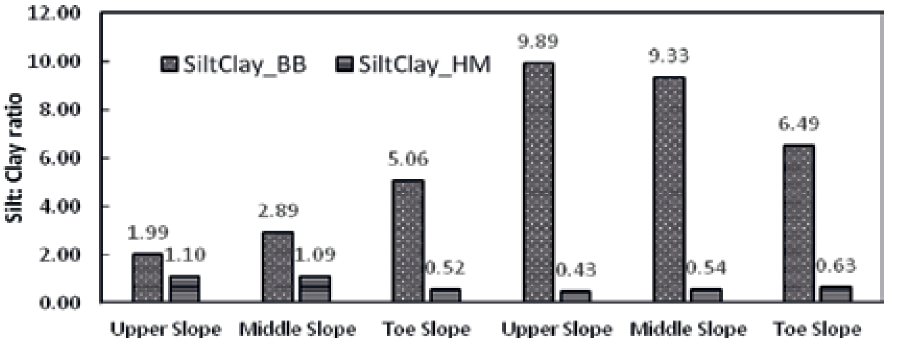

Figure 5. Silt clay ratio, Butha Buthe (BB) and Ha-Matela (HM).

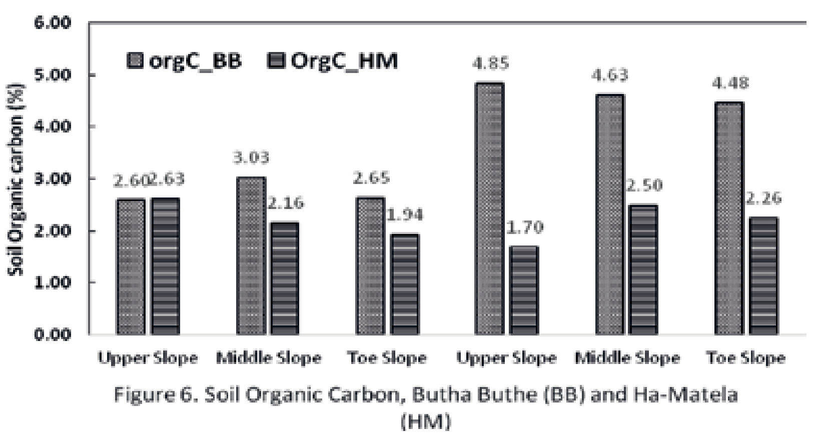

Figure 6. Soil organic carbon, Butha Buthe (BB) and Ha-Matela (HM).

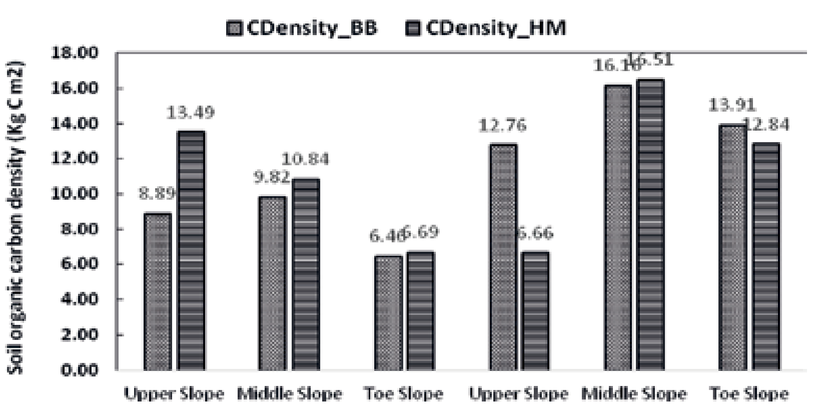

Figure 7. Soil organic carbon density, Butha Buthe (BB) and Ha-Matela (HM).

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Thus, one of the reasons for higher SOC in Butha Buthe wetlands is due to high altitude (i.e. 2000–3483 m) and low temperature (i.e. ≤8°C) in winter periods. Furthermore, the SOC density was observed in the Butha-Buthe wetlands (6.69–16.51 kgC m^{-2}) compared to that in the Ha-Matela (6.46–13.91 kgC m^{-2}) (**Figure 7**). These results showed that wetland soils in the former site are much more stable and would not be easily eroded. Serval authors had attributed higher soil organic carbon density to several factors and these includes type of land use and soil management practices and these can significantly influence soil organic SOC dynamics and C flux from the soil [22, 55–59]. The vegetation isotopic $\delta^{13}C$ and $\delta^{15}N$ across the

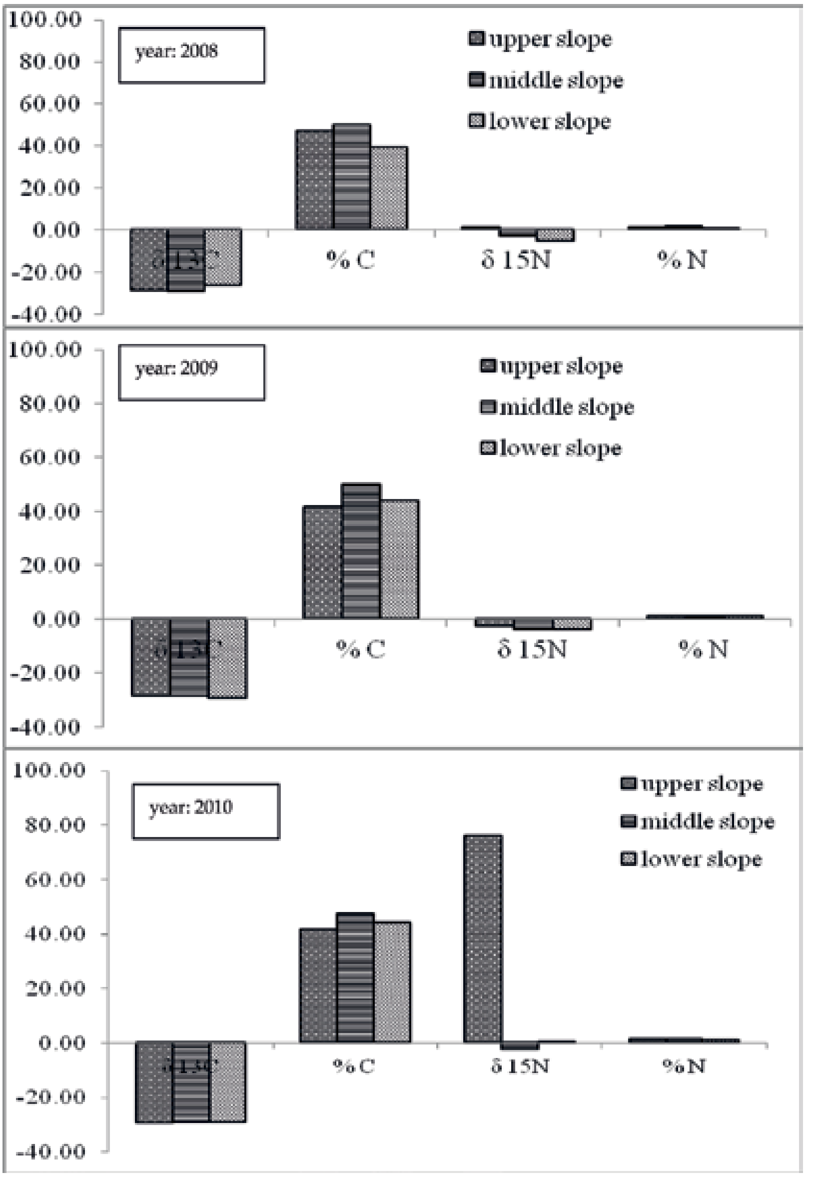

Figure 8. Isotopic $\delta^{13}C$, %C, $\delta^{15}N$ and % N of vegetation, Butha Buthe.

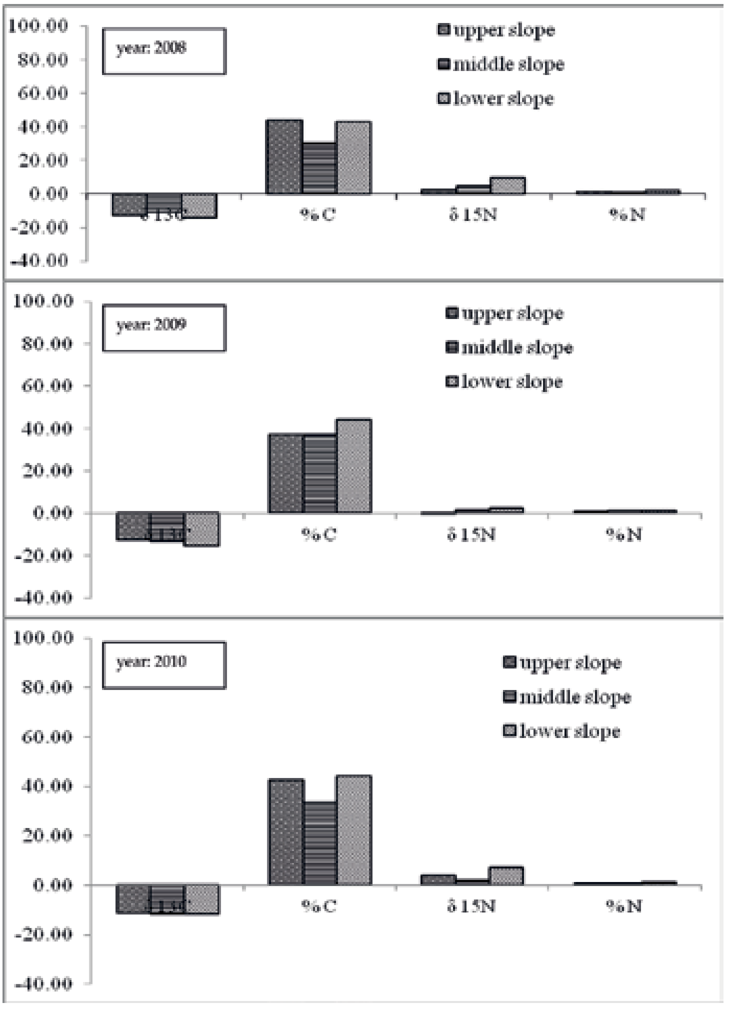

Figure 9. Isotopic δ^{13}C, %C, δ^{15}N and % N of vegetation, Ha-Matela.

two wetlands and years (2008–2010) are shown in **Figures 8** and **9**. The less negative values of Isotopic δ^{13}C (Ha-Matela), compared to Butha Buthe is an indication of degradation [29, 36, 60]. High δ^{15}N in Butha Buthe is ascribed to nutrient enrichment as a result of anthropogenic activity (i.e. livestock grazing [61].

4. Conclusions

Human influences have led to disturbances in the wetland ecosystems in Lesotho. The study showed that despite the fact that soil characteristics can be used to assess changes in the ecosystems, environmental isotopes of C and N in aquatic plants responded positively to nutrient increase due to δ^{13}C values in plants. Results showed wetlands located in the Lowlands (Ha-Matela) AEZ are much more degraded and heavily impacted as indicated by low base cations (K, Ca, Mg and Na), lower silt/clay ratios as well as lower SOC contents and SOC density, higher bulk density and less negative $\delta^{13}C$ compared to that of Mountains AEZ (Butha Buthe). However, the severity of degradation, can be shown by the $\delta^{13}C$ values as these values are sensitive indicators of nutrient stress and $\delta^{13}N$ served as a robust indicator of wetland eutrophication. These results showed that soil indicators used as well as stable isotopes signatures used (i.e. $\delta^{13}C$ and $\delta^{13}N$) may be used as monitoring tools for wetland management and restoration.

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Conflict of interest

No conflict of interests.

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Chapter 3

Metals Pollution in Tropical Wetlands

Ong Meng Chuan and Kamaruzzaman Yunus

Additional information is available at the end of the chapter

http://dx.doi.org/10.5772/intechopen.82153

Abstract

Metals pollution has drawn worldwide attention due to increase of anthropogenic contaminants to the coastal area, especially wetlands area. Metals are indestructible and have toxic effects on living organisms. Sediment can act as an indicator of metals pollution due to the ability of the sediment that can trap metals through complex physical and chemical process. Therefore, they are always used as geo-marker for identifying the possible source of metals pollution. Besides that, wetlands such as mangrove have a diverse diversity of organisms that provide proteins to local communities such as clam, oyster, crab, and fishes. Therefore, it is important for us to know the levels of metals in the sediment and those organisms that we consume nowadays that live at the mangrove area. Such findings can provide important information on the seafood safety level and potential impact especially to humans via consumption according to the provisional tolerable weekly intake and daily intake.

Keywords: metals, sediments, geo-marker, organisms, permissible level

1. Introduction

Wetlands ecosystem such as mangrove ecosystem can be defined as the interface between land and sea in tropical and sub-tropical latitude where the mangrove plant can survive in conditions of high salinity, strong winds, extreme high and low tides, high temperature, and anaerobic muddy soils (**Figure 1**). This well-developed morphological and physiological adaptation to these extreme conditions is not present in other groups of plants [1]. Due to these extreme conditions, mangrove ecosystem is rich in biodiversity and constitutes a unique fauna and fauna, above the sediment and underneath the sediment.

Figure 1. Tropical wetlands ecosystem in Malaysia. Photo by Ong Meng Chuan.

Figure 2. Tropical mangrove ecosystem that can be found in Malaysia coastal. Photo by Mokhtar Ishak.

Mangrove forests such as *Rhizophora* sp. (**Figure 2**) are important ecosystems ecologically and economically toward human beings and organisms that live in the mangrove area. These forests provide breeding and feeding ground for various aquatic organisms such as fishes, shellfishes, reptiles, and some land organisms such as monkeys and snakes. For example, some fishes such as sea bass, the juvenile will stay in this mangrove area before they move to the ocean when they were adult. Besides that, mangrove forest also plays an important role in protecting shorelines from erosion or in some places, minimizing the strong current from tsunami. This protection indirectly can protect the communities that live in coastal area.

2. Metals pollution

Unlike other pollutants, which may be visibly buildup in the environment, trace metals in the environment may accumulate unnoticed to toxic levels. These metals pollutants in the aquatic environment can come from natural or anthropogenic sources. Metals are serious pollutant in our natural environment due to their toxicity, persistence, and bioaccumulation problems

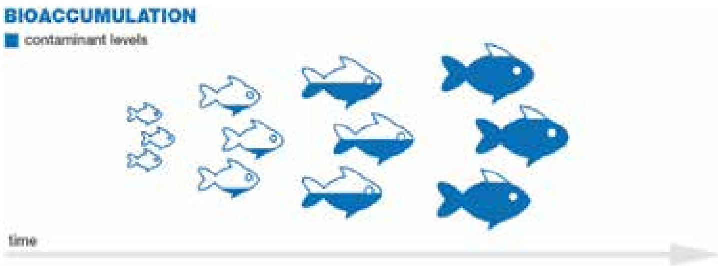

Figure 3. Bioaccumulation process of metals concentration in fish. Picture adapted from http://www.hydro-industries.co.uk/case-studies.htm?id=10 [2].

(**Figure 3**). Some are highly toxic and persistent, and have a strong tendency to become concentrated in marine food webs. Excess of these metal levels in aquatic environment may pose a health risk to humans and to the environment.

Organisms require certain trace amounts of some metals, including cobalt, copper, iron, manganese, and zinc in their growth process. Excessive levels of essential metals in the environment, however, can be detrimental to the organism itself. Besides that, nonessential metals of particular concern to surface water systems are cadmium, chromium, mercury, lead, and arsenic, and these metals have no biological function. Metals pollution in aquatic environment can be categorized into four major groups [3] according to their pollution potential:

i. Very high pollution potential—Ag, As, Cd, Cr, Cu, Hg, Pb, Sb, Sn, Te, and Zn

ii. High pollution potential—Ba, Bi, Ca, Fe, Mn, Mo, Ti, and U

iii. Moderate pollution potential—Al, Au, B, Be, Br, Cl, Co, F, Ge, K, Li, Na, and Ni

iv. Low pollution potential—Ga, I, La, Mg, Nb, Si, Sr, Ta, and Zr

3. Geochemical mapping

Distribution of metals in surficial sediments from industrial effluents and urban sewage discharged into the wetlands ecosystem and aquatic environment without proper cleaning can easily be identified through metals spatial variations in sediments. Geochemical mapping can be used as a tool for visualization, which enhanced by computer-aided modeling using geographical information system (GIS) to make it easier to identify the possible locations of contaminated area. Nowadays, due to the rapid developments of computer technology, GIS applications are receiving increasing interest in environmental geochemistry study [4, 5]. It is becoming increasingly popular to incorporate digitized and computerized technologies in studies of marine environmental pollution. These technologies may include GIS and global positioning system (GPS) in the interpretation and presentation of data and in geochemical modeling (**Figure 4**).

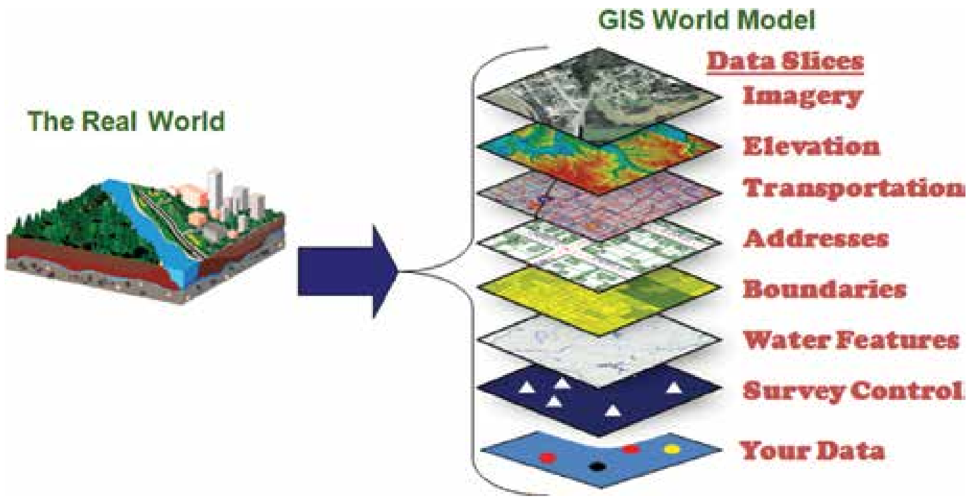

Figure 4. Example of geographical information system (GIS) mapping in environmental studies. Photo adapted from https://technofaq.org/posts/2017/07/thoughts-on-the-future-of-gis-what-will-change-in-50-years/ [6].

GIS is a tool for decision making, using information stored in a geographical form. Some researchers defined major requirements and functions of GIS and mentioned spatial data handling tool for solving complex geographical problems [7–9]. Besides, GIS is increasingly used in environmental pollution studies because of its ability in spatial analysis and interpolation,

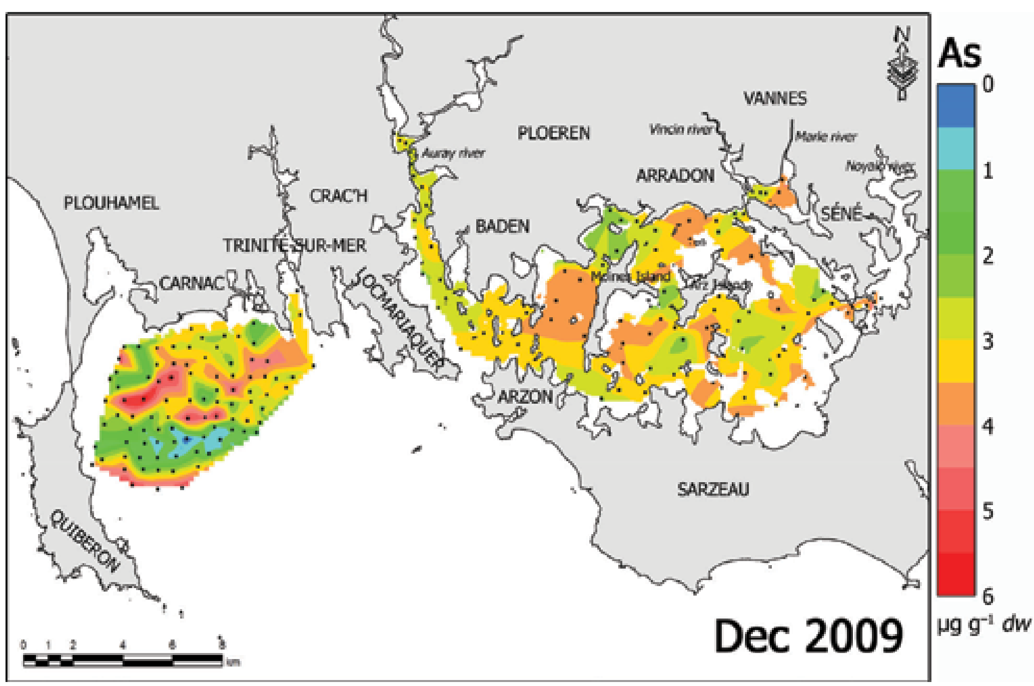

Figure 5. Concentration of Arsenic (As) in sediment of South Brittany waters (Bay of Quiberon and Gulf of Morbihan), France. Figure by Ong Meng Chuan using ArcGIS software 9.3.

and spatial interpolation utilizes measured points with known values to estimate an unknown value and to visualize the spatial patterns [10, 11]. For example, **Figure 5** shows the concentration map of Arsenic in surficial sediment from South Brittany waters analyzed by using ArcGIS software 9.3.

4. Sediment as geo-marker

Sediments are widely used as geo-markers for monitoring and identifying the possible sources of pollution in the coastal environments since sediments are the main sink for various pollutants (**Figure 6**). Sediments can serve as a metal pool that can release metals to the overlying water via natural or anthropogenic processes, causing potential adverse health effects to the ecosystems. Most metals are bound in the fine-grained fraction (<63 μm), mostly because of its high surface area-to-grain size ratio and humic substance content, where they have a potentially greater biological availability than those in the larger (2 mm–63 μm) sediment fraction.

Meanwhile, sediment cores (**Figure 7**) can provide chronologies of contaminant concentrations and a record of the changes in concentration of chemical indicators in the environment. Metal accumulation rates in sediment cores can reflect variations in metal inputs in a given system over long periods of time. Hence, the study of sediments core provides historical record of various influences on the aquatic system by indicating both natural background levels and the man-induced accumulation of metals over an extended period of time.

Figure 6. Different types of sediment can be collected from wetlands ecosystem. Photo by Ong Meng Chuan.

Figure 7. Core sample collected from mangrove environment used for metals proxy study. Photo by Ong Meng Chuan.

5. Assessment of sediment pollution status

To evaluate the metals contamination in sediment, determined element concentrations were compared with background concentrations. Literature data on average world shale or sediment cores or sediments from pristine such as undisturbed wetlands, non-industrialized regions were analyzed to establish the background values. However, to reduce the metals variability caused by the grain sizes and mineralogy of the sediments, and to identify anomalous metals contribution, geochemical normalization has been used with various degrees of success by employing conservative elements [12, 13]. Various elements have been proposed in the literatures to be clay mineral indicators and hence to have the potential for the environmental studies. Some of them are lithium, Li [14–16]; aluminum, Al [17, 18]; scandium, Sc [19]; cesium, Cs [20, 21]; cobalt, Co [22]; and thorium, Th [23, 24]. Among above conservative elements, Li and Al have been widely applied in wetlands and mangroves study [25–27]. Li also has been proposed by Loring [14] as an alternative for Al in high latitude areas in Western Europe and North America. Alternatively, Li meets the basic criteria for use as a normalizing element for metals pollution [14] because of several factors, namely, it is a lattice component of fine-grained major trace-metal-bearing minerals such as the phyllosilicates and clay minerals; it reflects the granular variability of its host mineral component, and it is a conservative element.

The absolute concentration of metals in marine sediments never indicates the degree of contamination coming from either natural or anthropogenic sources because of grain-sizes distribution and mineralogy [26, 28, 29]. Normalization of metals concentrations to grain sizes, specific surface area and reactive surface phases such as Li and Al is a common technique to remove artifacts in the data due to differences in depositional environments [30–34]. This allows for a direct comparison to be made between contaminant levels of samples taken from different locations. One of the most common normalization techniques is converting trace metal concentrations to enrichment factors (EF) by normalizing metals concentrations to a common element (usually Al or Fe) [35–37]. The EF value can be calculated according to the following formula:

$$\text{Enrichment Factor (EF)} = \frac{(\text{Metal concentration/Normalizer})_{\text{sample}}}{(\text{Metal concentration/Normalizer})_{\text{background}}}$$

Based on the researches by several geochemists [38–41], if an EF value is between 0 and 1.5, it is suggested that the metals may be entirely from crustal materials or natural weathering processes. If an EF is greater than 1.5, it is suggested that a significant portion of metals have arisen from noncrustal sources or anthropogenic pollution [24, 42].

Another commonly used criterion to evaluate the heavy metals pollution in sediments is the index of geoaccumulation (I_{geo}) originally introduced by Muller [43] in order to determine and define heavy metals contamination in sediments by comparing current concentrations with the background levels. Similar to metal enrichment factor, I_{geo} can be used as a reference to estimate the extent of metal pollution in sediments. The I_{geo} is defined by the following equation:

$$I_{geo} = \log_2 (C_n/1.5B_n)$$

where C_n is the measured concentration of the examined element (n) in the sediment and B_n is the geochemical background concentration of the element (n). Factor 1.5 is the background matrix correction factor due to the lithogenic effects [43]. The upper continental crust values of the metals of interest are the same as those used in the aforementioned enrichment factor calculation [44]. Muller [43] has distinguished seven classes of the I_{geo} from Class 0 to Class 6. The highest class (Class 6) reflects at least 100-fold environment above the background value (**Table 1**).

Tomlison et al. [45] elaborated that the application of pollution load index (PLI) provides a simple way in assessing mangrove, estuarine, and coastal sediment quality. This assessment is a quick tool in order to compare the pollution status of different places [46]. PLI represents the number of times by which the metal concentrations in the sediment exceed the background concentration, and give a summative indication of the overall level of metals toxicity in a particular sample or location [47, 48]. The PLI can provide some understanding to the public of the surrounding area about the quality of a component of their environment, and indicates the trend spatially and temporarily [49]. In addition, it also provides valuable information to the decision makers toward a better management on the pollution level in the studied region.

PLI is obtained as contamination factors (CFs). This CF is the quotient obtained by dividing the concentration of each metal with the background value of the metal. The PLI can be expressed from the following relation:

$$PLI = (CF_1 \times CF_2 \times CF_3 \times CF_4 \times CF_n)^{1/n}$$

where, n is the number of metals studied and the CF is the contamination factor. The CF can be calculated from:

$$CF = (\text{Metals concentration in samples/Background metals concentration})$$

Class	Value	Sediment quality
0	$I_{geo} \leq 0$	Practically uncontaminated
1	$0 < I_{geo} < 1$	Slightly contaminated
2	$1 < I_{geo} < 1$	Moderately contaminated
3	$2 < I_{geo} < 1$	Moderately to heavily contaminated
4	$3 < I_{geo} < 1$	Heavily contaminated
5	$4 < I_{geo} < 1$	Heavily to extremely contaminated
6	$5 < I_{geo} < 1$	Extremely contaminated

Table 1. Classification of sediment quality based on I_{geo} value.

The PLI value more than 1 can be categorized as polluted whereas less than 1 indicates no pollution at the study area [50, 51].

6. Aquatic organisms as biomarker

Lying in the second trophic level in the aquatic ecosystem, shellfish species have long been known to accumulate both essential and nonessential metals. Many researchers have reported the potentiality of using mollusks, especially mussel and oyster species, as bio-indicators or bio-markers for monitoring the metals contamination of the aquatic system (**Figure 8**). Beside as a bio-marker for marine pollution studies, mollusks species also been used in ecotoxicology and toxicity studies. Individual bio-monitors respond differently to different sources of bioavailable chemical elements for example, in the solution, in sediments, or in foods. To gain a complete picture of total metals bioavailability in a marine habitat, it is necessary, therefore, to use a correct bio-monitor that can reflect the element bioavailability in all available sources [52]. Such comparative use of different bio-monitors should allow the identification of the particular source of the contaminant elements [53].

Living organisms in aquatic environment can transport pollutants and contaminants into, within, and out of the marine aquatic ecosystem. These organisms can ingest the pollutants via water and food, and inhale them as they breathe and feed [54]. Once in the body, some contaminants pass quickly while others can be retained for long periods and accumulate in body tissues, particularly fatty tissues [55]. Some of the chemical elements that show the greatest bioaccumulation are those that do not dissolve in water, but instead dissolve in fats and oils (i.e., mercury and PCBs). In some cases, the accumulation of pollutants is intensified in carnivorous animals high in the food chain, ranging from big organism such as fishes and to human [56].

Figure 8. Some examples of organism commonly used for environmental biomonitoring study. Photo by Ong Meng Chuan.

7. Tolerable intake

Beside fishes, shellfish such as oysters and mussels are an important source of dietary protein in coastal communities. Depending on consumer, those shellfish can be “swallowed” or masticated normally, increasing the surface contact between food and digestive fluids. The consumer will consume whole soft part of the shellfish (**Figure 9**); therefore, in the pollution study which relates to human health, the metals content is examined in toto or shellfish flesh.

To safeguard public health, who consumes these organisms, maximum acceptable concentrations of toxic contaminants have been established in various countries. As a result, there is a specific legislation for shellfish, which establishes the maximum allowed concentration for metals (**Table 2**).

Figure 9. Oyster in toto tissue use for metals study in relation to human health. Photo by Ong Meng Chuan.

	Cu	Zn	Cd	Pb	As	Hg	References
Shellfish							
European community	n.m.	n.m.	1	1.5	n.m.	0.5–1.0	[57]
Spain	20	n.m.	1	5	n.m.	0.5	[58]
Australia	30	150	2	2	1	0.5	[59]
China	n.m.	n.m.	0.1	0.5	1.0	0.3	[60]
Hong Kong	n.m.	n.m.	2	6	1.4	0.5	[61]
Singapore	n.m.	n.m.	1	2	1	0.5	[62]
Food category not specific							
Malaysia	30	50	1	2	n.m.	0.5	[63]
Thailand	20	133	n.m.	1.0	2	0.5	[64]
Brazil	30	50	1	2	n.m.	0.5	[65]

n.m.: not mentioned.

Table 2. Maximum permissible levels (expressed in mg/kg wet weight) of metals in shellfish from different countries or regions.

International scientific committees such as the Joint FAO/WHO Expert Committee on Food Additives (JECFA), regional scientific committees such as the European Union and national regulatory agencies generally use the safety factor approach for establishing acceptable of tolerable intakes of substances that exhibit thresholds of the toxicity of contaminants. JECFA derives tolerable intakes, expressed on either daily or weekly basis, for contaminants [66]. Lead, Cd, As, and Hg are not removed rapidly from human body and for this category of pollutants, provisional tolerable weekly intakes (PTWIs) are calculated and expressed on a weekly basis because the pollutant may accumulate within the human body over a period of time [67]. The term tolerable is used because it signifies permissibility rather than acceptability for the pollutants intake unavoidably associated with the consumption.

8. Conclusion

Wetlands are well known to researcher as an ecosystem that are highly sensitive to pollution effects and can change the ecosystem's biogeochemistry process. Sediment and organisms from wetlands ecosystem are important to describe the environmental quality that act as geo-marker and biomarker, respectively. The assessment of metals pollution in the ecosystem has been carried out in different parts of the world and represents the impact of human activities toward the ecosystem. Although some of the metals are present in low concentration, their impacts on wetland ecosystems are significant because of their toxicity especially toward organisms and human. Due to the importance of wetlands to us, it is important to evaluate and monitor the ecosystem health and understand their contamination status to maintain the stability of the environment.

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Conflict of interest

The authors whose names are listed immediately below certify that they have no affiliations with or involvement in any organization or entity with any financial interest (such as honoraria; educational grants; participation in speakers' bureaus; membership, employment, consultancies, stock ownership, or other equity interest; and expert testimony or patent-licensing arrangements), or nonfinancial interest (such as personal or professional relationships, affiliations, knowledge or beliefs) in the subject matter or materials discussed in this manuscript.

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Chapter 4

Macroalgae Species as Zonal Indicators of Coral Reef: A Case Study from Bet Shankhodhar Reef, India

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Abstract

Tropical coral reefs are major habitats for marine macroalgae or seaweeds. Macroalgae represent a key functional group among the coral reef communities and perform vital ecological functions like reef structure stabilisation, production of tropical sands, nutrient retention and recycling, primary productivity and trophic support. Coral reef macroalgae are comprised of three major pigment-group-based phyla: Chlorophyta (green algae), Heterokontophyta or Ochrophyta (brown algae) and Rhodophyta (red algae). Green macroalgae or Chlorophyta contain chlorophyll *a* and *b* pigments in the same proportion as that of higher plants along with β-carotene and xanthophylls and have significant industrial or commercial value. Chlorophyta members commonly inhabit the littoral zone with strong sunlight. This chapter highlights micro-level habitat preference of green macroalgae or Chlorophyta species sampled from Bet Shankhodhar Reef from the Gujarat coast of India as a unique case study. This study identifies four Chlorophyta species: *Halimeda tuna* (Ellis & Solander) Lamouroux, *Caulerpa sertularioides* (S. Gmelin) Howe *f. brevipes* (J. Agardh) Svedelius, *Valonia aegagropila* C. Agardh and *Valoniopsis pachynema* (Martens) Børgesen, as indicator species of the backreef zone. Shallow tidal pools in the backreef zone of Bet Shankhodhar Reef are preferred microhabitats for *C. sertularioides* and *V. aegagropila*.

Keywords: coral reef, macroalgae, Chlorophyta, microhabitat, indicator species

1. Introduction

Tropical coastal and near-shore marine wetlands include a wide range of habitats with different structural complexities (e.g., rock coasts, soft-sediment coasts, estuaries, mangroves, salt marshes, coral reefs and seagrass beds). The intricate, three-dimensional, underwater landscape of coral reefs provides necessary habitat to one-third of known marine species [1] including some of the rare avifauna [2]. Coral reefs are critical eco-resources [3] to many of the maritime tropical and subtropical nations. For many of these nations, their physical foundation to national economies depend on reef-related ecosystem goods and services [4, 5].

Marine macroalgae or seaweeds occupy variety of habitats offered by the coastal and near-shore marine wetlands. Macroalgae represent a key functional group among the coral reef communities and perform vital ecological functions like reef structure stabilisation, production of tropical sands, nutrient retention and recycling, primary productivity and trophic support [6]. Coral reef macroalgae are comprised of three major pigment group-based phyla: Chlorophyta (green algae), Heterokontophyta (brown algae) and Rhodophyta (red algae). This systematic classification is based on the composition of pigments involved in photosynthesis [7]. The presence of chloroplasts and subsequent capacity to photosynthesize allow reef macroalgae to play the vital ecological role of primary producers in a reef ecosystem. Other than their ecological roles as habitat formers and primary producers, reef macroalgae are economically important. They are important sources of food, fodder, fertiliser, medicinal compounds and industrial raw materials.

Marine green algae or Chlorophyta are naturally abundant and record high biodiversity in tropical coral reefs and lagoons, often intermixed with associated seagrass habitats [6, 8]. Chlorophyta have predominantly green chlorophyll pigments: chlorophyll *a* and *b* in the same proportion as that of higher or vascular plants along with accessory carotenoid and xanthophyll pigments. Structurally, green seaweeds range from thread-like filaments to thin sheets and can be spongy, gelatinous, papery, leathery or brittle in texture [8]. Their morphological appearance is shaped through their cell division process [9]. Chlorophyta are generally siphonaceous or giant-celled forms which employ a unique cytoplasmic streaming or blade abandonment mechanism to eliminate epiphytes [8]. Certain genera of filamentous and sheet-like green algae are stress tolerant and can be potential indicators of freshwater seeps, disturbed areas of the habitat, areas of low herbivory and significant areas with an overabundance of nutrients [8].

In general, Chlorophyta are usually found in the littoral zone with strong sunlight. Availability of suitable substrate, light quality and quantity, availability of nutrients, intra- and interspecific competition, herbivory and grazing are major factors that delimit spatial and temporal occupancy of macroalgae in a given habitat [6]. Algal pigments and their photosynthetic capability and adaptations to different light levels lead to their depth zonation within the habitat. This chapter explores the concept of species-specific microhabitat preference of green macroalgae in a coral reef habitat of India based on field survey data.

2. Study area: Bet Shankhodhar Reef

Coral reefs provide a hard and stable habitat for algal settlers as compared to any other soft sediment coastal habitats like beaches, spits, estuaries and mudflats. India has a coastline length of 7500 km with diverse coastal habitats which support rich seaweed biodiversity [10].

The state of Gujarat shares 1600 km of the Indian coastline and represents the northwestern most part of the peninsular India. Gujarat coast is known to harbour a rich diversity of seaweeds as its rocky segments provide suitable environment for macroalgal settlement and growth [11]. Gujarat coastline falls within the geographical limits of 20°08′–24°40′ north latitudes and 68°10′–74°28′ east longitudes. From north to south, the Gujarat coast can be divided into four major coastal ecological components: (i) Kori Creek, (ii) Gulf of Kachchh, (iii) Saurashtra coast from Okha to Porbandar and (iv) Gulf of Khambhat [12], situated in three distinct macro-geomorphological settings of a deltaic creek, two gulfs and a rocky coast.

Gulf of Kachchh (**Figure 1A** and **B**) marks the northernmost limit of reef development on the continental shelf of India [13]. Gulf of Kachchh is a funnel-shaped, east-west-oriented, seismically active indentation between the Kachchh mainland and Saurashtra/Kathiawar peninsula of Gujarat state in India [14]. This gulf occupies an area of 7350 km^2 with an average depth of 30 m [15]. Gulf of Kachchh represents a high-energy, semi-diurnal, macro-tidal environment with varying tidal amplitude of 4 m at its mouth to 7 m in the inner gulf [14]. The southern shore of the gulf is relatively smooth and has an assemblage of ecologically sensitive ecosystems including coral reefs, seagrass beds, seaweeds, mangroves and tidal flats [15].

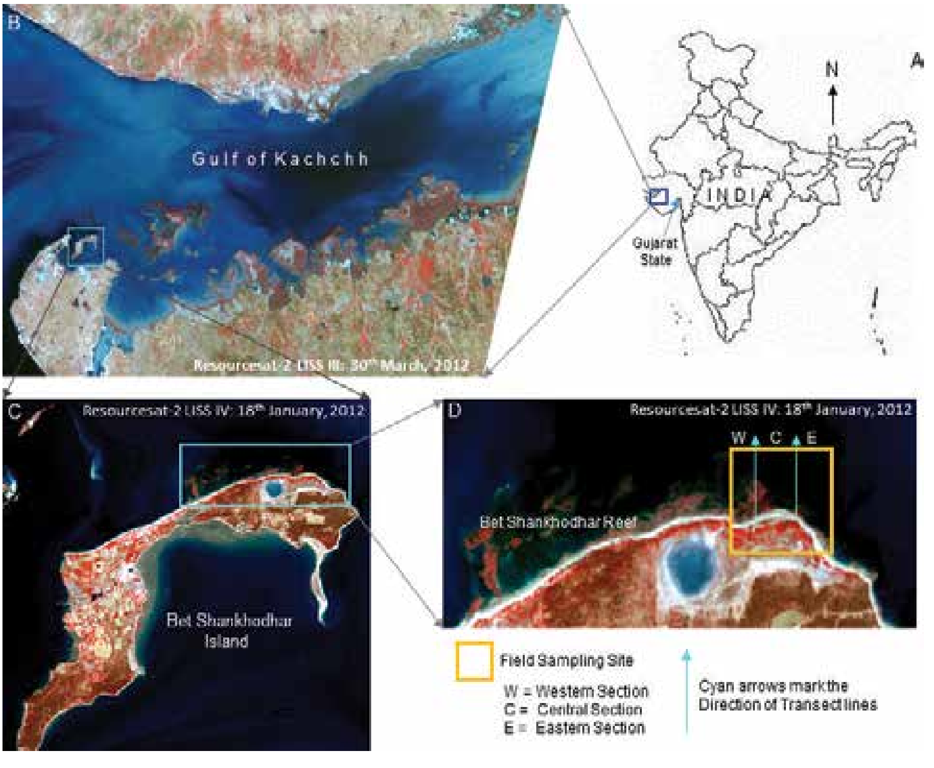

Figure 1. Location of the study site at Bet Shankhodhar Reef, India. (A) Location of Gulf of Kachchh in India, (B) location of Bet Shankhodhar Island in Gulf of Kachchh, (C) location of coral reef area in Bet Shankhodhar Island, (D) study/field sampling site at Bet Shankhodhar Reef.

The coral reefs in Gulf of Kachchh are predominantly patchy structures built up on wave-cut sandstone banks [16] on the southern shore of the gulf along with 34 adjoining islands [17]. These coral reefs are mainly comprised of fringing structure with all sub-types (i.e., platform, patch and coral pinnacles; [13]) restricted to a vast intertidal region [18]. Gulf of Kachchh coral reefs are adapted to extreme environmental conditions: high temperature ranges (10–35°C), high salinity ranges (25–40 ppt), large tidal ranges, strong tidal currents and heavy sediment loads [19]. As a result of isolation and above-mentioned extreme environmental conditions, the species diversity of corals in this region is low [20]. The coral reefs of Gulf of Kachchh are under International Union for Conservation of Nature (IUCN) Category I Marine Protected Area (MPA). Gulf of Kachchh Marine Sanctuary and Marine National Park were established in 1980 and 1982, respectively [21].

The present study was carried out in the coral reef area adjacent to Bet Shankhodhar Island (**Figure 1B** and **C**) situated to the east of Okhamandal area on the mainland coast and 2 km away from the Okha Port. The island owes its name Bet Shankhodhar to its unique shape resembling that of a conch shell [22]. Bet Shankhodhar Island has a fringing reef area (**Figure 1C**) of 28 hectares to its north [22] adjacent to a narrow strip of beach with significant exposures of beach rock [23]. This reef was selected for the present case study for its reported diversity of 120 species of macroalgae [22]. The study site on Bet Shankhodhar Reef is located within the coordinates of 22°28′36″ N–22°28′52″ N latitudes and 68°08′14″ E–69°08′40″ E longitudes and covered a survey area of 0.35 km^2 (**Figure 1D**). The survey area of the reef was further divided into three micro-zones in the north-south direction based on their topographical and geomorphological characteristics and level of tidal inundation. These three

Figure 2. Microhabitat zones of Bet Shankhodhar Reef. (A) Exposure of subtidal zone, (B) Backreef zone I and (C) Backreef zone II.

zones (**Figure 2**) are: (i) subtidal zone or the fore reef, (ii) Backreef zone I and (iii) Backreef zone II. The subtidal zone or the fore reef was the northernmost zone which got exposed only during the spring tides while the backreef zone I or the intertidal reef flat was interspersed with large rock pools. The backreef zone II was the southernmost zone, adjacent to the beach and was characterised with coastal lapiés and smaller rock pools [23].

3. Field data collection and analysis

3.1. Field sampling of seaweeds/macroalgae

The study area of Bet Shankhodhar Reef was divided into three sections in the west-east direction, as: (i) western, (ii) central and (iii) eastern sections (**Figure 1D**) for systematic field sampling and equal representation of the reef habitat. Seaweed sampling was routinely carried out for 2 years: from April 2013 to April 2015 with sampling exercise coinciding with the annual cycles of seaweed abundance and growth, that is, local seasons of post-monsoon (October-November), winter (December-February), spring (March) and summer (April-June), respectively. Field surveys/samplings were carried out during low-tide exposures of the reef following line intercept transects (LITs). For quantitative assessment of the seaweeds in the given area, the GPS-tagged, LITs were laid perpendicular to the coast in a seaward direction with the help of a 50-m-long rope [24]. The length of the transect essentially depended on the tidal exposure of the reef during the field surveys. The minimum and maximum transect lengths surveyed were 52 and 372.5 m, respectively. The maximum depth of the subtidal zone sampled for the present study is 1 m. Quadrats of 1 m^2 were positioned over the transects for quantitative seaweed sampling wherever macroalgal growth, density and diversity were visibly high. A total of 182 GPS-tagged quadrats was sampled for the seaweeds over a total of 23 transects on the reef site.

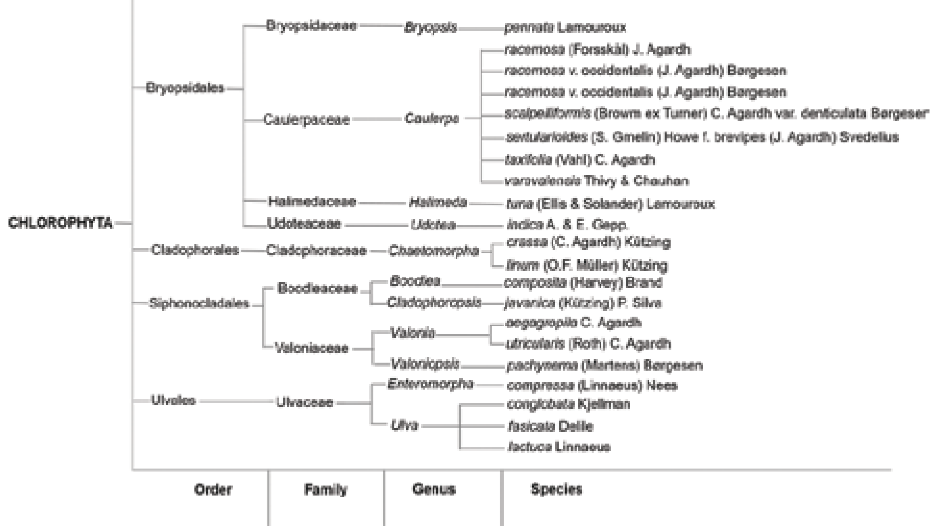

Figure 3. Cladogram of Chlorophyta species sampled from Bet Shankhodhar Reef.

3.2. Field data analysis

Macroalgae samples collected from the field were taken to laboratory for herbarium preparation and sample identification. Morphological criteria and reproductive structures of the algae specimens were analysed for taxa identification. A cladogram (**Figure 3**) was prepared for the sampled Chlorophyta in order to generate classification statistics, that is, number of genera and species pertaining to different families and genera.

4. Results and discussion

Twenty-one species of Chlorophyta (**Table 1**) were identified based on the field survey data collected for 2 years: April 2013 to April 2015. These species belonged to 4 orders, 7 families and 11 genera as shown in the cladogram (**Figure 3**). As the Chlorophyta species distribution is tagged with the zonal morphology of the Bet Shankhodhar Reef as a part of this study, it is found that the sampled species can be classified into five major groups as per their zonal or microhabitat occurrences. These five groups are: (i) subtidal and Backreef zone I species, (ii) Backreef zone I species, (iii) Backreef zone I and zone II species, (iv) ubiquitous species and (v) chance factor species. The taxonomic details of the species classified under these five major groups are mentioned below.

i. Subtidal and Backreef zone I species: the subtidal and backreef zone I species include four Chlorophyta species from two families: Caulerpaceae and Udoteaceae. *Caulerpa racemosa* v. *occidentalis* (J. Agardh) Børgesen and *Caulerpa scalpelliformis* (Brown ex Turner) C. Agardh var. *denticulata* Børgesen and *Caulerpa taxifolia* (Vahl) C. Agardh belong to the Caulerpaceae family while *Udotea indica* A. & E. Gepp represents the Udoteaceae family.

ii. Backreef zone I species: Backreef zone I species include four members from three families: Caulerpaceae, Halimedaceae and Valoniaceae. *Caulerpa sertularioides* (S. Gmelin) Howe *f. brevipes* (J. Agardh) Svedelius represents the Caulerpaceae family while *Halimeda tuna* (Ellis & Solander) Lamouroux belongs to the Halimedaceae family. *Valonia aegagropila* C. Agardh and *Valoniopsis pachynema* (Martens) Børgesen are members of the Valoniaceae family.

iii. Backreef zone I and zone II species: the backreef zone I and zone II species again include four species representing three families: Boodleaceae, Ulvaceae and Valoniaceae. *Boodlea composita* (Harvey) Brand and *Cladophoropsis javanica* (Kützing) P. Silva represent the first family while *Ulva conglobata* Kjellman belongs to the Ulvaceae family. *Valonia utricularis* (Roth) C. Agardh is the other species identified in the backreef zone I and zone II. These four species were found in both the backreef zones from all three sections of Bet Shankhodhar Reef and were absent in the subtidal zone.

iv. Ubiquitous species: Chlorophyta species found in all the three reef zones were considered as ubiquitous species. For Bet Shankhodhar Reef, ubiquitous species included five Chlorophyta species representing three families. *Caulerpa veravalensis* Thivy & Chauhan belonged to Caulerpaceae family while *Chaetomorpha crassa* (C. Agardh) Kützing belonged

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to Cladophoraceae family. The other three species: *Enteromorpha compressa* (Linnaeus) Nees, *Ulva fasciata* Delile and *Ulva lactuca* Linnaeus are members of the Ulvaceae family.

v. Chance factor species: the Chlorophyta species encountered only once during the two-year field sampling are considered as chance factor species. In the case of Bet Shankhodhar

Sr. No.	Chlorophyta species	Field site: Bet Shankhodhar Reef								
		Months								
		O	N	D	J	F	M	A	M	J
1	*Boodlea composita* (Harvey) Brand	×	×	√	√	√	×	×	×	×
2	*Bryopsis pennata* Lamouroux	×	×	×	√	×	×	×	×	×
3	*Caulerpa racemosa* (Forsskål) J. Agardh	×	×	×	×	×	√	×	×	×
4	*Caulerpa racemosa* (Forsskål) J. Agardh *V. macrophysa* (Sonder ex Kützing) Taylor	×	×	×	√	×	×	×	×	×
5	*Caulerpa racemosa v. occidentalis* (J. Agardh) Børgesen	×	×	√	√	√	×	√	×	×
6	*Caulerpa scalpelliformis* (Brown ex Turner) C. Agardh var. *denticulata* Børgesen	×	×	×	×	×	√	√	×	×
7	*Caulerpa sertularioides* (S. Gmelin) Howe *f. brevipes* (J. Agardh) Svedelius	×	×	√	√	√	×	×	×	×
8	*Caulerpa taxifolia* (Vahl) C. Agardh	×	×	√	√	√	√	√	×	×
9	*Caulerpa veravalensis* Thivy & Chauhan	√	×	√	√	×	×	×	×	×
10	*Chaetomorpha crassa* (C. Agardh) Kützing	×	×	×	√	√	√	√	×	×
11	*Chaetomorpha linum*(O. F. Müller) Kützing	×	×	×	×	×	√	×	×	×
12	*Cladophoropsis javanica* (Kützing) P. Silva	×	×	×	×	√	√	√	×	×
13	*Enteromorpha compressa* (Linnaeus) Nees	×	×	√	√	√	√	×	×	×
14	*Halimeda tuna* (Ellis & Solander) Lamouroux	√	×	×	×	×	×	√	×	×
15	*Udotea indica* A. & E. Gepp.	×	×	×	×	√	×	√	×	√
16	*Ulva conglobata* Kjellman	×	×	√	√	×	×	×	×	×
17	*Ulva fasciata* Delile	×	×	×	×	√	√	√	×	×
18	*Ulva lactuca* Linnaeus	×	×	√	√	√	√	√	×	×
19	*Valonia aegagropila* C. Agardh	×	×	√	×	√	×	×	×	×
20	*Valonia utricularis* (Roth) C. Agardh	×	×	√	√	×	×	√	×	×
21	*Valoniopsis pachynema* (Martens) Børgesen	×	×	√	√	×	×	×	×	×

The calendar months are denoted with the first letter, for example, O = October, starting with October and continuing up to June indicating local post-monsoon, winter, spring and summer seasons; √ denotes presence and x denotes absence of the species.

Table 1. Chlorophyta species observed during different months at Bet Shankhodhar Reef.

Reef, four species were found as chance factor species. These species include *Bryopsis pennata* Lamouroux from the Bryopsidaceae family. This species was found in the fore reef zone of the western section of the reef. Two species of *Caulerpaceae*: *Caulerpa racemosa* (Forsskål) J. Agardh and *Caulerpa racemosa* (Forsskål) J. Agardh v. *macrophysa* (Sonder ex Kützing) Taylor were encountered in the subtidal zone of the western section and the backreef zone I of the central section, respectively. The remaining species, *Chaetomorpha linum* (O. F. Müller) Kützing, member of Cladophoraceae family, was also observed for once in the subtidal zone of the western section of the reef.

From the preceding classification of the 21 Chlorophtya species, as per their spatial occurrences on the reef, the following 4 species: *Caulerpa sertularioides*, *Halimeda tuna*, *Valonia aegagropila* and *Valoniopsis pachynema,* are found exclusive to the back reef zone 1 of Bet Shankhodhar Reef.

The establishment of seaweeds or marine macroalgae within a habitat zone involves complex physical interactions as well as biological, ecological and chemical processes at the microscale [25]. These processes include release of propagules by reproductive adults, migration of propagules to suitable substrates, initial adhesion to the substratum surface, permanent attachment and further growth and development [25]. Availability of suitable substrate, light quantity and quality and nutrients are three major abiotic factors that control the settlement and growth of macroalgal communities within an ecosystem [6].

Halimeda is a globally significant, calcifying, green macroalgae genera strongly associated with tropical coral reef habitats [26]. *Halimeda* species are widely distributed across the reefs indicating different reef conditions and are considered important primary producers of the backreef and lagoon habitats [6]. These species prefer moderate energy environments like shallow backreef and lagoon habitats, while in the fore-reef zone, *Halimeda* species may occur in large populations even at greater depths beyond 100 m [26]. Approximately 75% of *Halimeda* species prefer consolidated or gravelly habitats such as against sand or mud substratum [6] and the same was experienced with *H. tuna* at Bet Shankhodhar Reef (**Figure 4**: Plate A: 1A and 1B).

Caulerpa is another major Chlorophyta genus, commonly found in the tropical and subtropical coastal waters throughout the world [27]. *Caulerpa* species generally occur in the intertidal and subtidal zones and prefer sandy and rocky reef substrates. They are also important primary producers of the backreef zone and lagoon habitats similar to that of *Halimeda* genera [6]. *C. sertularioides* is rather considered as a secondary metabolite which yields different potentially bioactive compounds, both toxic and non-toxic [27]. This chemically defended species inhabits tropical reefs with high fish populations [27] and prefers unconsolidated sand or soft mud substratum in the shallow tidal pools [28]. In Bet Shankhodhar Reef, this species is found in abundance in the tidal pools (**Figure 4**: Plate A: 2A and 2B) in the backreef zone I along with other *Caulerpa* species.

Valonia aegagropila species from the Valoniaceae family was also found in the backreef zone I of Bet Shankhodhar Reef (**Figure 5**: Plate B: 1A and 1B) during the field surveys. This species is identified as a lower mid-littoral zone species, inhabiting shallow tidal pools and prefers

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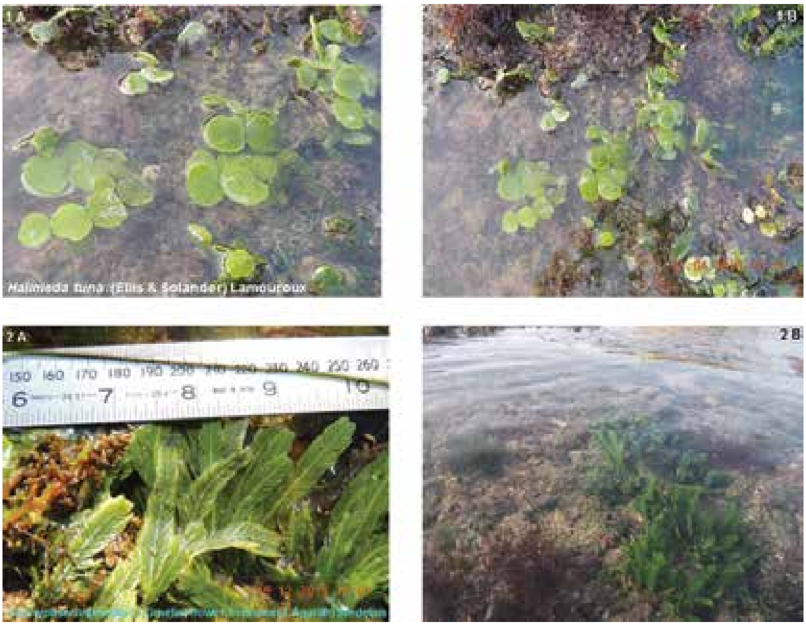

Figure 4. Indicator Chlorophyceae species of Backreef zone I of Bet Shankhodhar Reef: Plate A (1A: *Halimeda tuna* species and 1B: *H. tuna* in its habitat; 2A: *Caulerpa sertularioides* species and 2B: *C. sertularioides* in shallow tidal pool).

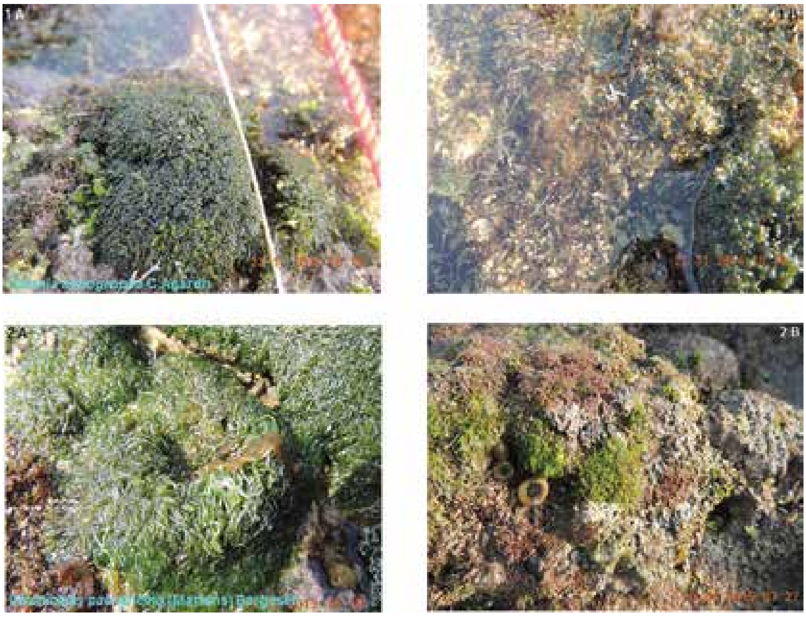

Figure 5. Indicator Chlorophyceae species of Backreef zone I of Bet Shankhodhar Reef: Plate B (1A: *Valonia aegagropila* species and 1B: *V. aegagropila* in its habitat; 2A: *Valoniopsis pachynema* species and 2B: *V. pachynema* on rocky substratum).

intertidal rocks and coralline stones as suitable substratum [10]. Other species from the same genus have been reported from reef front zones of shelf-edge atolls of northwestern Australia [29]. Occurrence of this species is reported from the rocky substratum at the intertidal sampling sites of Uran coast of Navi Mumbai, Maharashtra, India [30].

Valoniopsis pachynema is another Valoniaceae family member which forms stiff cushions or spongy mats on intertidal rocks of coralline origin [10]. This is a common tropical sea species and prefers hard substratum like intertidal rocks and dead corals. It forms green, hairy clumps and appears as turfs in littoral zones dominated by high wave actions. In our study site, this species was found in the backreef zone I, on the dead reef substrate in association with invasive zootaxa: *Zoanthus* (**Figure 5**: Plate B: 2A and 2B).

5. Conclusions

This study has identified four Chlorophyta species: *Caulerpa sertularioides*, *Halimeda tuna*, *Valonia aegagropila* and *Valoniopsis pachynema,* exclusive to the backreef zone I of Bet Shankhodhar Reef. *H. tuna* generally occurs in the sub-littoral zone and the infra-littoral fringe while rest of the Chlorophyta species occurs in the mid-littoral zone. Two of the species, *C. sertularioides* and *V. aegagropila,* prefer shallow tidal pools as their microhabitat within the backreef zone. All the four species grow on intertidal rocks having a calcareous or coralline origin. However, *C. sertularioides* prefers a thin veneer of fine sediments on the intertidal rocks as a suitable substrate to settle and grow. Thus, *C. sertularioides* prefers soft sediment substratum as compared to other three species. Since *Halimeda* and *Caulerpa* genera are well-known primary producers in backreef habitats, presence of these four species indicates backreef zone or environment for the Bet Shankhodhar Reef. Presence of chlorophyll *a* and chlorophyll *b* as the main accessory pigment in these species restricts the distribution of these sub- and mid-littoral species to relatively shallow depths of the reef with strong sunlight as compared to other reef algae belonging to other pigment groups.

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Section 3

Geographic Information System (GIS) and Remote Sensing Applications

Chapter 5

Wetland Monitoring and Mapping Using Synthetic Aperture Radar

Mohammed Dabboor and Brian Brisco

Additional information is available at the end of the chapter

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Abstract

Wetlands are critical for ensuring healthy aquatic systems, preventing soil erosion, and securing groundwater reservoirs. Also, they provide habitat for many animal and plant species. Thus, the continuous monitoring and mapping of wetlands is necessary for observing effects of climate change and ensuring a healthy environment. Synthetic Aperture Radar (SAR) remote sensing satellites are active remote sensing instruments essential for monitoring wetlands, given the possibility to bypass the cloud-sensitive optical instruments and obtain satellite imagery day and night. Therefore, the purpose of this chapter is to provide an overview of the basic concepts of SAR remote sensing technology and its applications for wetland monitoring and mapping. Emphasis is given to SAR systems with full and compact polarimetric SAR capabilities. Brief discussions on the latest state-of-the-art wetland applications using SAR imagery are presented. Also, we summarize the current trends in wetland monitoring and mapping using SAR imagery. This chapter provides a good introduction to interested readers with limited background in SAR technology and its possible wetland applications.

Keywords: wetlands, SAR, satellites, monitoring, mapping

1. Introduction

Wetlands are defined based on the Canadian Wetland Classification System as land that is saturated with water long enough to promote wetland or aquatic processes as indicated by poorly drained soils, hydrophytic vegetation and various kinds of biological activity which are adapted to a wet environment [1]. Wetlands are important ecological systems which play a critical role in hydrology and act as water reservoirs, affecting water quality and controlling runoff rate [2]. Also, they are amongst the most productive ecosystems, providing food, construction materials, transport, and coastline protection. They provide many important environmental functions and habitat for a diversity of plant and animal species [2]. Furthermore, wetlands bring economic value with social benefits for people, providing significant tourism opportunities and recreation that can be a key source of income. For these reasons, the continuous and accurate monitoring of wetlands is necessary, especially for better urban planning and improved natural resources management [3]. The formation of wetlands requires the presence of the appropriate hydrological, geomorphological and biological conditions [2].

The Canadian Wetland Classification System divides wetlands into five classes based on their developmental characteristics and the environment in which they exist [1]. As shown in **Figure 1**, these classes are: bogs, fens, marches, swamps, and shallow water. Bogs (**Figure 2a**) are peatlands with a peat layer of at least 40 cm thickness, consisting partially decomposed plants. Bogs surface is usually higher relatively to the surrounding landscape and characterized by evergreen trees and shrubs and covered by sphagnum moss. The only source of water and nutrients in this type of wetlands is the rainfall [4]. Bogs are extremely low in mineral nutrients and tend to be strongly acidic [1].

Like bogs, fens (**Figure 2b**) are also peatlands that accumulate peats. Fens occurs in regions where the ground water discharges to the surface [1]. This type of wetlands is usually covered by grasses, sedges, reeds, and wildflowers. Typically, fens have more nutrients than bogs, and the water is less acidic [4]. Marshes (**Figure 2c**) are wetlands that are periodically or permanently flooded with standing or slowly moving water and hence are rich in nutrients [4]. Some marshes accumulate peats, though many do not. Marshes are characterized by non-woody vegetation, such as cattails, rushes, reeds, grasses and sedges [1]. Similar to marshes, swamps (**Figure 2d**) are wetlands that are subject to relatively large seasonal water level fluctuations [4]. Swamps are characterized by woody vegetation, such as dense coniferous or deciduous forest and tall shrubs. Some marshes accumulate peats, though many do not [1]. Shallow open water wetlands (**Figure 2e**) are ponds of standing water bodies, which represent a transition

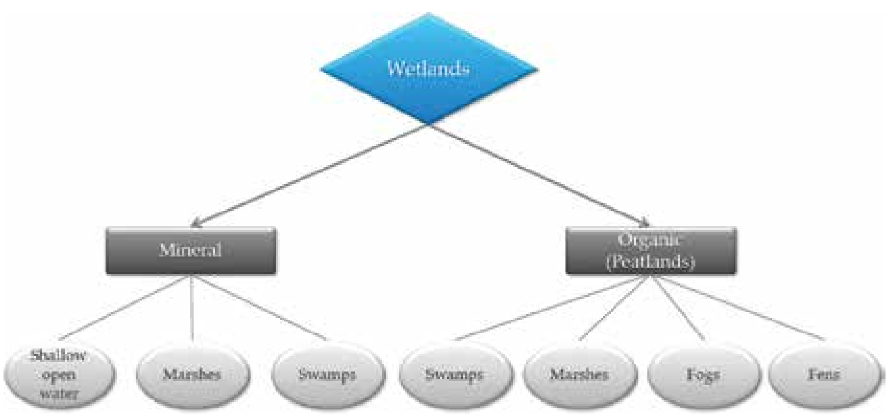

Figure 1. Wetland classes hierarchy.

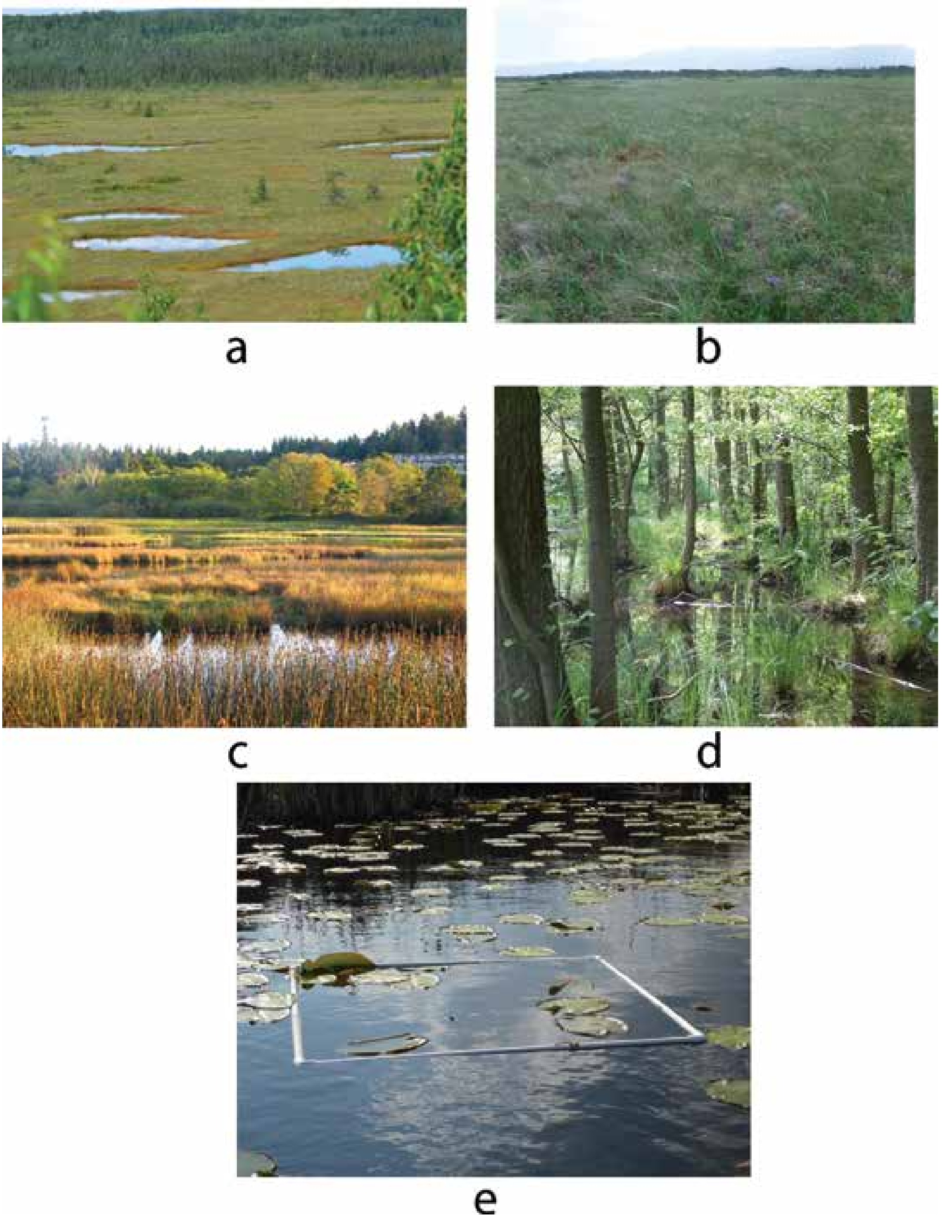

Figure 2. Wetland classes as defined by the Canadian Wetland Classification System: (a) bog, (b) fen, (c) marsh, (d) swamp and (e) shallow open water.

stage between lakes and marshes. This type of wetlands is free of vegetation with a depth of less than 2 m [1].

Spaceborne remote sensing technology is necessary for effective monitoring and mapping of wetlands. The use of this technology provides a practical monitoring and mapping approach of wetlands, especially for those located in remote areas [5].

2. Basic SAR concepts

Wetlands are usually located in remote areas with limited accessibility. Thus, remote sensing technology is attractive for mapping and monitoring wetlands. Synthetic Aperture Radar (SAR) systems are active remote sensing systems independent of weather and sun illumination. SAR systems transmit electromagnetic microwave from their radar antenna and record the backscattered signal from the radar target [6]. The sensitivity of SAR sensors is a function of the: (1) band, polarization, and incidence angle of the transmitted electromagnetic signal and (2) geometric and dielectric properties of the radar target [7]. Radar targets can be discriminated in a SAR image if their backscattering components are different and the radar spatial resolution is sufficient to distinguish between targets [6]. Conventional SAR systems are linearly polarized radar systems which transmit horizontally and/or vertically polarized radar signal and receive the horizontal and/or vertical polarized components of the backscattered signal (**Figure 3**). In SAR systems, polarization is referred to the orientation of the electrical field of the electromagnetic wave.

A single polarized SAR system is a SAR system which transmits one horizontally or vertically polarized signal and receives the horizontal or vertical polarized component of the returned

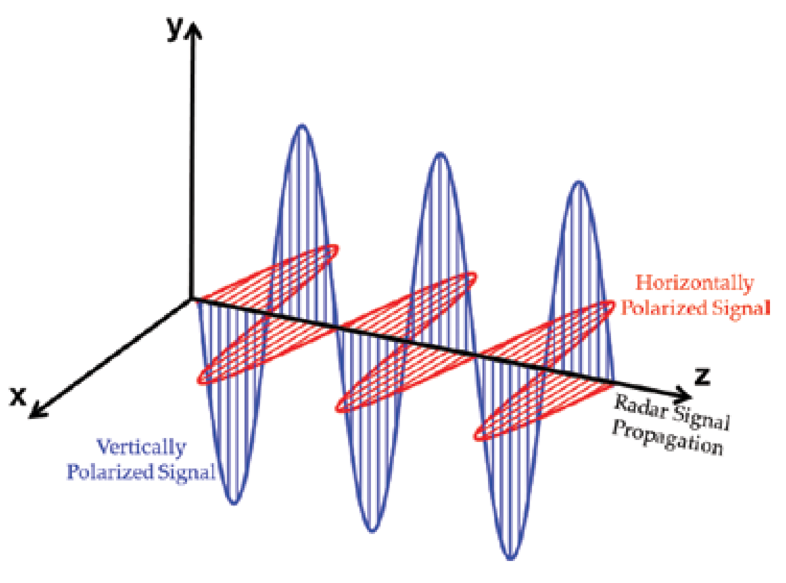

Figure 3. Horizontally and vertically polarized radar signal.

signal. A dual polarized SAR system is a SAR system which transmits one horizontally or vertically polarized signal and receives both the horizontal and vertical polarized components of the returned signal. A single or dual polarized SAR system acquires partial information with respect to the full polarimetric state of the radar target. A fully polarimetric SAR system transmits alternatively horizontally and vertically polarized signal and receives returns in both orthogonal polarizations, allowing for complete information of the radar target [6, 8]. While full polarimetric SAR systems provide complete information about the radar target, the coverage of these systems is half of the coverage of single or dual polarized SAR systems. Also, the energy required by the satellite for the acquisition of full polarimetric SAR imagery and the pulse repetition frequency of the SAR sensor are twice the single or dual polarized SAR systems.

A new SAR configuration named compact polarimetric SAR is currently being implemented in SAR systems, where a circular polarized signal (**Figure 4**) is transmitted and two orthogonal polarizations (horizontal and vertical) are coherently received [9]. Thus, the relative phase between the two receiving channels is preserved and calibrated, but the swath coverage is not reduced.

In comparison to the full polarimetric SAR systems, compact polarimetric SAR operates with half pulse repetition frequency, reducing the average transmit power and increasing the swath width. Consequently, this SAR configuration is associated with low-cost and low-mass constraints of the spaceborne polarimetric SAR systems. The wider coverage of the compact SAR system reduces the revisit time of the satellite, making this system operationally viable [10]. These advantages come with an associated cost in the loss of full polarimetric information. Hence, generally, a compact polarimetric SAR system cannot be "as good as" a full polarimetric system [11]. Such SAR architecture is already included in the current Indian Radar Imaging Satellite-1 (RISAT-1) and the Japanese Advanced Land Observing Satellite-2 (ALOS-2) carrying the Phased Array type L-band Synthetic Aperture Radar-2 (PALSAR-2). Also, compact polarimetric SAR will be included in the future Canadian RADARSAT Constellation Mission (RCM).

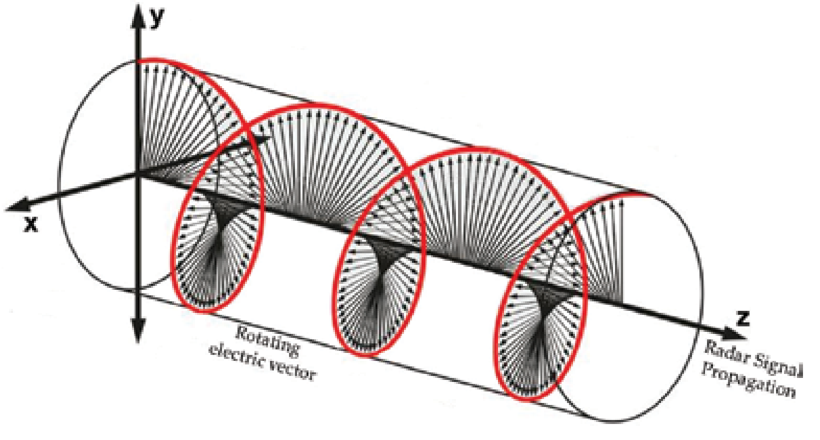

Figure 4. Circular polarized radar signal.

2.1. Polarimetric scattering vector

Fully polarimetric SAR systems measure the complete polarimetric information of a radar target in the form of a scattering matrix [S]. The scattering matrix [S] is an array of four complex elements that describes the transformation of the polarization of a wave pulse incident upon a reflective medium to the polarization of the backscattered wave and has the form [6]:

$$[\mathrm{S}] = \begin{bmatrix} \mathrm{HH} & \mathrm{HV} \\ \mathrm{VH} & \mathrm{VV} \end{bmatrix} \tag{1}$$

where H and V refer to horizontal and vertical polarized signals, respectively. The elements of the scattering matrix [S] are complex scattering amplitudes. For most natural targets including wetlands, the reciprocity assumption holds where HV = VH. The diagonal elements HH and VV are called co-polarized elements, while the off-diagonal elements HV and VH are called cross-polarized elements. Two polarimetric scattering vectors can be extracted from the target scattering matrix, which are the lexicographical scattering vector and the Pauli scattering vector [12]. Assuming the reciprocity condition, the lexicological scattering vector has the form:

$$\mathrm{K_l} = [\mathrm{HH\ VV\ 2HV}]^T \tag{2}$$

where the superscript T denotes the vector transpose. The multiplication of the cross-polarization with 2 is to preserve the total backscattered power of the returned signal. The Pauli scattering vector can be obtained from the complex Pauli spin matrices [6] and, assuming the reciprocity condition, has the form:

$$\mathrm{K_p} = \frac{1}{\sqrt{2}}[\mathrm{HH} + \mathrm{VV\ HH} - \mathrm{VV\ 2HV}]^T \tag{3}$$

Deterministic scatterers can be described completely by a single scattering matrix or vector. However, for remote sensing SAR applications, the assumption of pure deterministic scatterers is not valid. Thus, scatterers are non-deterministic and cannot be described with a single polarimetric scattering matrix or vector. This is because the resolution cell is bigger than the wavelength of the incident wave. Non-deterministic scatterers are spatially distributed. Therefore, each resolution cell is assumed to contain many deterministic scatterers, where each of these scatterers can be described by a single scattering matrix [$\mathrm{S_i}$]. Therefore, the measured scattering matrix [S] for one resolution cell consists of the coherent superposition of the individual scattering matrices [$\mathrm{S_i}$] of all the deterministic scatterers located within the resolution cell [6, 12].

An ensemble average of the complex product between the lexicological scattering vector $\mathrm{K_l}$ and $\mathrm{K_l^{*T}}$ leads to the so-called polarimetric covariance matrix [C], which has the form [6]:

$$[\mathrm{C}] = \mathrm{K_l.K_l^{*T}} = \left\langle \begin{bmatrix} |\mathrm{HH}|^2 & \mathrm{HHVV}^* & \sqrt{2}\mathrm{HHHV}^* \\ \mathrm{VVHH}^* & |\mathrm{VV}|^2 & \sqrt{2}\mathrm{VVHV}^* \\ \sqrt{2}\mathrm{HVHH}^* & \sqrt{2}\mathrm{HVVV}^* & 2|\mathrm{HV}|^2 \end{bmatrix} \right\rangle \tag{4}$$

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where $\langle \ldots \rangle$ denotes a spatial ensemble averaging assuming homogeneity of the random scattering medium and * the complex conjugate. Analogously, the so-called polarimetric coherency matrix [T] is formed by the complex product of the Pauli scattering vector K_p with its complex conjugate transpose K_p^{*T} and takes the form [6]:

$$[T] = K_p.K_p^{*T} = \frac{1}{2}\left\langle \begin{bmatrix} |HH+VV|^2 & (HH+VV)(HH-VV)^* & 2(HH+VV)HV^* \\ (HH-VV)(HH+VV)^* & |HH-VV|^2 & 2(HH-VV)HV^* \\ 2HV(HH+VV)^* & 2HV(HH-VV)^* & 4|HV|^2 \end{bmatrix} \right\rangle \tag{5}$$

The relationship between the covariance matrix [C] and the coherency matrix [T] is linear. Both matrices are full rank, hermitian positive semidefinite and have the same real non-negative eigenvalues, but different eigenvectors. Moreover, both matrices contain the complete information about variance and correlation for all the complex elements of the scattering matrix [S] [12].

A compact polarimetric SAR system transmits a right- or left-circular polarized signal, providing a scattering vector of two elements:

$$K_c = [RH\,RV]^T \tag{6}$$

where R refers to a transmitted right-circular polarized signal. A four-element vector called Stokes vector [g] can be calculated from the measured compact polarimetric scattering vector, as follow [11]:

$$[g] = \begin{bmatrix} g_0 \\ g_1 \\ g_2 \\ g_3 \end{bmatrix} = \left\langle \begin{bmatrix} |RH|^2 + |RV|^2 \\ |RH|^2 - |RV|^2 \\ 2\mathrm{Re}(RHRV^*) \\ -2\mathrm{Im}(RHRV^*) \end{bmatrix} \right\rangle \tag{7}$$

where Re and Im are the real and imaginary parts of a complex number. The first Stokes element g_0 is associated with the total power of the backscattered signal while the fourth Stokes vector is associated with the power in the right-hand and left-hand circularly polarized component [13]. The elements of the Stokes vector can be used to derive an average coherency matrix, which takes the form [14]:

$$[T_c] = \frac{1}{2}\begin{bmatrix} g_0 + g_1 & g_2 + ig_3 \\ g_2 - ig_3 & g_0 - g_1 \end{bmatrix} \tag{8}$$

2.2. Polarimetric scattering mechanisms

Radar backscattering is a function of the radar target properties (dielectric properties, roughness, target geometry) and the radar system characteristics (polarization, band, incidence angle). Three major backscattering mechanisms can take place during the backscattering process. These are the surface, double bounce and volume scattering mechanism (**Figure 5**).

In the case of surface scattering mechanism (**Figure 5**), the incident radar signal features one or an odd number of bounces before returns back to the SAR antenna. In this case, a phase shift of 180^{o} occurs between the transmitted and the received signal [6]. However, a very smooth surface could cause the radar incident signal to be reflected away from the radar antenna, causing the radar target to appear dark in the SAR image. In this case, scattering is called specular scattering. An example of such surfaces is the open water in wetlands [12]. In the case of double bounce scattering mechanism (**Figure 5**), the incident radar signal hits two surfaces, horizontal and adjacent vertical forming a dihedral angle, and almost all of incident waves return back to the radar antenna. Thus, the scattering from radar targets with double bounce scattering is very high. The phase difference between the transmitted and the received signal is equal to zero. Double bounce scattering mechanism is frequently observed in open wetlands, such as bog and marsh, as the results of the interaction of the radar signal between the standing water and vegetation [15]. In the case of volume scattering mechanism (**Figure 5**), the radar signal features multiple random scattering within the natural medium. Usually, a large portion of the transmitted signal is returned back to the SAR sensor, causing rise to cross polarizations (HV and VH). Thus, illuminated radar targets with volume scattering appear bright in a SAR image. Volume scattering is commonly observed in flooded vegetation wetlands due to multiple scattering in the vegetation canopy.

In general, the penetration capabilities and the attenuation depth of radar signal in a medium, such as flooded vegetation, increases with the increasing of the wavelength [6, 12]. **Figure 6** presents the penetration of radar signals for different bands. As shown in **Figure 6**, X-band SAR has a short wavelength signal with limited penetration capability, while L-band SAR has long wavelength signal with higher penetration capability. C-band SAR is assumed as a good compromise between X- and L-band SAR systems. As shown in **Figure 6**, the scattering mechanism of a radar target could be affected by the penetration depth of the radar signal. Thus, dense flooded vegetation could present volume scattering mechanism in X- or C-band SAR (return from canopy), but double bounce scattering mechanism in L-band due to scattering process from trunk-water interaction (**Figure 6**) [12].

Different decomposition methods have been proposed to derive the target scattering mechanisms for both full polarimetric [6, 16–24] and compact polarimetric [11, 25] SAR data. One of the earliest and widely used decomposition methods is the Cloude-Pottier decomposition [17].

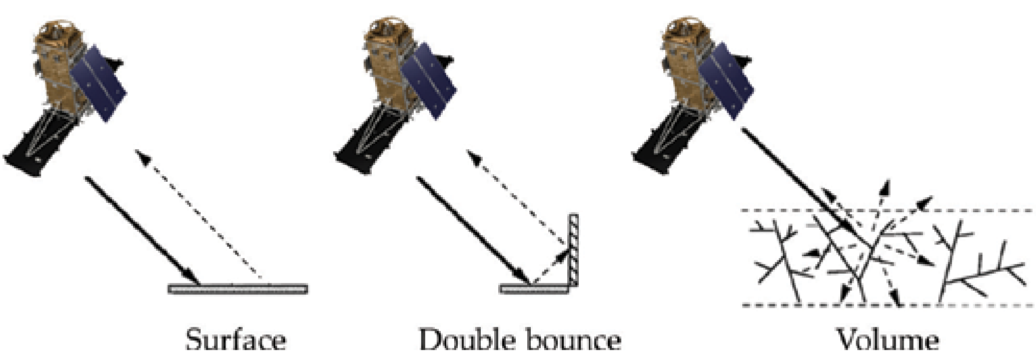

Figure 5. The three major scattering mechanisms: surface, double bounce and volume.

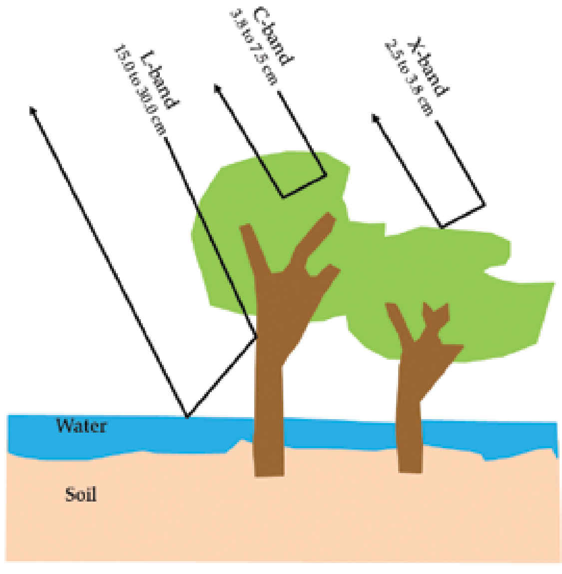

Figure 6. The radar signal penetration for different bands.

This method is incoherent decomposition method based on the eigenvector and eigenvalue analysis of the coherency matrix [T]. Given that [T] is hermitian positive semidefinite matrix, it can always be diagonalized using unitary similarity transformations. That is, the coherency matrix can be given as

$$[\mathrm{T}] = [\mathrm{U}][\Lambda][\mathrm{U}]^{*T} = [\mathrm{U}]\begin{bmatrix} \lambda_1 & 0 & 0 \\ 0 & \lambda_2 & 0 \\ 0 & 0 & \lambda_3 \end{bmatrix}[\mathrm{U}]^{*T} \tag{9}$$

where $[\Lambda]$ is the diagonal eigenvalue matrix of [T], $\lambda_1 \geq \lambda_2 \geq \lambda_3 \geq 0$ are the real eigenvalues and [U] is a unitary matrix whose columns correspond to the orthogonal eigenvectors of [T]. Based on the Cloude-Pottier decomposition, three parameters can be derived [17]. The polarimetric entropy H ($0 \leq \mathrm{H} \leq 1$) is defined by the logarithmic sum of the eigenvalues

$$\mathrm{H} = -\sum_{\mathrm{i}=1}^{3} \mathrm{P_i} \log_3 \mathrm{P_i} \tag{10}$$

where $\mathrm{P_i} = \lambda_\mathrm{i} / \sum_{\mathrm{i}=1}^{3} \lambda_\mathrm{i}$. This parameter is an indicator of the number of effective scattering mechanisms which took place in the scattering process [6]. The anisotropy A ($0 \leq \mathrm{A} \leq 1$) describes the proportions between the secondary scattering mechanisms

$$\mathrm{A} = \frac{\lambda_2 - \lambda_3}{\lambda_2 + \lambda_3} \tag{11}$$

The anisotropy A provides additional information only for medium values of H because in this case secondary scattering mechanisms, in addition to the dominant scattering mechanism, play an important role in the scattering process [6]. The alpha angle α ($0 \le \alpha \le 90^{o}$) provides information about the type of scattering mechanism

$$\alpha = \sum_{i=1}^{3} P_i \alpha_i \tag{12}$$

where $\cos(\alpha_i)$ in the magnitude of the first component of the coherency matrix eigenvector e_i ($i = 1, 2, 3$).

Another widely used polarimetric decomposition method is the Freeman-Durden method [18]. Contrary to the Cloude-Pottier decomposition, which is a purely mathematical construct, the Freeman-Durden decomposition method is a physically model-based incoherent decomposition based on the polarimetric covariance matrix. It relies on the conversion of a covariance matrix to a three-component model. The results of this decomposition are three coefficients corresponding to the weights of different model components. A polarimetric covariance matrix [C] can be decomposed to a sum of three components, corresponding to volume, surface, and double bounce scattering mechanisms [18]:

$$[C] = f_v[C]_v + f_s[C]_s + f_d[C]_d \tag{13}$$

where f_v, f_s, and f_d are the three coefficients corresponding to volume, surface, and double bounce scattering, respectively. The Freeman-Durden decomposition is particularly well adapted to the study of vegetated areas [18]. Thus, it is widely used for multitemporal wetland monitoring to track changes of shallow open water to flooded vegetation [26].

Scattering mechanism information can also be obtained using compact polarimetric SAR data. Two decomposition methods are commonly used. The first is the m-δ decomposition method [11], which is based on the degree of polarization of the backscattered signal $m = \sqrt{g_1^2 + g_2^2 + g_3^2}/g_0$ and the relative phase $\delta = \operatorname{atan}\left(g_3/g_2\right)$ and has the form [11]:

$$\begin{bmatrix} V_d \\ V_v \\ V_s \end{bmatrix} = \begin{bmatrix} \sqrt{g_0 m \frac{(1-\sin\delta)}{2}} \\ \sqrt{g_0(1-m)} \\ \sqrt{g_0 m \frac{(1+\sin\delta)}{2}} \end{bmatrix} \tag{14}$$

where V_d, V_v, and V_s refer to double bounce, volume, and surface scattering mechanisms, respectively. The second decomposition method is the m-χ decomposition [25], which is based on the degree of polarization m and the ellipticity $\chi = \operatorname{asin}\left(-g_3/mg_0\right)/2$, and has the form [25]:

$$\begin{bmatrix} P_d \\ P_v \\ P_s \end{bmatrix} = \begin{bmatrix} \sqrt{g_0 m \frac{(1+\sin 2\chi)}{2}} \\ \sqrt{g_0(1-m)} \\ \sqrt{g_0 m \frac{(1-\sin 2\chi)}{2}} \end{bmatrix} \tag{15}$$

where P_d, P_v, and P_s refer to even bounce, volume, and odd bounce scattering mechanisms, respectively.

3. SAR wetland applications

3.1. Change detection

The accurate, effective, and continuous identification and tracking of changes in wetlands is necessary for monitoring human, climatic and other effects on these ecosystems and better understanding of their response. Wetlands are expected to be even more dynamic in the future with rapid and frequent changes due to the human stresses on environment and the global warming [27]. Different methodologies can be adopted to detect and track changes in wetlands using SAR imagery, depending on the type of the change and the available polarization option. For example, a change in the surface water level of a wetland area due to e.g. heavy rainfall could extend the wetland water surface, causing flooding in the surrounding areas. Such a change can be easily detected using SAR amplitude images before and after the event acquired with similar acquisition geometry. The specular scattering of the radar signal can highlight the open water areas (dark areas due to low returned signal). Spatiotemporal changes in wetlands as dynamic ecosystems could be interpreted using SAR amplitude imagery only. This is because changes within wetlands could change the surface type illuminated by the radar. Sometimes, the change could be more complex with alternations in surface water, flooded vegetation and upland boundaries. In this case, the additional polarimetric information from full or compact polarimetric SAR is necessary for the detection and interpretation of changes within wetlands.

As shown in **Figure 7**, a change within a wetland from wet soil with a high dielectric constant to open water is usually accompanied with a change in the radar backscattering from surface scattering with a strong returned signal (**Figure 7a**) to specular reflection with a weak returned signal (**Figure 7b**). The change in wetland could also be due to its seasonal development over time. Hence, intermediate marsh with large vegetation stems properly oriented could allow for double bounce scattering mechanism (**Figure 7c**). As the marsh develops, the strong observed double bounce scattering mechanism gradually decreases in favor of the volume scattering (**Figure 7d**) from the dense canopy of the fully developed marsh [28]. Thus, polarimetric decomposition methods enable the identification of wetland classes (e.g. flooded vegetation) and monitoring changes within these classes by means of the temporal change in the backscattering mechanisms. The role of decomposition methods for identification and monitoring of wetlands was highlighted in a number of recent studies [26, 29–31]. Another way of monitoring changes within wetlands could be through polarimetric change detection methodologies using full [10], compact [10, 32], or even coherent dual [33] polarized SAR imagery. These methodologies are based on polarimetric coherency/covariance matrices. Herein, changes are flagged without information about the scattering mechanisms, which occurred during the scattering process. Test statistics, such as those proposed in [34, 35], were proven effective for polarimetric change detection over wetlands.

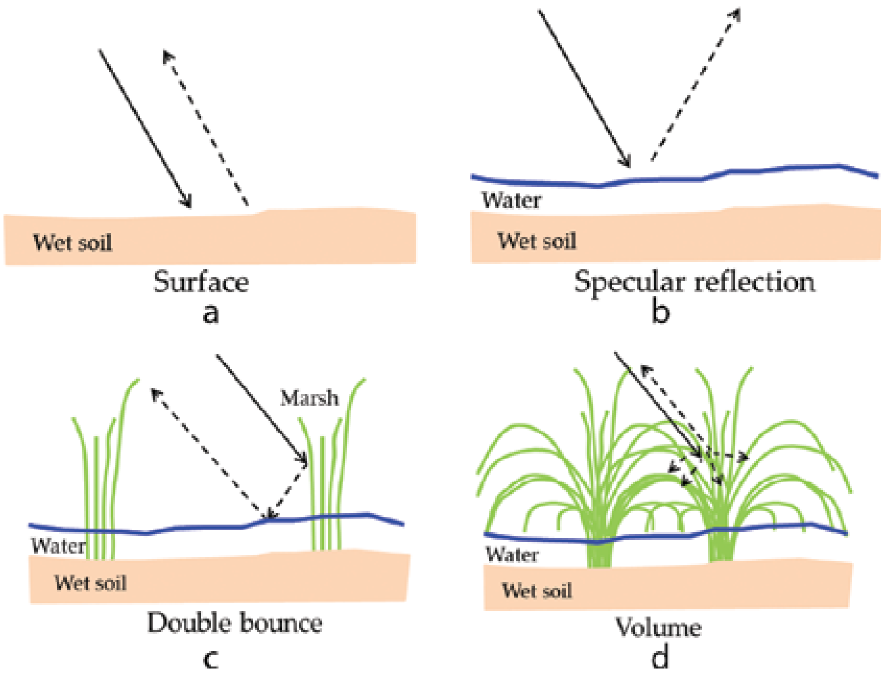

Figure 7. (a) Surface scattering mechanism from wet soil, (b) radar signal reflection from shallow open water, (c) double bounce scattering mechanism from signal interaction with vegetation stems and water surface and (d) volume scattering due to random scattering within the dense flooded vegetation canopy.

3.2. Wetland mapping

Ever since the launch of the Earth Resources Technology Satellite (ERTS) in 1972 there has been interest in using satellite remote sensing as a tool for wetland mapping and classification because the traditional air photo and field visit approaches are too costly and time consuming [36]. Wetlands are difficult to map and classify due to a large degree of spatial and temporal variability as well as structural and spectral similarities between wetland classes. Over the last decade or so a state-of-the-art approach for wetland classification has emerged. This is an object based classification approach using multi-source input data (optical and SAR) with a machine learning classification algorithm and a quality Digital Elevation Model (DEM) for identifying terrain suitability for wetlands or surface water [37–40]. Using this approach, greater than 90% accuracy is often achieved for a wide variety of wetland classification systems [41].

The early satellite SAR systems produced single channel intensity only output data, which limited its value for land cover and wetland classification. This type of data when used synergistically with optical data improved the wetland classification compared to using optical data alone but only to a minor degree [37, 42–45]. This is largely due to the ability of the SAR wavelengths to penetrate wetland vegetation and "see" the underlying water, thereby improving the flooded vegetation class discrimination. The flooded vegetation tends to produce a double bounce scattering mechanism, as explained earlier, which increases the intensity of the backscatter. HH polarization is best for this due to the enhanced penetration in vegetation.

As one goes up the polarization hierarchy from single channel intensity only data to dual-channel, compact polarimetry, and full polarimetry data sets the information content increases and the wetland classification subsequently improves [11, 46–50]. In general, dual channel SAR's and polarization ratios outperform single channel intensity only data systems and compact polarimetric data is better than dual channel data. Fully polarimetric data consistently shows the best information content for wetland classification by using polarimetric parameters derived from the data matrices, or polarimetric decompositions such as the Cloude and Pottier [17], Freeman-Durden [18], or Touzi [24] decompositions. You can use the decompositions or polarimetric parameters such as the polarization phase difference to identify flooded vegetation due to the double bounce effect increasing intensity and producing the phase shift. The Shannon Entropy has also proven useful for wetland mapping [51] and may have some benefit for finding the transition from flooded to saturated soil and between flooded vegetation and open water. This is a two-parameter model with one parameter relating to intensity and the other polarization diversity, and it may be simpler than using the decompositions.

There have been numerous frequency effect evaluations since the early observations of enhanced scattering from flooded vegetation on SEASAT imagery [52]. This effect for swamps and many vegetated wetlands with high biomass is quite evident in L-band data due to the increased canopy penetration and better interaction with the water/trunk/stem interface, resulting in the double bounce scattering mechanism [53, 54]. It is also evident at C-band and in some cases at X-band depending on the biomass and density of the canopy and the subsequent wavelength dependent penetration [41, 55–60]. In general, X- and C-band are preferred for herbaceous wetlands and less dense canopies while L-band is preferred for woody wetlands such as swamps and other wetland classes with high biomass.

SAR data has also proven effective for mapping peatlands, which is becoming more important because of climate change and carbon emission issues [61–64]. Due to the penetration of these longer wavelengths and the ability to penetrate beneath the plant canopy, there have been some indications that L-band polarimetric SAR can be used to differentiate between bog and fen peatlands due to the sensitivity of the water flow characteristics beneath the surface [65].

SAR does not penetrate water so provides little information on invasive aquatic submersive plants, but L-band and to a lesser degree C-band have shown some success at identifying invasive *Phragmites* [66]. This tall dense invasive provides significant SAR backscatter and can be separated from other land-cover due to this characteristic and its location in the landscape. It helps to use LiDAR as well as SAR due to the relative height and landscape position of the *Phragmites* [67].

In general, one wants to use a steep incidence angle for woody wetlands or flooded vegetation mapping in order to enhance the penetration to reach the water surface and realize the enhanced scattering effect due to double bounce scattering between the vegetation and the water surface. A shallow angle may be preferred if the focus is on the open water mapping as this can enhance the contrast between the specular scattering of surface water and the flooded vegetation with volume and double bounce scattering. Recent reviews of wetland remote sensing and SAR are provided in [40, 41, 68–70].

3.3. Dynamic surface water and flooded vegetation mapping

The specular backscatter from calm water surfaces allows for easy discrimination of open water from upland and flooded vegetation using SAR data. At the same time, the double bounce scattering from flooded vegetation allows discrimination from upland and open water as described earlier. This, combined with the all-weather data collection capabilities, makes SAR an ideal sensor for mapping flood as well as dynamic surface water and flooded vegetation [40].

Flood mapping is operational with SAR data in many countries using data from a variety of SAR systems from X- to L-band (see for example [71–74]). Intensity thresholding techniques have traditionally been used for open water mapping [75]. Texture, cross-polarization data, and other techniques are being developed to solve the problem when the water is brighter due to wind or current induced roughness, as well as to automate the process [76–79].

As described in the section on wetland mapping the double bounce effect and enhanced scattering from flooded vegetation makes SAR a good sensor for mapping flooded vegetation from non-flooded vegetation [40, 59, 80]. This allows the delineation of wetland extent and with multi-temporal data can be very useful for monitoring seasonal and/or annual changes in the wetland size and extent. [81] showed that flooded vegetation tends to remain coherent using InSAR techniques and this can then be used to map wetland type and extent. Thus, SAR is an ideal sensor for monitoring the spatially and temporally dynamic flooded vegetation components of wetlands.

The development of standard coverages, like that used for the Sentinel program, results in stacks of data with the same geometry and facilitates the use of temporal filters for speckle noise reduction. The use of a multi-temporal filter rather than the conventional spatial filtering approach can be an effective way to reduce the speckle while maintaining the spatial resolution and the ability to detect small objects and edges [82, 83]. This also allows the use of intensity metrics rather than thresholds to separate water from land, which can also help solving the wind roughness problem [84]. The multi-temporal coverage provided by SAR systems enables generating hydro-period and dynamic surface water as well as flooded vegetation masks [85, 86]. This enables better mapping of temporary, seasonal and ephemeral water bodies as well as the permanent water bodies, which are static and much easier to map. A recent review of SAR flood mapping and flood studies with SAR is provided in [87], while [88] provides a review of flooded vegetation mapping with SAR.

3.4. Water level monitoring

Wetland interferometric synthetic aperture radar (InSAR) is a relatively new application of the InSAR technology that detects water level changes over wide areas with 5–100 m pixel resolution and several centimeters vertical accuracy [89–91]. The wetland InSAR technique works where vegetation emerges above the water surface due to the “double bounce” effect, in which the radar pulse is backscattered twice from the water surface and vegetation [53]. InSAR observations were successfully used to study wetland hydrology in the Everglades [90–93], Louisiana [94–96] and the Sian Ka’an in Yucatan [97].

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One of the key issues in using the InSAR observations for assessing wetland hydrology is the calibration of the InSAR observations, which are relative in both space and time. In time, the measurements provide the change in water level (not the actual water level) that occurred between the two data acquisitions. In space, the measurements describe the relative change of water levels in the entire interferogram with respect to a zero change at an arbitrary reference point, because the actual range between the satellite and the surface cannot be determined accurately. However, the relative changes between pixels can be determined at the cm-level. In many other InSAR applications, such as earthquake or volcanic induced deformation, the reference zero change point is chosen to be in the far-field, where changes are known to be negligible [98]. However, in wetland InSAR, the assumption of zero surface change in the far-field does not hold, because flow and water levels can be discontinuous across the various water control structures or other flow obstacles.

The calibration stage requires additional information on water level changes, which can be derived from various sources. In areas monitored by stage (water level) stations, as in the Everglades, the stage data can be used for the InSAR calibration, as conducted by [90]. Another calibration technique relies on spaceborne radar altimetry, which detects absolute water level changes over a few km wide footprints with accuracy of 5–10 cm [94]. However, the altimetry observations are limited in space and time, as the radar altimeter data can only be acquired along the satellite tracks, which are spaces roughly 100 km apart. Also, the altimetry data is not always synchronized with the InSAR observations, which are acquired by different satellites.

3.5. Wetland biomass estimation

Wetland biomass is of increasing interest due to methane emission contributions to climate change from degraded and thawing wetlands. Wetland change can also be used as an indicator of climate change impacts. Wetland vegetation biomass can therefore be an important indicator of carbon sequestration in wetlands and is essential for understanding the carbon cycle of these ecosystems. SAR data has the potential to estimate vegetation biomass in wetlands because radar is particularly sensitive to the vegetation canopy over an underlying water surface [99]. The biomass of totora reeds and bofedal in water-saturated Andean grasslands was mapped with ERS-1 data in [100]. The goal was to protect this ecosystem from overgrazing. They found that the backscatter signal of ERS-1 was sensitive to the humid and dry biomass of reeds and grasslands and their biomass maps were useful for the livestock management in the study region. [101] developed regression and analytical models for estimating mangrove wetland biomass in South China using RADARSAT images. [102, 103] also found that L-band ALOS PALSAR can be used to estimate the aboveground biomass because of the correlation between HH and HV backscatter signals. C-band backscatter characteristics from RADARSAT-2 data were used by [104] to estimate the biomass of the Poyang Lake wetlands in China. Also, [105] used ENVISAT ASAR data to estimate wetland vegetation biomass in Poyang Lake. These studies have shown that it is possible to estimate above water biomass in wetlands with SAR data.

4. Trends in wetland mapping and monitoring with SAR

As shown in the previous section, spaceborne SAR remote sensing technology is recognized as essential tool for effective wetland observation. With the presence of global warming and its associated risks on Earth systems, there is an expressed interest in increased temporal and spatial resolution of satellite measurements. Thus, a trend toward increased temporal and spatial resolution of SAR imagery is noted in recent and future SAR missions. The Sentinel-1 SAR mission with its two identical SAR satellites (Sentinel-1A&B) is a good example of a recent SAR mission with a spatial resolution ranging from 5 m to 100 m and a revisit time of 6 days. This high temporal and spatial resolution is expected to be even higher in the near future with the launch of the RCM in late 2018. The RCM is expected to provide SAR imagery in a spatial resolution ranging from 1 m to 100 m, in a revisit time of only 4 days [32]. The increased temporal and spatial resolution would be required to adequately monitor wetlands and characterize the actual implications of climate change. Also, it is expected to further improve our understanding of climate change in wetlands and water quality, allowing ecosystem managers and decision makers to have sufficient information regarding wetland preservation.

With the availability of different remote sensing data with various information contents, the application of multi-source data for advanced wetland applications is demonstrated in a number of studies; see for example [2, 44, 61, 67, 106]. In addition to SAR imagery, experiments on the integration of topographic and remote sensing data, such as optical imagery and LiDAR data, were conducted. The ultimate objective of these experiments was the improved mapping accuracy of wetlands. The integration of SAR imagery with optical and topographic data from multiple sensors was shown in [44, 106] to be necessary for improved wetland mapping and classification during the growing season. However, the integration of SAR imagery and LiDAR data did not improve significantly the classification accuracy of wetland in [61, 67]. The modern advances in remote sensing technology and the availability of multi-source information are shifting the manner in which Earth observation data are used for wetland monitoring, indicating the need for automated and efficient techniques. Different studies, such as [2, 44, 61, 106], have highlighted the effectiveness of machine learning algorithms for automated wetland classification. An example of these algorithms is the Random Forest (RF) classification algorithm proposed in [107]. This shift toward the automated machine learning algorithms comes to fulfill the requirement for operational wetland monitoring systems.

The continuing advancements in computer processing power and software development as well as the trend toward free and open access to remote sensing imagery, such as those from the current Sentinel satellites and the future RCM, are enabling the ingestion of data into a centralized archive. This also supports the application of a standard rapid processing chain to generate analysis-ready wetland products. The provision of analysis-ready products to a wide range of users would revolutionize the role of remote sensing in Earth system science [108].

5. Conclusions

This chapter highlighted the SAR remote sensing technology and its potential for wetland monitoring and mapping. It was shown that a wide range of wetland applications can be addressed using SAR remote sensing imagery. SAR data with enhanced target information provided by full or compact polarimetric SAR systems can provide information for advanced wetland applications. In many studies, the information about the polarimetric scattering mechanisms was found necessary for observing the temporal development of wetlands and detecting their changes. This chapter shows that the fusion of multi-source data improves wetland mapping, especially during the growing season. Furthermore, a relatively new application of the InSAR technology is currently implemented for water level monitoring. Given the problem of climate change, wetland biomass estimation using SAR imagery is becoming necessary for the evaluation of methane emission contributions to climate change from degraded and thawing wetlands. The current advanced computing capabilities along with the shift toward free and open access remote sensing data are enabling analysis-ready products for a wide range of users.

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Conflict of interest

Authors declare no conflict of interest.

Acronyms and abbreviations

SAR	synthetic aperture radar
HH	horizontal transmitted horizontal received signal
VV	vertical transmitted vertical received signal
HV	horizontal transmitted vertical received signal
VH	vertical transmitted horizontal received signal
RH	right circular transmitted horizontal received signal
RV	right circular transmitted vertical received signal
RISAT-1	Radar Imaging Satellite-1
ALOS-2	Advanced Land Observing Satellite-2
PALSAR-2	Phased Array type L-band Synthetic Aperture Radar-2
RCM	RADARSAT Constellation Mission

InSAR	interferometric synthetic aperture radar
RF	random forest
DEM	digital elevation model

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Chapter 6

Mapping and Monitoring Wetland Dynamics Using Thermal, Optical, and SAR Remote Sensing Data

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Additional information is available at the end of the chapter

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Abstract

Wetlands are transition zone where the flow of water, the cycling of carbon and nutrients, and the energy to form a unique ecosystem are characterized by its hydrology, soils, and vegetation, between dryland and water. Over the years, remote sensing techniques have proven to be a successful tool for monitoring wetlands. Both optical and microwave earth observation sensors can be used for monitoring wetlands. Land surface temperature (LST), as one of the most important variables in physical processes of the Earth, is one of the unexplored parameters for studying wetland dynamics. In this chapter, seasonal LST, SAR data values (dual polarization VV + VH), as well as the seasonal normalized difference water index will be explored, and the relation between them will be analyzed. For this purpose, satellite images from Landsat 8 and Sentinel-1, over a wetland area, were downloaded, preprocessed, and analyzed. As a study case, Seyfe Lake located in the central Anatolian part of Turkey has been selected. The results show Seyfe Lake's seasonal dynamics and the relation between the investigated parameters. The results helped in understanding the wetland seasonal dynamics which can be used in better managing and monitoring wetlands using remote sensing data.

Keywords: wetlands, remote sensing, land surface temperature, Landsat, synthetic aperture radar, normalized difference water index

1. Introduction

As one of the most productive natural ecosystems, wetlands are of great significant importance for hydrological and ecological processes. Wetlands have remarkable features in the landscape, which provide numerous beneficial services for people, fish, and wildlife. Wetlands can be defined as areas filled or soaked with water at least for one part of the year. The complex hydrology of wetlands controls the source, amount, and temporal and spatial distribution of sediment and nutrient movements and influences the distribution of flora and fauna [1]. Over the past decades, threats like global climate change, and land-use conversion, arise the vulnerability of wetlands, and it is known that since the 1900s, the loss of wetlands has gained considerable attention with more than 50%. Thus, a need for continuous monitoring of the wetlands and their behavior is crucial for their protection and sustainable management. Remote sensing techniques are often less costly and time-consuming for large geographic areas compared to conventional field mapping [2] and have been a successful tool for monitoring wetlands in the past few decades.

1.1. A short review on remote sensing for wetland mapping and monitoring

Over the past few decades, remote sensing technology has been the most effective tool to acquire both spatial and temporal wetlands data. Remote sensing data has been used in a number of wetland research areas such as mapping wetland changes [3–5], carbon cycle and climate warming in wetland environments [6], and hydrology dynamics in wetlands [4]. A recent review of wetland remote sensing [7] states that the number of wetland remote sensing publication has drastically risen since the 1990s. In the same review, the contribution of medium spatial resolution data for wetland studies is elaborated. Both optical and microwave earth observation sensors can be used for monitoring wetlands; thus, optical satellites are mostly effective in vegetation monitoring as well as the wetlands' change mapping [8]. However, data from optical satellites have been used for mapping surface wetland water using different techniques that applied range in complexity and general applicability [9–11].

Synthetic aperture radar (SAR) technology provides significantly important data with its ability to image landscape through cloud cover, day and night, which often can be a limitation for optical sensors. Over the years, SAR data have been successfully used in different land-use/land cover applications. Recently, the European Space Agency launched two twin SAR satellites in order to continue the ERS and Envisat missions, Sentinel-1A and Sentinel-1B. The satellites carry a C-band (~5.7 cm wavelength) SAR instrument offering data products in single (HH or VV) or double (HH + VH or VV + VH) polarization. With the launch of the Sentinel-1 mission, a new era for SAR mapping has begun; thus, the use of C-band in monitoring changes and mapping wetlands has increased. Research cover is not limited to the following topics: mapping and characterization of hydrological dynamics [12], short-term change detection in wetlands [13], assessment of carbon flux and soil moisture in wetlands [14], as well as a combination of Sentinel-1 and Sentinel-2 data for wetland classification [15]. The relation between radar and optical/thermal data can be significant for better understanding and managing wetlands. Several studies have investigated the relation between different satellite data in different land covers [16]. However, the relation between SAR values and land surface temperature values within a wetland area has not been a subject of a delicate investigation.

Land surface temperature (LST) is one of the most important variables in physical processes of the Earth, and it is one of the unexplored parameters for studying wetland dynamics [17]. LST is closely related to the surface energy balance and the water status of the land cover, and it depends on the radiative energy that the land absorbs [18]. With the latest technological developments in remote sensing, many Earth observation satellites like Landsat, Sentinel-3, MODIS, and ASTER operate in the thermal infrared region offering thermal bands for retrieving thermal maps of the Earth's surface. Landsat 8 is the latest satellite from the Landsat legacy, and it offers 100-m thermal data. Retrieving LST using Landsat data has been the subject in many studies resulting in several methods and algorithms [19, 20]. LST from satellite images has been used in different studies such as studies related to urban heat islands [21], earthquake monitoring [22], water extraction [23], climate changes, etc. One of the important parameters to understand the extensive range of existing processes in the wetland areas is the LST [24].

Normalized difference water index (NDWI) is one of the several indices commonly used in wetland studies [7]. The first remote sensing water index developed in 1995 is similar to the simplicity of the normalized difference vegetation index (NDVI); the NDWI uses two narrow channels centered near 0.86 and 1.24 μm or near the infrared and the short-wave infrared band which measures the liquid water molecules in [25] vegetation canopies that have interacted with solar radiation. Afterward, new water index developed to extract open water features using the green and the near-infrared band was proposed [26] and later modified by applying short-wave infrared instead of the near-infrared [27]. However, the use of short-wave infrared for water bodies' extraction within a wetland area is preferable, particularly in the presence of high level of suspended sediment. The NDWI product varies between −1 and +1 and depends on the water content in the observed area. Higher values correspond to high water content. NDWI values higher than zero are considered to be open water areas, while values close to zero are considered to be contents with high moisture.

The goal of this chapter is to present the seasonal wetland dynamics using SAR data values from Sentinel-1, LST data, as well as modified normalized difference water index (MNDWI) values retrieved from Landsat 8 compared with field measurements. Also, the relation between these significant parameters will be investigated and discussed.

1.2. Landsat 8 data

Starting from 1972, the Landsat program is the longest Earth observation platform. The most recent satellite, Landsat 8, has been launched in February 2013. Over the years, the Landsat data as a valuable resource for global research has been used in different applications. Landsat 8 carries two instruments, operational land imager (OLI) and thermal infrared sensor (TIRS). OLI collects data from nine, while TIRS collects data from two spectral bands (**Table 1**).

Taking into consideration the complex construction of wetland as a transition between terrestrial and open water aquatic ecosystems [28], wetlands and their properties are not easily detectable with optical satellite sensors [29]. However, wetlands are more sensitive to some parts of the electromagnetic spectrum than others. Thus, the near-infrared is sensitive to biomass content, in combination with the green band, and can give valuable information for the water content, as well as for the soil wetness [30]. On the other hand, the short-wave infrared bands are more sensitive to a moisture content of both soil and vegetation, and they are particularly useful in separating wetland from dry lands.

Spectral band	Wavelength (μm)	Resolution (m)
Band 1—Coastal/Aerosol	0.433–0.453	30
Band 2—Blue	0.450–0.515	30
Band 3—Green	0.525–0.600	30
Band 4—Red	0.630–680	30
Band 5—NIR	0.845–0.885	30
Band 6—SWIR-1	1.560–1660	30
Band 7—SWIR-2	2.100–2.300	30
Band 8—Panchromatic	0.500–0.680	15
Band 9—Cirrus	1.360–1.390	30
Band 10—LWIR-1	10.30–11.30	100
Band 11—LWIR-2	11.50–12.50	100

Table 1. Landsat 8 band specification.

As a follower of the Landsat 8 mission, the upcoming Landsat 9 which is expected to be launched in 2021 will continue to observe the Earth and collect valuable data for researchers all over the world.

1.3. Sentinel-1 data

The latest SAR satellite developed by the European Space Agency, Sentinel-1, is an imaging radar satellite at C-band (5.405 GHz) consisting of a constellation of two satellites, Sentinel-1A, launched on 3 April, 2014, and Sentinel-1B, launched on 22 April 2016. The C-SAR instruments support the operation of dual polarization: HH + HV and VV + VH. Their main cover applications are monitoring sea ice zones and the Arctic environment; surveillance of marine environment; mentoring land surface motion risks; mapping of land surfaces like forest, water, and soil and agriculture; and mapping in support of humanitarian aid on crisis situation [31, 32].

As a result of a number of environmental circumstances and system factors, satellite images can be often distorted in geometry and brightness which requires preprocessing of the images before their use [33]. SAR data are exposed to radiometric and geometric distortions that should be removed or minimized. The preprocessing of the Sentinel-1 SAR data can be easily done with the Sentinel-1 Toolbox integrated in SNAP, and it contains few steps: (i) data preparation, (ii) radiometric calibration, (iii) multilooking, (iv) spackle reduction, (v) terrain correction, and (vi) dB conversion.

The data preparation consists of selecting the study area, selecting the data type needed for the study, selecting the date, and downloading the SAR product.

Radiometric calibration corrects the SAR image so that the pixel values represent the radar backscatter of the reflected surface. SAR calibration provides imagery in which the pixel values can be directly related to the radar backscatter of the scene.

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Multilooking processing is used in order to produce a product with nominal image pixel size [34]. Multilooks can be generated by averaging over the range and/or azimuth resolution cells which improves the radiometric resolution but degrades the spatial resolution of the SAR image. Multilooking can be an optional processing since it is not necessary when terrain correction is applied to an image.

Compared to optical image data, the biggest difference in the appearance of radar imagery is its poor radiometric quality [33]; thus, it is difficult to make a visual interpretation of a SAR image. Speckle can be caused by random constructive and destructive interference resulting in salt and pepper noise over the SAR image [34]. As speckle is one of the biggest noises in SAR data, it should be reduced before performing any analyses.

Terrain correction geocodes the image by correcting SAR geometric distortions with the help of digital elevation model (DEM), and it produces a map projected product. With geocoding the image is being converted from slant range or ground range geometry into a map coordinate system. Terrain correction corrects SAR geometry effects such as foreshortening, layover, and shadows.

The last step of the preprocessing of Sentinel-1 SAR image is to convert the image in decibel scaling which can be done automatically in SNAP with right-click on the terrain-corrected image and selection of the linear to/from dB option (Eq. (1)):

$$\sigma^{\circ}\mathrm{dB} = 10\ \log_{10}\left(\sigma^{\circ}\right) \tag{1}$$

where $\sigma^{\circ}dB$ is sigma nought in decibel scale and σ° is the radiometric calibrated/speckle-reducted/terrain-corrected Level-1 SAR product.

2. Case study

In order to investigate wetlands' condition and seasonal changes, in this chapter, we review case study of thermal, optical, and radar remote sensing data and their relation for better understanding of wetland dynamics. As a study area, Seyfe Lake, located in the central Anatolian region, has been selected (**Figure 1**).

2.1. Study area

Turkey is the richest country in wetlands among Europe and the Middle East countries and with its geographical location plays an important part in the migration path for water birds. Seyfe Lake is located in the central Anatolian region, or 200 km northeast from Turkey's capital Ankara and 30 km east from Kirsehir. Seyfe Lake is a salty internal lake formed the base of a closed catchment area in a tectonic pit covering area approximately 10,700 ha. Over the years over 186 different bird species have been observed. Seyfe Lake is a first-degree natural site area under the protection of natural structure and ecological character pledged by the Ramsar Agreement in 1994. The lake is surrounded with agricultural areas, and according to the field

Figure 1. 14 June 2018, Seyfe Lake.

Figure 2. Seyfe Lake, June 1987 (left) and June 2018 (right).

observation on 14 June 2018, most of the agricultural fields around the lake are wheat, barley, and chickpeas.

This internationally valuable wetland area has been losing its value over the years. As seen in **Figure 2**, the water area has drastically decreased since 1987.

2.2. Data and methods

Satellite images from both Landsat 8 and Sentinel-1 satellites were used for observing the seasonal changes in Seyfe Lake. As the time schedule is different for both of the satellites, the images with the smallest time gap were chosen for further investigation. Thus, from five dates, three were approximately 2 days apart, while two images were taken on the same day (**Table 2**).

The methods in this study include both field measurements and remote sensing measurements and techniques (**Figure 3**). The field measurements were performed on 14 June 2018, while the remote sensing data used in the seasonal change analyses were from different months in 2017.

Landsat-8	Sentinel-1	Difference
19 February 2017	21 February 2017	~2 days
23 March 2017	23 March 2017	~10 h
27 June 2017	27 June 2017	~10 h
30 August 2017	01 September 2017	~2 days
17 October 2017	19 October 2017	~2 days

Table 2. Satellite data acquisitions over the study area, Seyfe Lake.

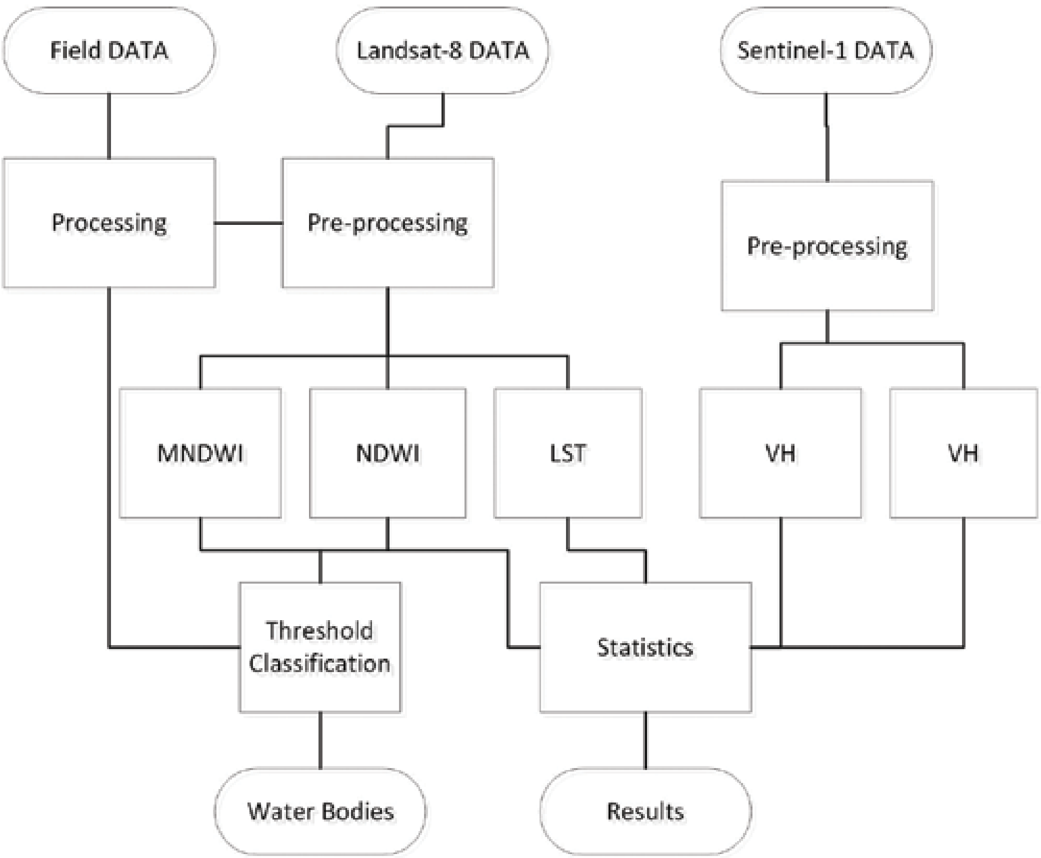

Figure 3. Flowchart of the used methodology.

The field measurements were used for comparison and more accurate classification of the water objects in the study area. Using Landsat 8 satellite data, LST, NDWI, and MNDWI were extracted, and afterward a threshold analysis and classification were made. Sentinel-1 data were preprocessed and the values from the dual polarization data were obtained. The relation between the optical, thermal, and radar satellite data was also investigated. The NDWI, MNDWI, LST, and the dual polarization (VV, VH) values were extracted using approximately

160 random points. The same points were used in all of the satellite images used in this study and were then used for several statistical analyses.

2.2.1. Field measurements

Before going to the field, the availability of satellite imagery was investigated. As the Landsat 8 satellite has a temporal resolution of 16 days, the overpass over the Seyfe Lake was estimated on 14 June 2018. As expected, the overpass happened on 14 June 2018, around 8:20 Coordinated Universal Time (UTC), or around 11:20 local time. The field measurements were taken around 11:00–12:00 local time. At the time of the measurements, the weather was clear, sunny, and hot with a maximum air temperature of 33°C. According to the meteorological records, in the past 2 weeks, three light rains have occurred in the study area, where 3 days before the field measurements, it has rained with over 1 mm (1 kg/m^2).

Several parameters of the study area were taken into consideration during the field measurements: soil moisture, soil texture, soil color, and surface roughness.

The measurements were done 2 km along the study area (**Figure 4**).

According to the field measurements based on the feel of wet soil, the soil at all measurement points can be described as silty clay [35]. The soil moisture observation in the study area showed that three different types of soil can be distinguished: dry soil, moist soil, and wet soil. At the first measurement station (**Figure 4**, detail 1), approximately 3–5 m of the soil can be classified as dry soil with a minimal to no vegetation cover. However, since the spatial resolution of used satellite images, Landsat 8 is 30 m, it is impossible to separate the mentioned dry

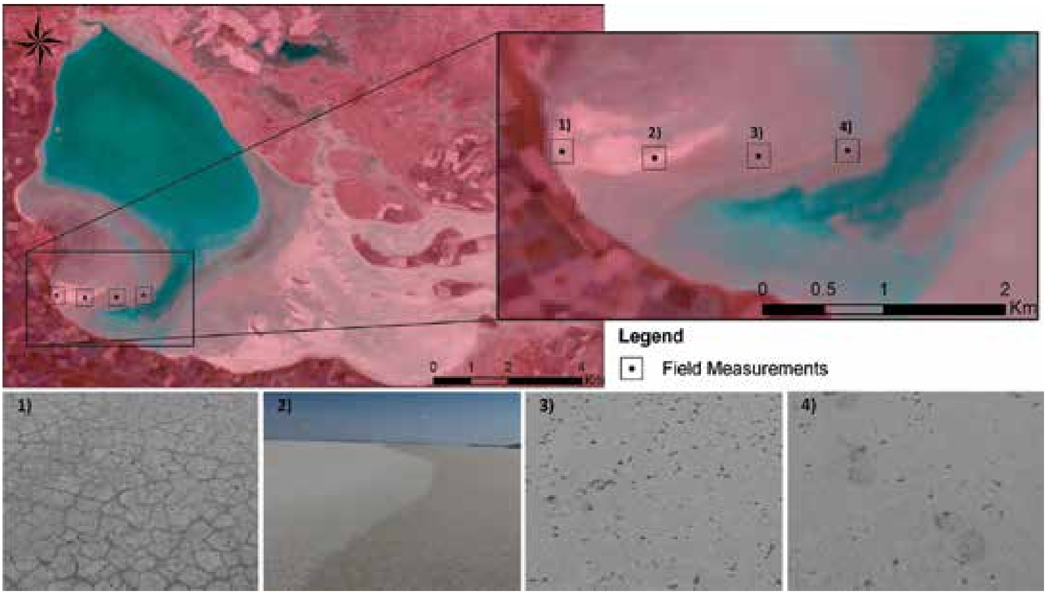

Figure 4. Field measurements details, satellite image (red, green, blue: 6, 3, 4).

soil from the further investigated land cover. Further ahead, when touched with hand, the level of moist in the soil could be felt, and the level of moist becomes even higher at the second measurement station, where the difference could be both felt and seen as in **Figure 4**, details 2 and 3. At the fourth measurement station (**Figure 4**, detail 4), the soil was significantly wet, and the field conditions did not allow further investigation/measurements.

2.2.2. NDWI threshold analysis

Using several developed spectral water indices from two or more spectral bands requires an appropriate threshold in order to extract water bodies. The NDWI values range from −1 to +1 where values higher than zero are classified as water bodies. However, with careful adjustment of the NDWI threshold, more accurate results could be achieved [25, 36]. Taking into consideration the field measurements, NDWI threshold analysis on two different water indices was made in order to separate not just the water bodies but also the wetlands in the study area. The first index uses green and near-infrared band (NDWI) [26], while the second band uses green and middle near-infrared band (MNDWI) [27]. In the first case, a threshold value of zero has been set to separate the water bodies from the other land cover features, while in the second case, values close to zero (−0.1 and 0.1) were also set as a threshold value.

2.2.3. LST estimation

With the development of the thermal remote sensing technology, retrieving LST has become the topic for a number of research. A recently developed tool in Erdas Imagine calculates the LST from Landsat 8 satellite data in few steps (**Figure 5**).

First, the top of atmospheric (TOA) spectral radiance (Lλ) is calculated using Eq. (2):

$$L\lambda = M_L * Q_{cal} + A_L - O_i \tag{2}$$

where M_L represents the band-specific multiplicative rescaling factor, Q_{cal} is the value of the thermal band, A_L is the band-specific additive rescaling factor, and O_i is the correction for the thermal band [37]. After the digital numbers (DNs) have been converted to reflection, the data from the thermal band are converted from spectral radiance to brightness temperature (BT) using the thermal constants provided in the metadata file Eq. (3) file (**Table 3**):

$$BT = \frac{K_2}{Ln\left[\left(\frac{K_1}{L\lambda}\right) + 1\right]} - 273.15 \tag{3}$$

Normal difference vegetation index (NDVI) is needed for further calculation of the proportion of vegetation Eq. (4):

$$NDVI = \frac{NIR(band5) - R(band4)}{NIR(band5) + R(band4)} \tag{4}$$

The proportion of vegetation is calculated according to Eq. (4). NDVI values for vegetation and soil ($\mathrm{NDVI}_v = 0.5$ and $\mathrm{NDVI}_s = 0.2$) are suggested to apply in global conditions [38]:

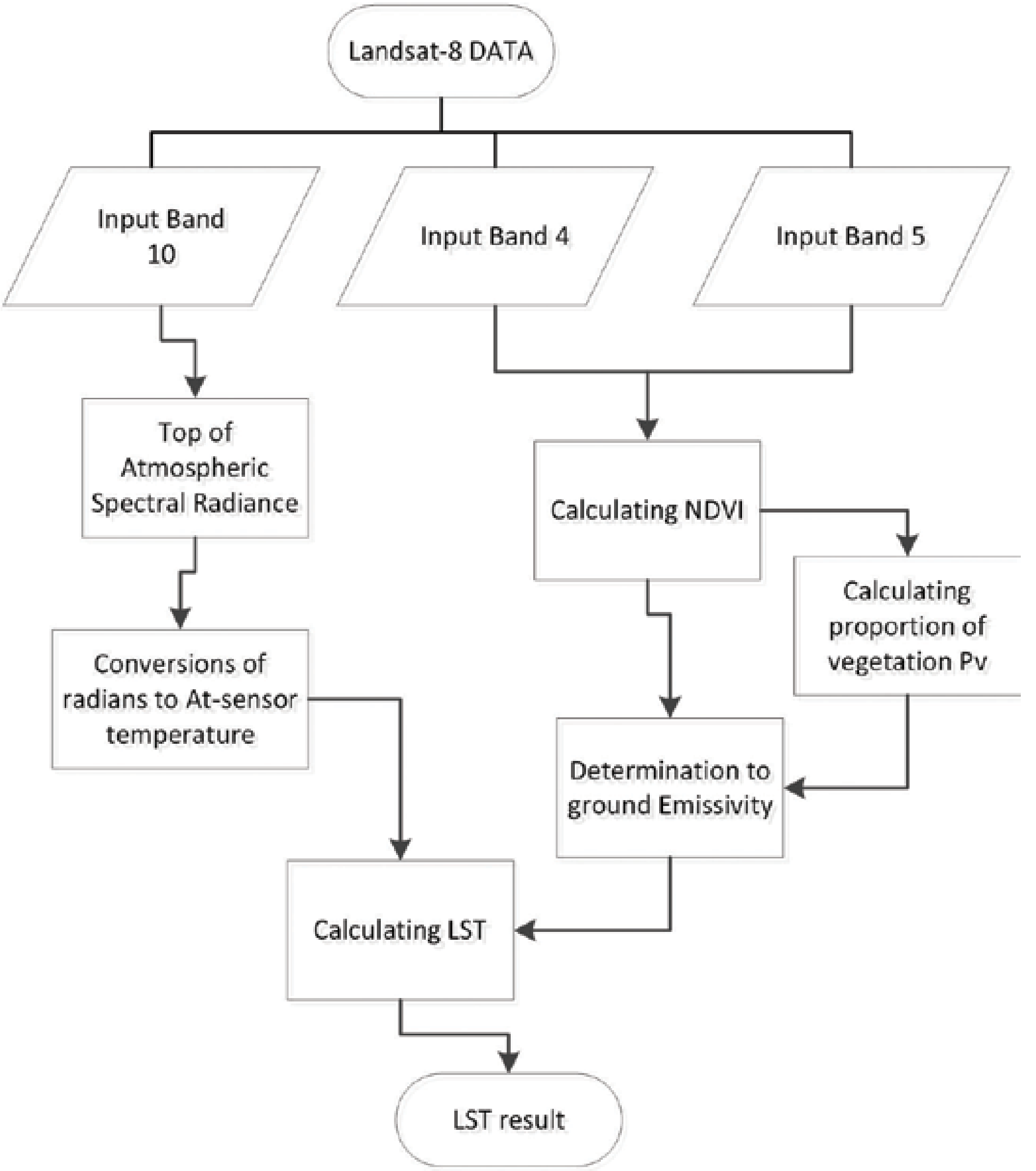

Figure 5. Flowchart of the LST calculation.

$$P_V = \left(\frac{NDVI - NDVI_s}{NDVI_v - NDVI_s}\right)^2 \tag{5}$$

The emissivity can be calculated following Eq. (6):

$$\varepsilon_\lambda = \varepsilon_{v\lambda} P_v + \varepsilon_{s\lambda}(1 - P_v) + C_\lambda \tag{6}$$

where ε_v and ε_s are the vegetation and soil emissivities, respectively, and C represents the surface roughness taken as a constant value of 0.005 [39]. The condition can be represented

Thermal Constant–Band 10	
K1	1321.08
K2	777.89
Rescaling Factor–Band 10	
ML	0.000342
AL	0.1
Correction–Band 10	
Oi	0.29

Table 3. Metadata of Landsat 8 satellite image.

with Eq. (7) where the emissivity constant values are 0.991 for water, 0.962 for built-up areas/ bare soil, 0.966 for a mixture of soil and vegetation, and 0.973 for vegetated areas [19]:

$$\varepsilon_\lambda = \begin{cases} \varepsilon_{s\lambda}, & NDVI < NDVI_s \\ \varepsilon_{v\lambda} P_v + \varepsilon_{s\lambda}(1 - P_v) + C, & NDVI_s \leq NDVI \leq NDVI_v \\ \varepsilon_{s\lambda} + C, & NDVI > NDVI_v \end{cases} \tag{7}$$

NDVI values lower than 0 are considered to be water, NDVI values between 0 and 0.2 are considered to be bare soil, and NDVI values between 0.2 and 0.4 are considered to be mixtures of soil and vegetation cover; when the NDVI value is greater than 0.4, it is considered to be covered with vegetation.

The LST or the emissivity-corrected land surface temperature T_s is computed with Eq. (8) [40]:

$$T_s = \frac{BT}{\left\{1 + \left[\frac{\lambda BT}{\rho} ln\, \varepsilon_\lambda\right]\right\}} \tag{8}$$

where Ts is the LST in Celsius, λ is the wavelength of emitted radiance (λ = 10.895) [41], and ρ is a constant calculated with Eq. (9):

$$\rho = h\frac{c}{\sigma}\ (1.438\ x\ 10\text{–}2\ m\ K) \tag{9}$$

where σ is the Boltzmann constant (1.38 × 10–23 J/K), h is Planck's constant (6.626 × 10–34 J s), and c is the velocity of light (2.998 × 108 ms^{-1}).

3. Results

3.1. Field measurements: NDWI threshold

Comparing the field measurements with remote sensing data, different threshold NDWI and MNDWI values were set for more accurate classification. Since both NIR and SWIR parts of the electromagnetic spectrum are sensitive to water and wet areas, in this study, both NDWI and MNDWI were considered. The study area, Seyfe Lake contains water area, wetland area, and dry area. The dry area is known to be salt soil [42]. For the threshold value, salt soil areas were not taken into consideration.

As seen in **Figure 6**, the SWIR band is more sensitive to water bodies, as well as to soil moisture. While NIR can only detect the open water bodies, with SWIR beside open water bodies, both shallow water and moist soil can be distinguished from the other land covers.

The higher values of zero thresholds for both NDWI and MNDWI indices indicate water body areas. As seen in **Figure 7**, where the Landsat image is a combination of SWIR, red and green bands, the NDWI threshold results did not classify the shallow water areas as a water body while using the MNDWI successfully extracted both open water bodies and shallow water bodies. However, setting one threshold can only separate the land cover into two classes, in this case, water (NDWI > 0) and other (NDWI < 0), and the wet soils cannot be distinguished from the other land covers. Setting additional threshold values in both NDWI and MNDWI values helped us to distinguish two more classes, shallow water and moist soil, or wetland (**Figures 7** and **8**).

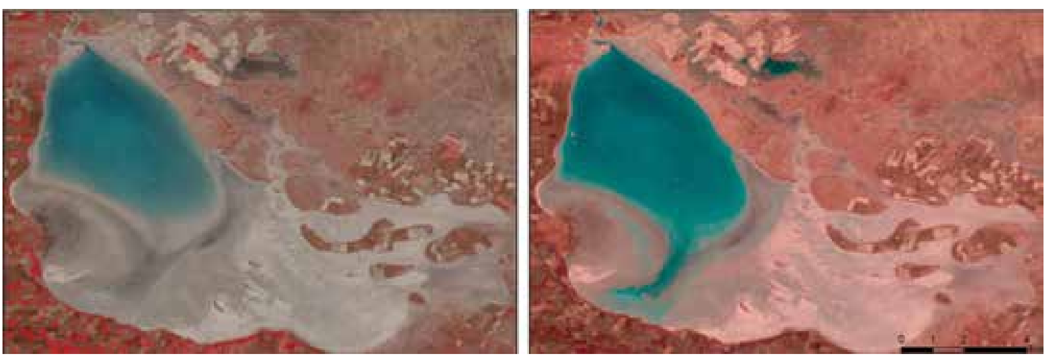

Figure 6. Seyfe Lake: RGB-NIR, red and green (left); RGB-SWIR, red and green (right).

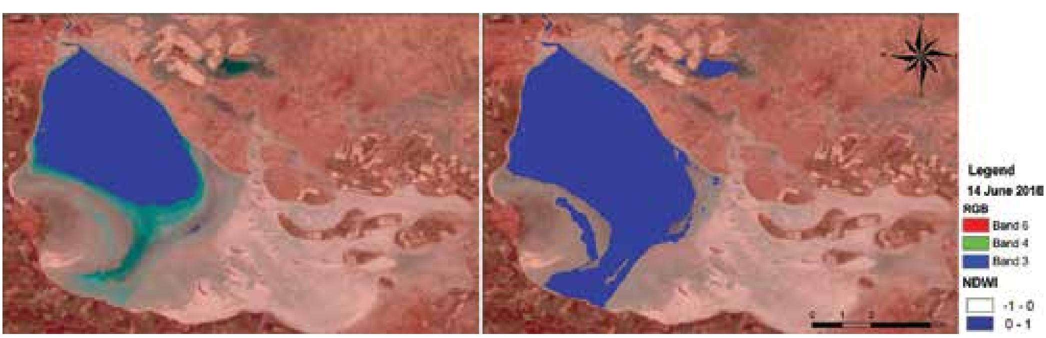

Figure 7. NDWI > 0 (left); MNDWI > 0 (right).

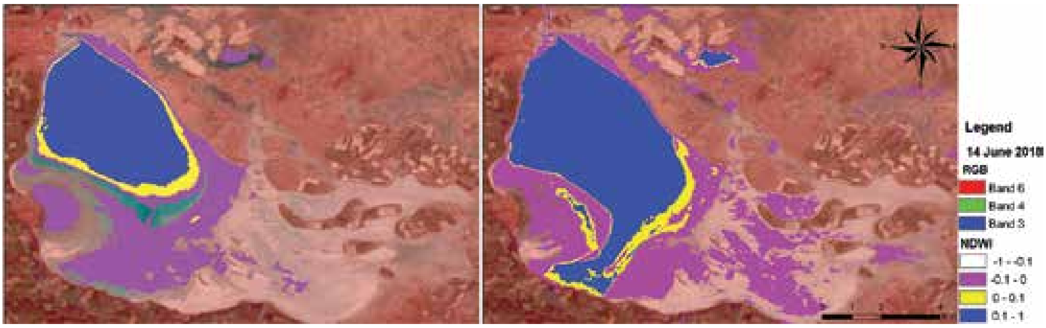

Figure 8. NDWI (left); MNDWI (right).

As it can be seen in **Figures 7** and **8**, the threshold of zero in the NDWI was not successful in extracting shallow water area, while the MNDWI successfully extracted both water object and shallow water areas, but both indices failed in extracting the wet soil within the study area. On the other hand, setting two additional thresholds close to zero (−0.1 and 0.1) showed successful results in distinguishing wetland, shallow water, and water bodies from the other land cover features. In the visual comparison with the field data, the results from the additional thresholds were satisfactorily accurate.

3.2. Seasonal changes

The results from the seasonal changes in the Sefye Lake showed drastic changes over the months. Using both NDWI and MNDWI, according to the field measurements, four classes have been determined, wet soil, salt soil, shallow water, and water. In **Figure 9**, the Sentinel-1, the NDWI classification, and the RGB images from every investigated month are given.

Using the pixel number of every class, the area of each class was calculated, and the results are presented in **Figure 10**. As it can be seen, after March, the water class has decreased for more than 75%, and the area of the shallow water and wet soil has not changed, while the area or the salt soil has significantly increased. The results from August have the highest area of salt soil and the lowest area of water, shallow water, and wet soil, while in October the areas of the salt soil class have decreased, and the areas of the wet soil and shallow water classes have increased. While the water area has been calculated to be more than 20 km^2 in March, in October the water area has been calculated to be 0 km^2.

LST is one of the most important parameters for wetland dynamics (**Figure 11**). The relation between the average LST values and the seasonal changes in Seyfe Lake using NDWI indices has also been investigated. The statistics between these two variables indicates a strong correlation between the LST changes and the changes in every class separately. Thus, the correlation between the wet soil (−0.51), shallow water (−0.58), and water (−0.75) indicates a negative relation, meaning that with the increase of the LST, the values of the mentioned classes decrease. The correlation between the LST and the salt soil has been calculated to be more than 0.8, meaning, with the increase of the LST, the increase of the salt soil (dry soil) area occurs.

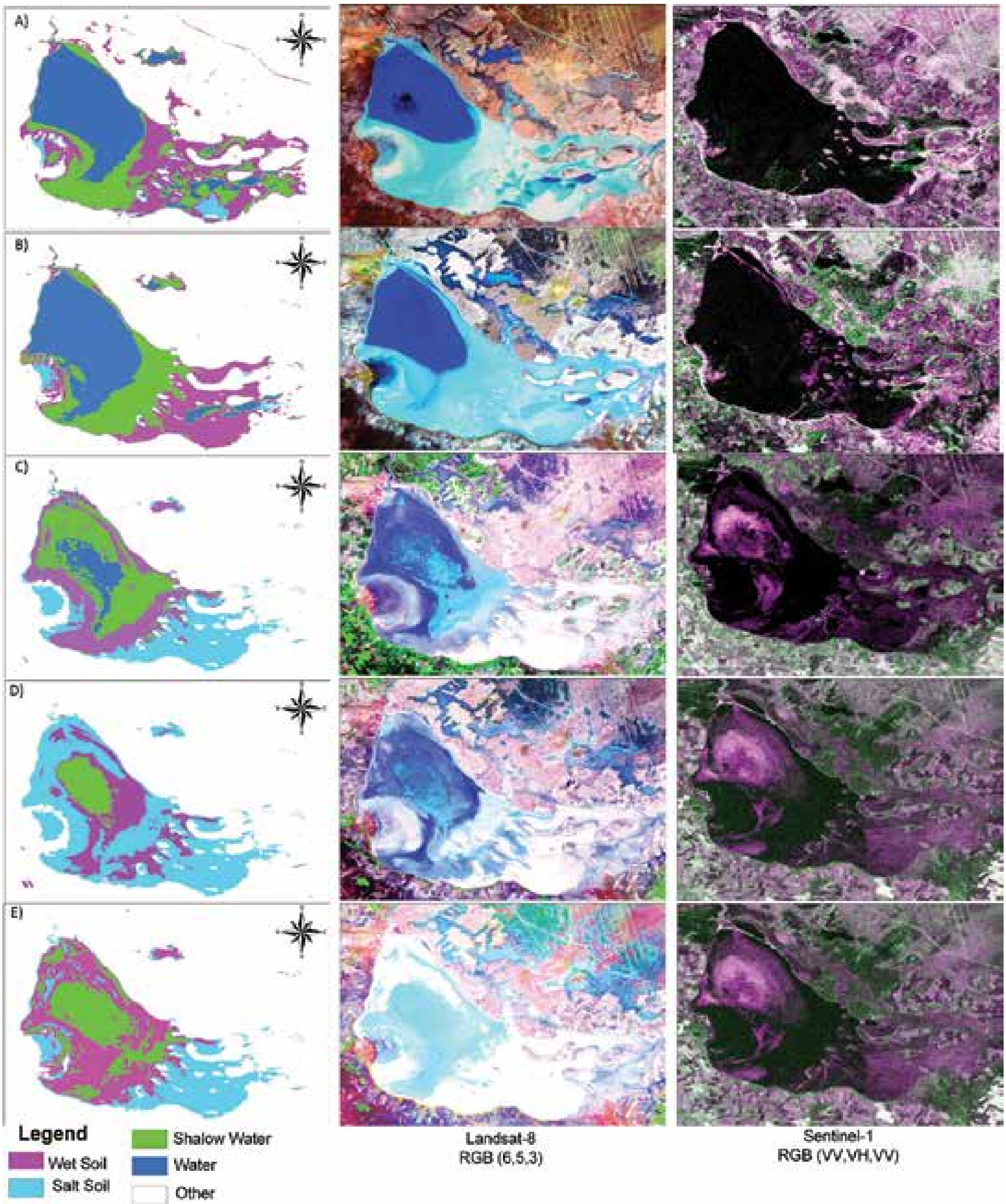

Figure 9. Seasonal wetland dynamic: (A) February, (B) March, (C) June, (D) August, and (E) October.

3.3. Remote sensing data statistics

Using the random point values that were added on the Seyfe Lake area, various statistical analyses were performed in order to find a correlation between the used remote sensing data.

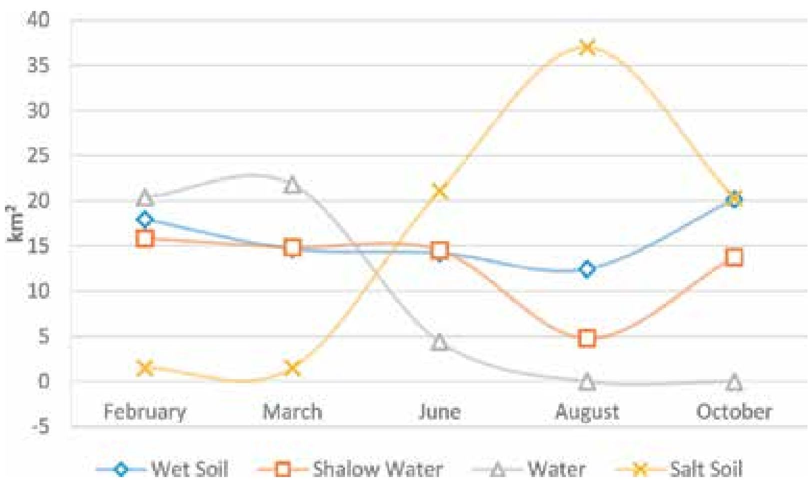

Figure 10. Seasonal Seyfe Lake dynamics.

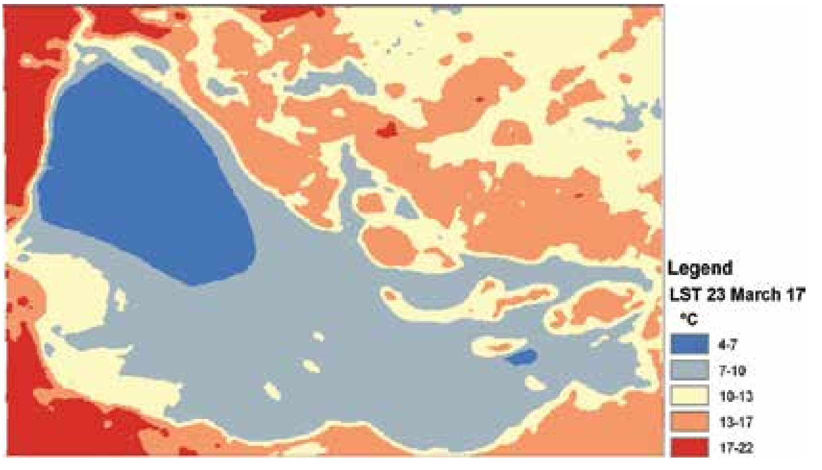

Figure 11. LST results from 23 March 2017.

Thus, the relation between VH, LST, and MNDWI and VV, LST, and MNDWI, between LST and MNDWI, and between VH and VV has been under detailed investigation (**Table 4**).

As seen in **Table 4**, the relation between the investigated parameters has been shown using three different statistical calculations; the multiple R is actually the correlation coefficient between the two variables, and it tells how strong the relationship between these variables is. Values close to 1 indicated a strong relation, while values close to 0 meant no relation at all. R^2, on the other hand, tells us how much of the change in the dependent variable can be explained by the independent variable. The R^2 value can be easily supported with significance variable. So when the significance variable is less than 0.05, the results are significant, or it means that the results did not occur by chance.

	Relation	VH/MNDWI	VH/LST	LST/MNDWI	VV/MNDWI	VV/LST	VV/VH
February	Multiple R	0.445	0.402	0.621	0.429	0.375	0.483
	R square	0.198	0.162	0.385	0.184	0.141	0.233
	Significance F	0.000	0.000	0.000	0.000	0.000	0.000
March	Multiple R	0.321	0.209	0.619	0.065	0.079	0.215
	R square	0.103	0.043	0.383	0.004	0.006	0.046
	Significance F	0.000	0.009	0.000	0.418	0.322	0.007
June	Multiple R	0.100	0.243	0.650	0.196	0.084	0.725
	R square	0.010	0.059	0.422	0.038	0.007	0.525
	Significance F	0.213	0.002	0.000	0.014	0.293	0.000
August	Multiple R	0.218	0.270	0.035	0.242	0.554	0.763
	R square	0.047	0.073	0.001	0.059	0.307	0.582
	Significance F	0.006	0.001	0.000	0.002	0.000	0.000
October	Multiple R	0.311	0.220	0.009	0.429	0.293	0.780
	R square	0.097	0.048	0.000	0.184	0.086	0.609
	Significance F	0.000	0.006	0.907	0.000	0.000	0.000

Table 4. Statistical results.

Since every season has different characteristics, the results were reviewed separately for each month and for each relation. February can be characterized with the lowest LST and high water level. The correlation coefficient between the investigated variables was highest in February for VH-MNDWI, VH-LST, and VV-MDWI. With the assumption that the LST is the independent variable, and the MNDWI is the dependent variable, the R^2 value shows that approximately 39% of the data or the rise of the temperature affects the water area. The results for March were not much different from the results from February with the only difference in the VV-MNDWI and VV-LST relation where the R values were significantly lowered. According to the statistics, the results from June have the most significant value in this study. As seen from the seasonal changes of the Seyfe Lake, in June the MNDWI values showed that the water area has slowly transformed into shallow water and wet soil. In this month the highest correlation coefficient of 0.61 was observed and also the highest R^2 of 0.422. Also, in June, the relation between VV and VH polarization was high with a correlation coefficient of 0.725 and R^2 of 0.52. While the other relation in August and October did not show significant results, the relation between the two polarizations, VV and VH, was noticeable with correlation coefficients higher than 0.76 and R^2 higher than 0.58 in both cases (**Figure 12**).

3.4. Summary

The initial investigation shows a strong relation between the parameters retrieved from the SAR, thermal, and optical satellite data and leads to better understanding of the wetland dynamics. Following the field measurements performed on 14 June 2018, a series of threshold analyses of NDWI and MNDWI were used in order to determine the optimal values for

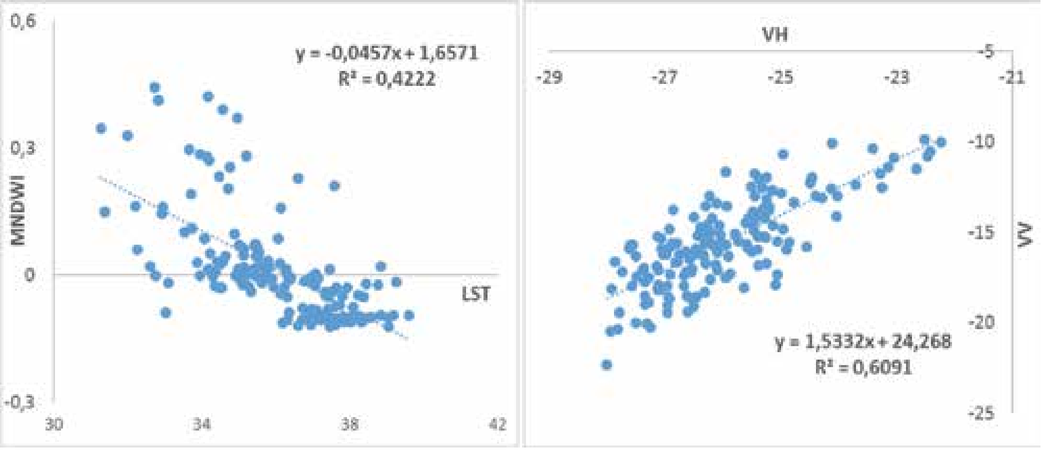

Figure 12. LST-MNDWI relation from June 2017 (left); VV-VH relation from October 2017 (right).

separating water bodies, shallow water, and wet soil from the other land cover classes. It has been concluded that MNDWI performs better in both water and wet soil extraction than NDWI. According to the MNDWI seasonal changes, Seyfe Lake is a very dynamic wetland, and its water area depends on the seasonal temperature. Thus, the water area is inversely proportional to the air temperature. So, for a temporal monitoring of the Lake, in this case, it is recommended to use annual data from the same season/month.

Wetlands ensure critical habitat for wildlife such as migrating water birds. Information gathered from monitoring wetlands may help land managers about the quality of wetlands. Remote sensing data has been the most useful tool to achieve spatial and temporal information about wetlands.

4. Discussion and conclusion

Wetlands are some of the most important ecosystems on Earth if wetlands' function is managed properly which provides tremendous fish and wildlife habitat but also improves groundwater quality and natural floodwater control. Global climate change and anthropogenic impact degrade wetlands, which creates a serious problem in identifying and quantifying wetland areas. Consequently, it is critical to be able to assess the status and quality of our remaining wetlands.

Remote sensing is a major source of spatial information about the land cover. Although a number of studies have investigated the relation between SAR and optical sensors in different land classes, the relation between backscatter values, NDWI, and LST values in wetland classes has not been a subject of a delicate investigation. In this chapter we investigate the relation between several remote sensing parameters, from a different aspect, trying to find a significant relation that will explain the wetland dynamics for better mapping, monitoring, and managing of wetland areas. The results in this study show a strong relation between the investigated variables. However, while some variables have strong relation when the water area is high, others have stronger relation when the lake is completely dried. The results of the statistical analyses from the month of June were taken to be highly significant from two points of view; the correlation between LST and MNDWI was more than 0.65, while the R^2 was more than 0.42. Taking in consideration the relation between the average temperature from each month and the area of each class, it can be concluded that with a correlation of −0.75, the water area depends on the temperature, or as the temperature increases, the water area decreases.

As stated in previous studies, with careful adjustment of the NDWI threshold, more accurate results could be achieved [25, 36]. Thus, it can be concluded that the values close to zero in the MNDWI threshold can give valuable information about the soil moisture. The positive values close to zero (0.1) indicate the shallow water areas, while the negative values close to zero (−0.1) indicate wet soil.

Even though in more of the cases the relation between the Landsat 8 and the Sentinel-1 parameters was high, in some cases their correlation was close to zero. Thus, while the changes of the VH values in March could be explained about 20% with the help of LST, the VV values could be explained about 20% with the help of MNDWI in October. The relation between the two polarization has highest values when the water level is at its lowest level because at that time the study area is mostly covered with dry soil. For a better understanding of wetland dynamics, we recommend using new techniques and different data fusion for exploring the full potential of remote sensing in wetland monitoring, supported with field measurements.

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Conflict of interest

The authors declare no conflict of interest.

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A Collection of Novel Algorithms for Wetland Classification with SAR and Optical Data

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Additional information is available at the end of the chapter

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Abstract

Wetlands are valuable natural resources that provide many benefits to the environment, and thus, mapping wetlands is crucially important. We have developed land cover and wetland classification algorithms that have general applicability to different geographical locations. We also want a high level of classification accuracy (i.e., more than 90%). Over that past 2 years, we have been developing an operational wetland classification approach aimed at a Newfoundland/Labrador province-wide wetland inventory. We have developed and published several algorithms to classify wetlands using multi-source data (i.e., polarimetric SAR and multi-spectral optical imagery), object-based image analysis, and advanced machine-learning tools. The algorithms have been tested and verified on many large pilot sites across the province and provided overall and class-based accuracies of about 90%. The developed methods have general applicability to other Canadian provinces (with field validation data) allowing the creation of a nation-wide wetland inventory system.

Keywords: canadian wetlands, remote sensing, SAR, optical imagery, wetland inventory

1. Introduction

1.1. What are wetlands?

Wetlands are among the most productive and biodiverse ecosystems in the world, covering an estimated 5–10% of the total global land surface [1]. For comparison, forests (the most dominant terrestrial ecosystem) make up an estimated 30% of the total global land surface [2, 3]. Though the term wetland has various definitions depending on the country of origin or application, most definitions share three common characteristics: the presence of water at or near the surface, the presence of unique soil conditions, and the presence of vegetation adapted to the wet conditions [4, 5]. Despite these commonalities, wetlands manifest in a variety of forms that have resulted in the production of numerous classification systems [6–8].

Wetlands form as a result of complex interactions among climatological, geological, geographical, geomorphological, chemical, floral, and faunal components of the environment [5, 9]. Variations within each of these environmental components and the way in which these components interact can produce wetlands that, while sharing similarities in the sense that they have a water table near the surface or vegetation adapted to wetland conditions [5], appear to be vastly different. The umbrella of wetlands includes ecosystems such as flooded forests with tall trees, sprawling tree-less bogs, rice paddies [10], and even transitory pools of water present only during the rainy season [11]. Certainly, what is, and is not, considered a wetland depends on governing body, location, and area of study [7]. In Canada, one wide-spread classification system describing these variable ecosystems is the Canadian Wetland Classification System (CWCS) [8]. See **Table 1** and **Figure 1** for examples of wetland classes described by the CWCS.

Although a popular topic today, the biology and beneficial services provided by wetlands were historically not well understood, and in the face of growing global populations and increasing urban and industrial sprawl, wetlands have been extensively lost and damaged [12, 13]. Currently, it is estimated that between 54 and 70% of the world's wetlands have been destroyed of damaged [1, 13]. Threats to wetlands today include not only land-use conversion but also complex global phenomena such as climate change [14]. This loss in turn has resulted in a decrease in the quality and quantity of locally and globally important ecosystem services that are often difficult to replace [15].

1.2. Wetlands functions and services

In recent times, there has been increased interest in wetlands due to both the historic and present rates of loss and a better understanding of the benefits wetlands provide to humans, other animals, and plants. These benefits, generally referred to ecosystem values or services, are the result of the natural functional processes that wetlands carry out through interactions

Wetland class	Wetland description
Bog	Peatland dominated by *Sphagnum* moss species and ericaceous shrubs, receiving water only from atmospheric sources.
Fen	Peatland dominated by graminoids (sedges and grasses and brown mosses, receiving water from multiple (precipitation, ground, surface) sources.
Swamp	Peatland or mineral wetland dominated by woody vegetation, potentially with standing water during certain times of the year.
Marsh	Mineral wetland dominated by hydrophytic emergent vegetation such as emergent graminoids and forbs, with standing or moving water.
Shallow water	Mineral wetland dominated by submerged or floating vegetation, with standing water up to two meters deep.

Table 1. Canadian wetland classes [8].

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Figure 1. Wetland classes in Newfoundland and Labrador. From top left to bottom: bog, fen, swamp, marsh, and shallow water.

and feedback among their geographical, morphological, chemical, floral, and faunal components [16]. These functions are the natural processes wetlands conduct outside of the context of humans, and services are the benefits humans derive from wetlands, upon which monetary or well-being value may be derived [16]. Functions can include, for example, water storage or nutrient cycling, while associated services include flood protection, reduction of downstream nutrient loading respectively [16, 17]. Wetlands of different types [6, 8] carry out different functions and different rates, and thus, different types of wetlands provide different kinds of services of variable quality (see **Table 2**).

The types of services wetlands provide can range from recreational to natural disaster mitigation [16, 18]. For example, wetlands of many types support biodiversity at rates disproportionate to their area [1] and provide habitat for numerous unique or threated species [19, 20]. At regional and local scales, wetlands play roles in flood risk reduction, drought mitigation,

Wetland class	Services
Bog	Source of nutrients and organic carbon, water storage, groundwater recharge, carbon storage, fuel and fiber source, plant and animal habitat.
Fen	Flood regulation, climate regulation, water filtration, source of nutrients and organic carbon, carbon storage, plant and animal habitat.
Swamp	Flood regulation, erosion protection, climate regulation, water filtration, carbon storage, plant and animal habitat, recreation.
Marsh	Flood regulation, erosion protection, ground water recharge, climate regulation, water filtration, carbon storage, plant and animal habitat, recreation (fowl hunting).
Shallow water	Flood regulation, erosion protection, water filtration, plant and animal habitat, recreation (fishing).

Table 2. Services associated with the five Canadian wetlands [16, 17].

shoreline protection, nutrient cycling, pollutant and sediment filtering, and recreational activities such as berry picking or fowl hunting [3, 16, 21–23]. In some parts of the world, local economies rely heavily on wetlands in the form of fishing, agriculture, and peat-harvesting [24–26]. Numerous studies have shown the direct effects of wetland loss on humans both in terms of monetary and quality of life [27, 28]. At a global scale, wetlands play important roles in biogeochemical cycles and are of importance in considering the effects and mitigation of a changing climate [14, 28–30].

1.3. Canadian wetlands

In Canada, the national estimate of wetland extent states that there is ~150 million hectares (1.5 million km^2) of wetlands, making up roughly one-fourth of global wetlands [8, 20]. Based on estimates of land-area, Manitoba, Ontario, Newfoundland and Labrador (NL), and Saskatchewan have the greatest extents of wetlands, the majority of which are composed of peatlands [4]. It has been estimated that up to 70% of Canada's non-peat wetlands have been lost [8]. The loss of Canadian wetlands has been documented as far back as the seventeenth century, during which around 85% of the salt marshes in the Bay of Fundy were drained by Acadian settlers [31]. Although wetlands and the services they provided were generally poorly understood, the impact of their loss was felt by communities reliant on those services. The Mi'kmaq, for example, noted the decreased presence of ducks and geese in and around the Bay of Fundy during the time of Acadian drainage [31]. More recently, flooding in provinces like Manitoba has been partially attributed to wetland loss [32]. Despite such extensive loss, Canada continues to rank as one of the countries containing the greatest extent of wetlands [33], making up 24% of the total global wetlands [8].

The province of NL has an estimated 18% of its land area covered by wetlands, 17% of which is peatlands [4]. The dominance of peatlands (bogs, fens, and swamps) in NL is expected, given both the oceanic climate [34] and deglaciation roughly 10,000 years ago [35] that created landscape features such as depressions and ponds that are ideal for peatland development via terrestrialization (i.e., the process of vegetation occupying the saturated land adjacent to the lake encroaching further into the lake while depositing and building litter resulting in, over time, the filling of the lake) [36, 37]. Additionally, extensive areas of poorly drained soils and acidic and nutrient poor seepage waters, a result of the type of dominant bedrock, contribute to broad peatland coverage in the eastern portion of the island [34]. Other wetlands, such as marsh and shallow water, are comparatively less prominent both in size and number. NL has yet to conduct a province-wide inventory and, until recently, was the only Atlantic Canadian province that had not yet initiated one [26, 38]. Recently, a project conducted between 2015 and 2017 began the process of inventorying wetlands in the province through the development of a remote sensing-based methodology to inventory wetlands down to five classes (bog, fen, swamp, marsh, and shallow water [26, 38].

Effective management and protection of not only wetlands in NL but also wetlands around the word requires the development and application of numerous methodologies, including but not limited to inventories and maps, water level and vegetation monitoring, and condition assessment. Historically, these methods would require extensive, costly, and time-consuming *in situ* field work campaigns, and unfortunately, given the expansive nature or wetlands and the rate at which these ecosystems are being lost, *in situ* methods are infeasible. This is not only due to the cost and time budgets but also because most wetlands are located in remote areas that make field visits difficult or impossible [39]. These problems can be effectively addressed through applying remote sensing methods, and a suite of such applications can be seen in various researches being conducted currently in NL, including the use of synthetic aperture radar (SAR) and optical imagery for wetland classification and mapping [26, 38, 40, 41] and wetland water level monitoring [42].

1.4. Remote sensing of wetlands

Given the current need for up-to-date wetland inventories, as well as the widespread coverage of wetland, remote sensing (RS) has been demonstrated to be the most efficient and cost-effective method for wetland mapping, classification, and monitoring [19]. Since 2016, we have been working on developing state-of-the-art algorithms using remote sensing technologies for operational wetland classification. For more information on our ongoing wetland work, please refer to (www.nlwetlands.ca). The following sections present a summary of our developed methods, discussed in more detail in our journal publications. For a list of these publications, please see Conclusion.

2. Wetland classification using SAR data

SAR is an active imaging system, capable of recoding the electromagnetic spectrum at much longer wavelengths compared to optical sensors. Unlike optical sensors, which collect ground target information at the cellular and molecular level, SAR sensors are responsive to physical (e.g., water content and size) and structural (e.g., roughness) characteristics of ground targets [43]. Over the past two decades, synthetic aperture radar (SAR) sensors have provided valuable data for wetland vegetation mapping. In particular, they are of great use when the efficiency of optical sensors is hampered by cloud cover and day/night conditions. Furthermore, SAR signal penetration depth through vegetation and soil offers additional information unavailable from optical remote sensing data [44, 45]. This is of great importance for monitoring the flooding status of vegetation due to enhanced double bounce scattering effects. Notably, the primary characteristics of SAR signals, such as wavelength, polarization, and incidence angle, with regard to key specifications of the ground targets, such as dielectric constant, roughness, and structure, determine the amount of SAR backscattered energy detected by SAR sensors [43]. Despite these benefits, SAR images are affected by speckle noise that degrades the radiometric quality of image, imposing challenges for several subsequent SAR processing tasks [46, 47]. Fortunately, Mahdianpari et al. [41] demonstrated the effect of applying an efficient despeckling method on the accuracy of wetland classification.

2.1. SAR wavelength

SAR wavelength is another influential factor for wetland vegetation mapping. To date, most SAR satellites have operated in three microwave bands, including X-, C-, and L-bands with wavelength of 3.1, 5.6, and 23.6 cm, respectively. Each wavelength has its own advantages and disadvantages. The selection of an appropriate SAR wavelength depends on the wetland classes since the interaction of SAR wavelengths varies widely with different vegetation types depending on their size. For example, longer wavelengths (L-band) can pass through the vegetation canopy and detect water beneath the flooded trees and/or dense vegetation. Accordingly, several studies reported the superior capability of L-band relative to the shorter wavelengths (e.g., C- and X-band) for monitoring woody wetlands (e.g., swamp), since the incident SAR signal interacts with larger trunk and branch components [48, 49]. In particular, L-band holds great promise in discriminating between forested wetland (e.g., swamp) and dry forest [45, 50]. However, shorter wavelengths are preferred for monitoring herbaceous vegetation because SAR wavelength and vegetation canopies (e.g., leaf) are relatively the same size [51].

Observations from SEASAT L-band data were among the first applications of SAR data for mapping the flooding status of vegetation [52, 53]. Later studies confirmed the suitability of L-band observations for mapping inundation in forested wetlands using JERS-1 and ALOS PALSAR-1 [50, 54]. Following the successful launch of C-band satellites, such as ERS1/2 and RADARSAT-1, several studies have also examined the capacity of C-band observation for wetland mapping. Most of those early studies reported the superior capability of L-band for mapping forested wetlands relative to C-band [44, 55].

2.2. SAR polarization

Overall, the HH polarized signal has been the most efficient for monitoring the flooding status of vegetation, since it is more sensitive to double bounce scattering associated with tree trunks in swamp forest and stems in freshwater marshes [54, 56]. VV polarization can also be useful when plants have begun to grow in terms of height but have a less developed canopy [51]. This is because in the middle of the growing season the vertically oriented structure of vegetation enhances the attenuation of VV polarization signals and, as such, the radar signal cannot penetrate to the water surface below the vegetation [48]. Cross polarization observation (HV and VH) has also been characterized as being highly sensitive to differences in biomass [57].

2.3. Wetland mapping using PolSAR data

Although single-polarized SAR data have been less useful for wetland classification, they have demonstrated great promise for monitoring open water surfaces in different applications, such as water body extraction and flood mapping [57]. This is because of the side-looking data acquisition geometry of SAR sensors. In particular, a large portion of the microwave signals transmitted to calm open water are scattered away from the SAR sensor, and therefore, open water appears dark in a SAR image, making it distinguishable from surrounding land [45]. Unlike single-polarized SAR data, polarimetric SAR (PolSAR) imagery was found to be extremely useful for wetland vegetation mapping. This is because a full polarimetric SAR sensor (e.g., RADARSAT-2) collects the full scattering matrix, providing comprehensive information about ground targets for each imaging pixel [58]. Furthermore, PolSAR data allow the employment of polarimetric decomposition techniques to identify the different backscattering mechanisms of the ground targets and, accordingly, regions of flooded vegetation [45, 49, 59]. Unlike coherent decompositions (e.g., Krogager decomposition), which are only useful for man-made structures with deterministic targets, incoherent decompositions determine the relative contributions from different scattering mechanisms. Thus, they may be more efficient for obtaining information from natural scatterers, such as wetland ecosystems [59–61]. Cloude-Pottier, Freeman-Durden, Yamaguchi, Van Zyl, and Touzi decompositions are among the well-known incoherent decomposition techniques useful for wetland mapping using PolSAR data [45, 49, 61, 62].

Despite the efficiency of the polarimetric decomposition technique to characterize different scattering mechanisms of ground targets that correspond to different wetland classes, the accuracy of wetland classification could be improved. This is attributed to both the highly dynamic nature of wetland ecosystems and the similarity of different wetland classes. The former of which can be alleviated by using multi-temporal SAR data to accurately characterize wetland dynamics during growing seasons [41, 51, 61, 62]. Furthermore, some studies employed a large number of input features to tackle the problem of similarity between different wetland classes [63]. Despite the promising results obtained from such an approach to date, it may not necessarily be optimal approach due to both computational complexity and redundant information within a large number of input data. Furthermore, some wetland classes can be easily distinguished using a minimal of input features. For example, the shallow water class can be easily separated using a SAR backscattering analysis and employing a threshold. However, this similarity is more pronounced among herbaceous wetlands, indicating the necessity of incorporating a larger number of input data [45]. As such, a hierarchical classification scheme can be useful to optimize the number of input features according to the similarity of wetland classes, which should be distinguished at each classification level. Some recent studies also noted that the discrimination of wetland classes can be further increased by applying a feature weighting approach using the Fisher Linear Discriminant Analysis technique [61, 64]. Such an efficient approach eliminates the necessity for the inclusion of large number of input data.

2.4. Wetland mapping using compact polarimetry data

The information content within SAR data increases given the polarization hierarchy, starting from single polarization to dual polarization and reaching both compact and full polarimetric data [65]. Specifically, fully polarimetric data are of great importance for land cover and, in particular, wetland mapping. Such a SAR sensor is constructed based on the standard linear basis (i.e., horizontal [H] and vertical [V]), wherein the sensor interleaves pulse with H and V polarization toward the ground targets and record both received polarizations simultaneously and coherently [65]. As such, the first disadvantage of full polarimetric SAR sensors is a time constraint because two orthogonal polarizations are transmitted alternately. Furthermore, such a configuration implies complexity due to doubled pulse repetition frequency, as well as an increase in the data rate by a factor of four relative to a single-polarized SAR system [65]. Accordingly, the image swath width of FP SAR images is halved, resulting in reduced coverage and an increase in satellite revisit time [66]. Finally, this configuration allows a limited range of incidence angles compared to that of single/dual polarization modes [67].

An attractive alternative, which addresses the limitations of full polarimetric SAR sensors, is a compact polarimetry (CP) SAR configuration. The CP SAR image is expected to maintain polarimetric information as close as possible to that of full polarimetric SAR mode imagery while alleviating its primary limitations [68]. In particular, CP sensors collect a greater amount of scattering information compared to single- and dual-polarization modes while covering twice the swath width of full polarization SAR systems [69]. Thus, CP SAR configurations decrease the complexity, cost, mass, and data rate of a SAR system while preserving several advantages of a full polarimetric SAR system [70]. m-delta [71], m-chi [72], and m-alpha [73] are common decomposition techniques of compact polarimetry data. Importantly, the upcoming RADARSAT Constellation Mission (RCM), which will operate in the Circular Transmitting Linear Receiving (CTLR) mode, offers improved operational capabilities (e.g., ecosystem monitoring) along with a much shorter satellite revisit period. Specifically, RCM provides daily coverage over Canada with 350-km imaging swaths [74]. This is of great significance for highly dynamic phenomenon such as wetland complexes. Some recent studies reported the efficiency of simulated compact polarimetric data for wetland mapping [68, 75].

2.5. Wetland monitoring using InSAR

Hydrological monitoring of wetlands is another subject of interest, since they are water-dependent ecosystems. SAR images have shown to be useful for wetland hydrological monitoring using both SAR backscattering responses [76] and a more detailed and sophisticated technique, Interferometric SAR (InSAR) [77]. This is because the flooded and non-flooded statuses of vegetation in wetland environments have distinct differences in radar backscattering responses that play an important role in the hydrological monitoring of wetlands. Specifically, a time series analysis of SAR backscatter signatures has offered information of seasonal patterns of flooding in wetland ecosystems, and the enhanced SAR backscatter signature of flooded vegetation has been examined in a number of studies [76, 78–81].

Although several studies reported the potential of InSAR for wetland water level monitoring, its application in wetlands presents challenges. This is primarily due to the substantial altering of reflectance and energy backscatter of wetland environments, even within hours or days [82], and the low backscatter of the water surface. Under these conditions, interferometric coherence,

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which quantifies the degree of similarity of the same pixel in the time interval between two SAR acquisitions, cannot be maintained [51].

Interferometric coherence is a quality indicator of InSAR observations. The variation of coherence in wetlands is a function of the complex mixture of several factors that contribute to coherence maintenance. The temporal baseline is one of the main parameters that hampers the application of InSAR for wetland monitoring [83]. Herbaceous vegetation, one of the most substantive components of wetland ecosystems, may easily lose coherence within a day or week. In the case of using shorter wavelengths (e.g., C- and X-band), interferometric coherence may be lost due to the shallow penetration depths of the shorter wavelengths. In contrast, longer wavelengths have deeper penetration depth but have been previously associated with longer temporal baselines (46 and 44 days for ALOS PALSAR-1 and JERS-1, respectively), which could cause a loss of coherence. However, this drawback has been addressed in the currently operating L-band SAR sensor (i.e., ALOS-2), wherein the temporal baseline is 14 days. Thus, ALOS-2 repeat-pass SAR images offer a promising source of data for wetland InSAR applications. Geometric decorrelation caused by different satellite look angles, volumetric decorrelation caused by vegetation volume scattering [83, 84], the Doppler centroid effect, and co-registration error during interferometric processing [65, 85] are other sources of decorrelation over wetlands.

Despite these limitations, several studies reported the feasibility of InSAR for wetland water level monitoring. In particular, when the vegetation within or adjacent to standing water is able to backscatter the radar pulse toward satellite sensor, water level changes are observable from the phase data [86, 87]. Also, vegetation should not be too dense for the penetration of microwave energy [65]. The efficiency of the InSAR technique for wetland monitoring has been initially investigated in the Amazon floodplain [77]. Subsequent investigations have been carried out for a number of other wetland sites such as Florida Everglades [49, 77, 87, 88], the Louisiana Coastal wetland [56, 89], and China wetlands [89, 90].

In addition to hydrological monitoring of wetlands using InSAR, the interferometric coherence can be used for other wetland applications, such as change detection and classification [51, 91]. This is because coherence has a diagnostic function and can be used along with SAR backscatter and polarimetric decomposition techniques for classification of different wetlands. Each feature has specific characteristics and, accordingly, plays a different role for discriminating wetland classes. For example, SAR intensity depends on the electromagnetic structure of the targets, while the interferometric coherence reflects their mechanical and dielectric stability. Thus, an integration of different input feature augments land cover information and improves classification accuracy of wetland types [51].

3. Spectral and backscattering analyses of wetlands using multi-source optical and SAR data

Wetlands are complex landscapes and ecologically share similar characteristics. However, each wetland type contains its own specifications, which can be effectively investigated using various satellite imageries. In this regard, both optical and SAR data are the most common remote sensing data, which have so far proved to be significantly helpful in discriminating wetland species. Numerous types of features can be extracted from multi-source optical and SAR data. However, since all the extracted features cannot be inserted into a classification algorithm, the most important features should be selected for classification. As such, the best optical and SAR satellites, spectral bands, spectral indices, SAR features, SAR channels, back-scattering mechanisms, decomposition methods, and textural features can be defined for wetland studies. To this end, various separability measures have already been developed and employed for differentiating wetland classes.

Before separability analysis, several pre-processing steps should be performed on the datasets, the most important being variance analysis of field samples. This should be carried out on both individual classes and class pairs. For this, Eqs. (1) and (2) can be used, respectively.

$$Var = \frac{1}{N-1}\sum_{i=1}^{N}\left(x_i - \mu\right)^2 \tag{1}$$

$$F = \frac{Var_B}{Var_W} \tag{2}$$

in which, x_i indicates the value of a field sample; μ is the mean value of samples; N is the number of field samples in a feature; F indicates the Fisher-test; and Var_B and Var_W indicate the between and within variance values in each class pair, respectively. These two variance analyses are more important in the case of wetlands because they are complex environments, and thus, the field samples collected for a wetland class can contain high variance in satellite imagery, especially those acquired by the SAR systems. **Figure 2** illustrates an optical spectral band and a SAR feature, for which the variations of field samples are high, and consequently, they should be removed before separability analyses as noisy and poor features.

So far, different separability measures have been developed, which can generally be classified into two categories: parametric and non-parametric. Unlike parametric methods (e.g. t-test), non-parametric techniques, such as Mann-Whitney U-test, do not assume a normal distribution of the samples and evaluate the separability of samples by their ranks [92]. Considering

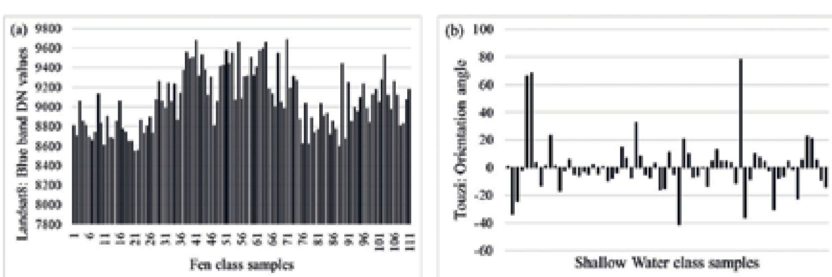

Figure 2. Spectral and Backscattering values for field samples for two types of wetlands: (a) Fen, and (b) Shallow water.

the high variance of field samples of wetlands, the recommendation is to employ a non-parametric distance. After removing the poor features using variance analyses and obtaining the separability measures that each feature provides, the most effective features are inserted into a classification algorithm to produce a highly accurate wetland map.

Table 3 summarizes the results of separability analyses performed by U-test on five wetland classes (bog, fen, marsh, swamp, and shallow water) using multi-source optical (RapidEye,

	Bog	Fen	Marsh	Swamp	Shallow water
Bog	×	CP: alpha Tz: alpha_s FD: double-bounce CP: entropy S1: HH/HV	Tz: alpha_s CP: alpha FD: volume-scattering CP: anisotropy R2: HV/TP	R2: HH/TP R2: HH/HV Anisotropy12 A2: HH/HV Polarization-Asymmetry	CP: anisotropy N_derd R2: HH/VV FD: volume-scattering N_serd
Fen	A: $\frac{Green}{Brightness}$ A: NDWI R: $\frac{Red\ Edge}{Brightness}$ L8: $\frac{NIR}{Brightness}$ S2: $\frac{Red\ Edge}{Brightness}$	×	S1: HH/HV N_derd CP: anisotropy R2: HH/HV R2: HH/TP	A2: HH/HV serd R2: HH/HV R2: HH/TP N_serd	N_derd CP: anisotropy R2: HH/VV R2: HH/HV N_serd
Marsh	A: $\frac{Green}{Brightness}$ A: NDWI S2: $\frac{Red\ Edge}{Brightness}$ S2: NDWI R: $\frac{NIR}{Brightness}$	A: $\frac{NIR}{Brightness}$ R: $\frac{Green}{Brightness}$ S2: NDWI A: NDVI A: SAVI	×	R2: HH/HV R2: HV/TP R2: HH/TP A2: HV serd	CP: anisotropy S1: VV/HV N_derd R2: VV/TP R2: VV/HV
Swamp	S2: $\frac{Red\ Edge}{Brightness}$ L8: $\frac{NIR}{Brightness}$ L8: $\frac{Green}{Brightness}$ S2: NDVI S2: SAVI	L8: NDVI L8: SAVI L8: $\frac{NIR}{Brightness}$ A: $\frac{Red}{Brightness}$ A: NDWI	L8: NDVI L8: SAVI L8: $\frac{NIR}{Brightness}$ A: $\frac{Red}{Brightness}$ S2: NDVI	×	R2: HH/HV N_derd CP: anisotropy serd N_serd
Shallow water	A: $\frac{Green}{Brightness}$ A: NDWI R: $\frac{Red\ Edge}{Brightness}$ R: $\frac{Green}{Brightness}$ R: NDWI	R: $\frac{Red\ Edge}{Brightness}$ R: NDWI R: $\frac{NIR}{Brightness}$ S2: NDWI S2: $\frac{Red\ Edge}{Brightness}$	R: $\frac{Red\ Edge}{Brightness}$ R: NDWI R: $\frac{NIR}{Brightness}$ S2: $\frac{Red\ Edge}{Brightness}$ S2: NDWI	R: $\frac{NIR}{Brightness}$ R: NDWI R: $\frac{Green}{Brightness}$ R: $\frac{Red}{Brightness}$ A: $\frac{Green}{Brightness}$	×
L8: Landsat-8 S2: Sentinel-2A S1: Sentinel-1 Tz: Touzi SAVI: soil adjusted vegetation index N_derd: normalized double-bounce eigenvalues relative difference		R: RapidEye R2: RADARSAT-2 CP: Cloude-Pottier NIR: near infrared NDWI: normalized difference water index serd: single-bounce eigenvalues relative difference		A: ASTER A2: ALOS-2 FD: Freeman-Durden NDVI: normalized difference vegetation index TP: total power N_serd: normalized single-bounce eigenvalues relative difference	

Table 3. The most important optical (provided in the lower left half of the table) and SAR (provided in the upper right half of the table) features for delineating each pair of wetland class in June and August, respectively (the features are ordered based on their separability measures).

Landsat-8, Sentinel-2A, and ASTER) and SAR (Sentinel-1, RADARSAT-2, and ALOS-2) data in NL, Canada. As is clear from this table, the ratio features provided the highest separability measures. $\frac{NIR}{Brightness}$ and $\frac{Red\ Edge}{Brightness}$ ratios are most efficient regarding the optical data, and the ratios of HH/HV and HH/TP obtained from RADARSAT-2 full-polarimetric data are the most important SAR features for separating wetlands.

Comparing the optical spectral bands, the NIR and Red Edge bands are most effective for discriminating wetland classes. Two main characteristics of wetlands are vegetation and water, which can be efficiently studied by these two bands. This demonstrates that it is more efficient to use the optical satellites, in which both NIR and Red Edge bands are included (e.g. Sentinel-2A and RapidEye). In this regard, Sentinel-2A, which provides free imagery, is superior for employment in operational wetland mapping and monitoring. The red band is also helpful in separating wetlands, especially discrimination between bog and other wetlands, because of bogs' red appearance. Additionally, there is a high overlap between the spectral signatures of wetlands in the green, SWIR, and TIR bands, and thus, there is a difficulty in using these bands for wetland studies. Finally, the blue band is not very useful in most of the cases.

Comparing various decomposition methods, including Freeman-Durden, Cloude-Pottier, Touzi, Van Zyl, Yamaguchi, and Krogager, it is observed that coherent decomposition techniques, such Krogager, are not recommended for wetland classification. The reason is that the coherent decompositions are mostly applicable for detecting man-made features in urban areas and less useful for naturally distributed targets such as wetland classes [93]. In addition, the Cloude-Pottier and Freeman-Durden methods are most optimum for separating wetland species. In this regard, the volume scattering component of Freeman-Durden and Anisotropy element of Cloude-Pottier are generally the best. Moreover, some SAR features extracted from the eigenvalue/eigenvector of the coherency matrix demonstrated a high potential for separating wetland class pairs and all wetland classes. In this regard, the serd, normalized serd and normalized derd, introduced by [94], are frequently selected for wetlands separation.

4. A multiple classifier system to improve classification accuracy of wetlands using SAR data

So far, numerous classification algorithms have been developed to classify various land covers, each containing its own advantages and limitations. Random Forest algorithm has proved its high potential for wetland classification in many studies (e.g. [40, 26, 61]). However, the most promising approach to obtain a high classification accuracy is fusing different classifiers in a way that the advantages of each are ensembled. The obtained ensemble classifier is called multiple classifier system (MCS [38, 95]). The system is more important when classifying complex landscapes, such as wetlands, because achieving high accuracy for individual classes is significantly challenging in these cases. This becomes even more serious when only SAR data are applied for discriminating wetlands. There are several studies which developed new MCSs to improve the classification accuracy of similar

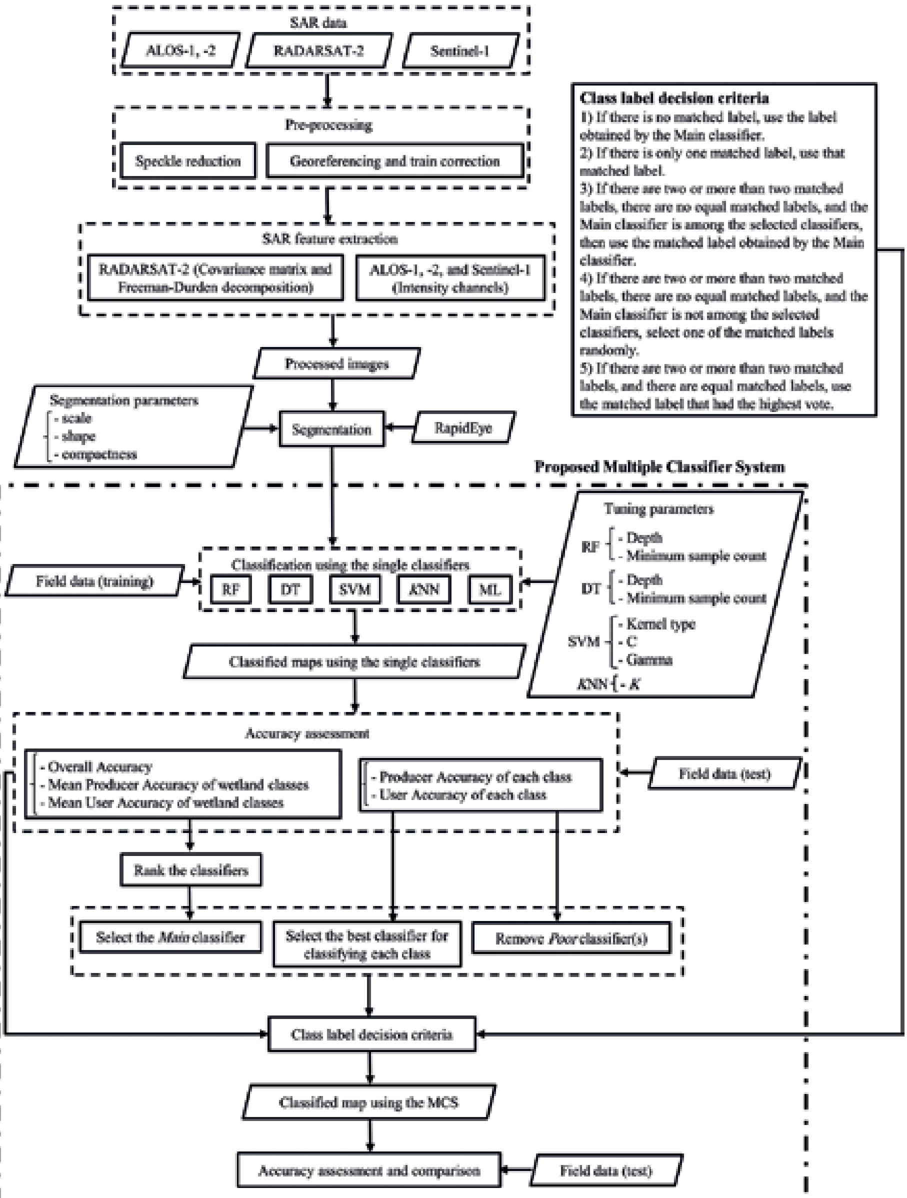

Figure 3. Proposed multiple classifier system by Amani et al. [38] to improve the classification accuracy of the complex environments.

landscapes (e.g. [96, 97]). Regarding wetland classification, Amani et al. [38] proposed a novel MCS to increase wetland classification accuracy using only SAR data in NL, Canada, in terms of both individual class and overall accuracies. The system initially removes poor classifiers and selects the best classification algorithm to identify each wetland class. Then, the final label is selected for each random pixel/object using the class label decision criteria introduced by the authors. The flowchart of the proposed MCS along with the corresponding criteria is illustrated in **Figure 3**. The proposed MCS outperformed the single classifiers and produced the highest producer and user accuracies for almost all wetland and non-wetland classes. It also increased the overall classification accuracy and kappa coefficient by 5–8 and 9–16%, respectively.

5. Conclusion

Wetlands are productive and diverse ecosystems providing numerous ecological services that are biologically important as well as playing a key role in surface water hydrology and flood risk. Wetlands are and have been threatened by land-use conversion, increased urbanization, industrial development, and climate change, resulting in more than half of the world's wetlands threatened, damaged, or destroyed. Earth observation provides a new cost-effective approach to mapping wetlands to aid in their management especially in remote and difficult to access regions. A combination of optical and SAR data provides adequate input data to use an object-based classification with machine learning algorithms such as Random Forest resulting in classification accuracies exceeding 90% for study sites in Newfoundland/Labrador.

For more details on some of the information discussed in this chapter, please refer to our published papers [3, 26, 38, 40–42, 45–48, 50, 51, 61, 64, 68, 69, 98–100]. While [42] is a literature review paper on the use of interferometric synthetic aperture radar (InSAR) data for water level monitoring of wetlands, the rest mainly introduces new machine learning methods for wetland classification using optical, SAR data, or the combination of both.

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Conflict of interest

The authors declare no conflict of interest.

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Section 4

Human-made Structures

Combining the Aesthetic and Ecological Aspects of Man-Made Structures on Coastal Wetlands

Lee-Hsueh Lee

Additional information is available at the end of the chapter

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Abstract

Man-made structures are used as adaptive solutions to natural and urbanization stressors of coastal wetlands. These structures alter the wetland environment not only impacting ecological value and habitats but also landscape esthetics. A green structure approach aims to re-establish the natural functions of wetlands; however, landscape esthetics of the relationship between man-made structures is required that also should not be neglected. Physical structures are tangible and shape the visual environment, which can influence people's esthetic preference. Pleasing scenery can arouse protective instincts and motivate public participation in wetland conservation. Man-made structures changed and limited landscape room, resulting in homogeneous environmental information in the landscape foreground, while hindering collection of environmental information from the background. The discordance of contextual cues between coastal wetlands and man-made structure affects the esthetics and preference of landscape. Therefore, consideration of both landscape esthetics and the ecological impact of man-made structures is an optimal coastal wetland restoration strategy. Here, a conceptual common ground between the visual and ecological aspects of man-made structures is proposed. This concept is applied to design man-made structures that will benefit landscape esthetics and mitigate wetland ecological impacts.

Keywords: environmental information, landscape preference, landscape room, shifting baseline, target scenery, viewing place

1. Introduction

Coastal wetlands are located in the terrestrial-aquatic transverse zone and are an important landscape type and ecosystem. These wetlands have high biodiversity, serve as a buffer zone for adjacent upland development, and provide multiple services such as protection of water quality, and flood and erosion control. Furthermore, coastal wetlands provide visual diversity and unique visual character, which significantly influences the well-being of people and their emotional attachment to the environment. For various reasons, half of the world's wetlands have disappeared since 1990, and therefore, wetlands have become the most threatened landscape type. Anthropogenic activities can impact the coastal wetland environment in different ways. Furthermore, coastal marshes and swamps are vulnerable to climate change and sea level rise.

Land use, economic development demands, reclamation of land from the sea, and natural oceanographic processes can alter the coastal wetland environment. To manage these anthropogenic and natural factors, man-made structures have been applied to protect and maintain the intertidal zone. Man-made structures can affect the coastal wetland ecology by reducing coastal area, disrupting natural water flow, and threatening species survival. Furthermore, such structures hinder people close to water, change the visual perception of the landscape, decrease the esthetic value, and weaken the environmental attachment for local people.

Although support for wetland conservation is strong, wetlands are disappearing. Urban sprawl and increase in population density are primarily attributable for wetland loss. Environmental education, besides policies and legislations, is a commonly adopted strategy to encourage public participation in wetland protection activities. Widespread wetland degradation and loss may imply a generational knowledge gap about environmental issues.

Due to land development activities, man-made structures are playing an increasingly dominant role in shaping the coastal wetland environment [1]. People were surrounded by the scenery in their daily life. Both older and younger generations are affected by modified coastal wetlands environments frequently. An understanding of the healthy ecological conditions in the past is lacking, and therefore, environmental norms continue to change. Loss of an ecological baseline will bring about still unknown challenges for coastal wetland conservation [2, 3].

It was widely hypothesized that landscape esthetic is a stimulus–response relationship based on the interaction between humans and the environment. The human perception of the environment is immediate and is accompanied by short-term emotional pleasure, while ecological esthetic is a knowledge-based cognitive experience where long-lasting pleasure is obtained through understanding. It is debated whether people can directly sense ecological quality; however, based on evolutionary and cultural theory, good landscape esthetics is associated with high ecological quality. For survival, people choose suitable habitats and alter them to suit their needs, while there is a sense of enjoyment and desire to live among scenery perceived as beautiful. There are common physical environmental elements, which affect landscape, ecological function, and the composition of visual image, individually. For example, ecological functions affect the appearance of a landscape and people appreciate the appearance; therefore, good ecological health would be inherently included in the landscape esthetic.

Ecologists have worked toward improving the public's environmental protection awareness through environmental education. The willingness to protect a habitat will triple if the target species is beautiful or if its habitat is attractive [4]. Therefore, environmental perception and experience can influence conservation behavior [5, 6]. In addition to wetland protection through policy and legislation, conservation approaches should consider both the esthetic and ecological impacts and aim to promote public participation in protection activities.

Man-made structures are built in the coastal intertidal area and as a result dominate the landscape and ecology of many coastal wetlands. To overcome the negative ecological impacts of man-made structures, environmentally friendly structures had been applied to coastal wetlands. Man-made structure changed the scale and openness of landscape room, and the state of environmental information, thus had effects on esthetic value and preference. The drawback to this purely ecological approach is that landscape esthetics have not been deeper considered.

The known influence of landscape esthetics on public ecological protection action implies that esthetic consideration during the development and application of man-made structures in coastal wetlands is necessary. The initial objective of man-made structures was to protect coastline and human habitats; however, protection of sensitive and ecologically important coastal wetlands should also be considered due to the knock-on benefits for humans and the environment. Under the coastal defense and undamaged habitat objective, an approach that improves landscape esthetics and healthy ecological functioning through refinement of the visual landscape of man-made structures could be crucial for influencing public perception and conservation action.

2. Landscape esthetic and ecology of healthy coastal wetlands

2.1. Experience of landscape with ecology

Landscape esthetic is based on the idea that human preference for a particular physical landscape is rooted in biological or evolutionary adaption [7–9]. The habitat theory and the information-processing theory provide insight into why people may prefer certain landscape characteristics. These theories suggest that human interactions with the environment are related to various survival behaviors. An environment benefited to people that provided those with the capacity to observe without being seen meant both prospect- and refuge-dominant landscape settings were more preferred.

Humans require environmental information to understand their surroundings. According to the information processing theory [9, 10], human perception is oriented to understand and react to the environment. High coherence setting means that the setting is orderly, and legibility can be more preferred. An environmental setting with high species richness and diversity indicates complexity. The human environmental perception of this situation could be mysteriousness in which humans react by exploring their surroundings and discover valuable resources. Conversely, highly homogenous or too heterogeneous environmental setting could induce an uninterested or fearful environmental perception, provoking a reaction to escape. The human perception of environmental settings and information provides critical guidance for determining which habitats are suitable.

When people interact with environment, they have an esthetic experience and emotional response. Together, these reactions influence the choice of the landscape. Spatial and temporal changes of landscape can result from ecological functions. These landscape changes, stemming from various ecological functions, will influence the esthetic perception of a landscape through time.

Positive responses to characteristics of a setting generally increase chances of survival or well-being. On the other hand, esthetic is also shaped by cultural expectations [11–13] and contemporary environmental behaviors [13]. Esthetic experiences drive landscape change in the context of habitat, leisure, recreation activities, and daily life. For example, in an esthetically pleasing environment, people are more prone to enjoy, have a connection with, and protect it. In an esthetically unpleasing, ugly, or unsafe environment, people would avoid it or seek to improve it (**Figure 1**). Environment in which improvements are usually made tends to be those which people enjoy or are preferable for land use. The resulting changes may or may not benefit landscape esthetic and ecology. The esthetic experience provides a good linkage between the human benefits of landscapes and healthy ecological functions, which is based on the evolutionary theory that a healthy ecological setting is associated with landscape characteristics that are esthetically preferable.

The perceptual cues stemming from the interaction between humans and environment can be used to assess which settings evoke particular reactions (**Figure 2**). Both evolutionary and cultural drivers suggest that ecological health is associated with a pleasing esthetic landscape.

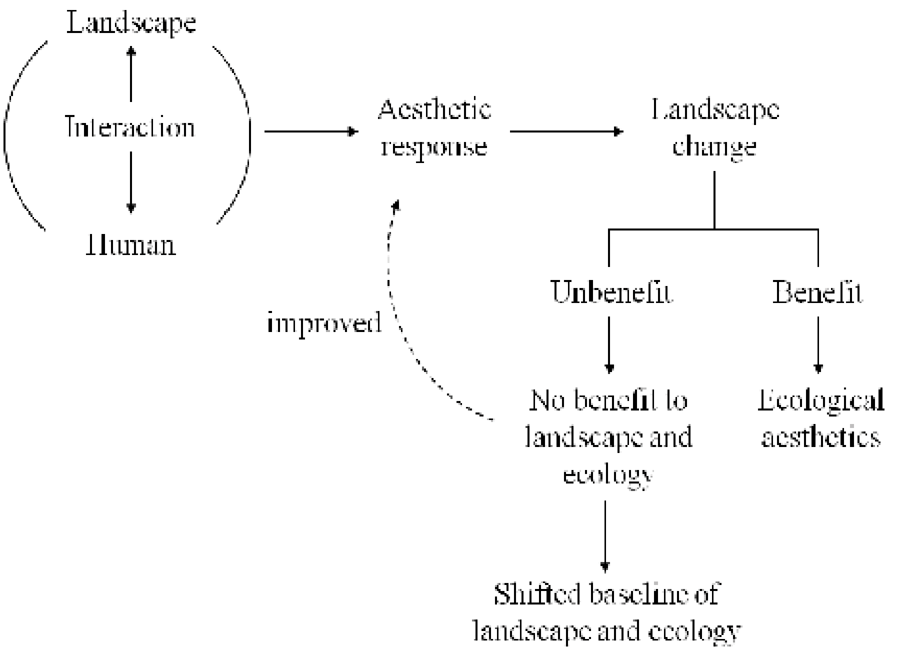

Figure 1. The process of landscape experience and resulting landscape change. The esthetic component was affected by the interaction between human and landscape. Therefore, landscape change may benefit both landscape and ecology (ecological esthetics). Otherwise, the existing situation will lead to generational knowledge gaps and shift the baselines of landscape and ecology. Therefore, integrating esthetic and ecological design approaches to improve unfriendly landscape and ecology coastal wetland is important.

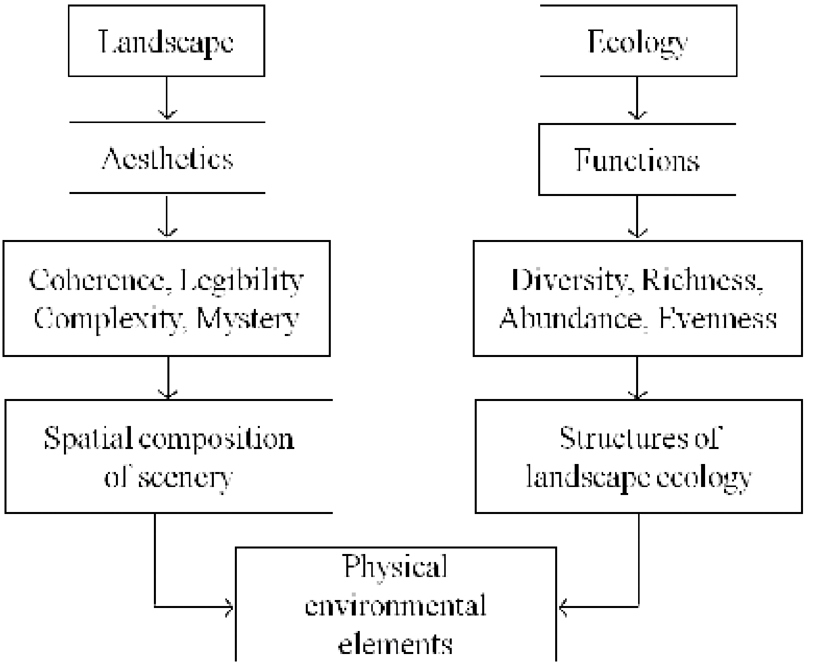

Figure 2. Landscape versus ecological experience. Landscape and ecology have common physical environmental elements, which include scenery and structure of landscape ecology. Four cognitive attributes of environmental information influence people's landscape esthetic preference. There are four ecological indicators of ecological health.

In this way, humans can sense environmental information, landscape esthetic, and ecological health. The arrangement of the physical characteristics of landscape significantly affects the perception of the landscape esthetic and thus ecological function. Environmental information culminates in four attributes to derive landscape preference. A coherent and orderly setting is easy to understand and as such enables people to feel legibility and secure; conversely, a complex setting made people fell mystery would encourage curiosity and stimulate exploration. A more detailed description of preferable landscape characteristics is shown in Kaplans' environmental preference matrix [9, 10, 14]. Ecologically, these characteristics correspond to species diversity, richness, evenness, and abundance, which taken together, constitute landscape ecology. Further aspects of landscape ecology include patch heterogeneity, disturbance, size, and edge structure and habitat naturalness and continuity [10]. Humans ascertain environmental information from the diversity and evenness of a patch, this setting is favorable as it is representative coherence and legibility and thus security. Natural landscapes are favored by human; the continuity of patches implies that plenty of environmental information is available in the middle to background, which could induce curiosity and exploration [15].

Humans are closely linked to the wetland environment and as such, human activity has altered the wetland landscape and ecological function. A wetland model of landscape esthetics versus ecological processes was created (**Figure 3**). At the opposite ends of the ecological processes, axes are ecological services and human activities, while on the landscape esthetics, axes are natural beauty and formal beauty. There were four principal types of wetlands included: (1) natural wetlands, (2) modified wetlands, (3) recreational wetlands, and (4) artificial wetlands, divided into the four quadrants in **Figure 3**.

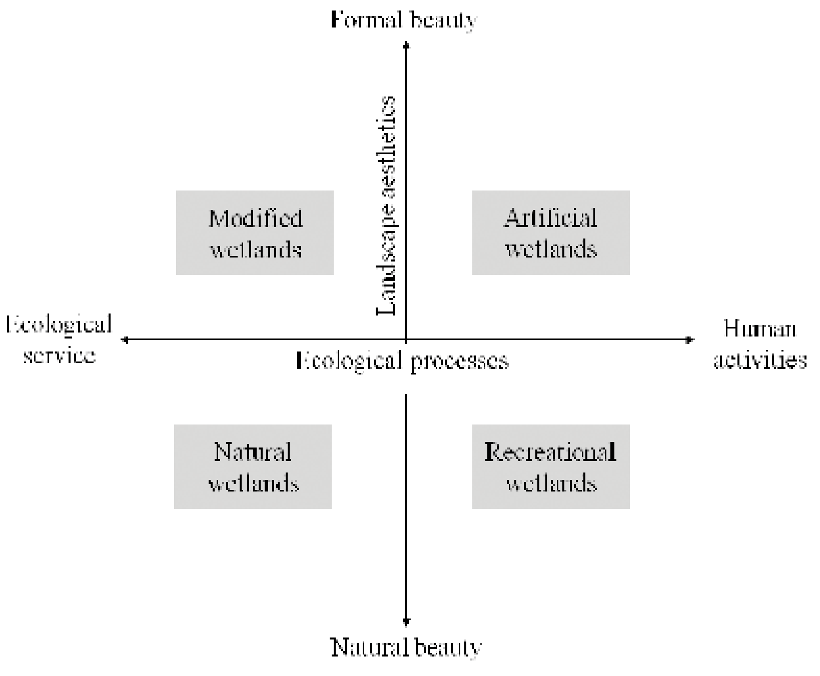

Figure 3. A matrix of the four types of wetlands in terms of landscape esthetics and ecological processes.

The natural wetland quadrant is united by ecological services and natural beauty. This indicates that natural interference of ecological process affected the ecological health of the wetland, and the natural process is the dominant pattern. Here, is more stability with fewer disturbances, and to affect ecosystem health coupled with landscape naturalness to deliver a high esthetic value.

Artificial wetlands are formed by human activities and constitute formal beauty. Artificial wetlands are related to in varying degrees of anthropogenic utilization, distributed from the center city to urban the fringe. Contrary to natural wetlands, human interventions, such as design-orientation, engineering, and maintained works, are practiced in here. Human activities highly limited ecological processes and functions but increase formal beauty.

Ecological services form modified wetlands, which are esthetically pleasing. These wetlands were modified to protect coastal from erosion or to meet land use demands. Ecological services decline as the number of man-made structures increases. Finally, recreational wetlands facilitate human activities and natural beauty. In recreational wetlands, recreational intensity directly disturbed to ecological quality, together with the naturalness of beauty.

In addition to the described two-dimensional framework, a Z-axis depicting design approaches is overlaid to form a three-dimensional model. The two ends of design approach are eco-oriented and engineering-oriented and would promote either natural beauty or formal beauty. This will impact the landscape esthetics, which, as previously described, influences ecological processes (**Figure 4**).

Natural beauty corresponds to an ecologically esthetic landscape, where the appearance of ecological function is visible, emphasizing the visual enjoyment of natural scenery. In

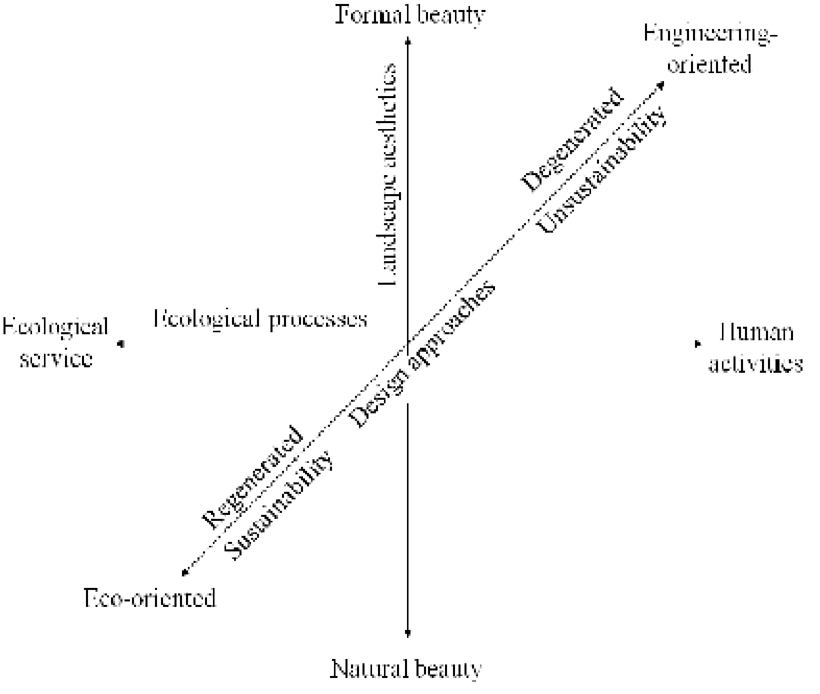

Figure 4. The three-dimensional model consisted of ecological processes, landscape esthetics and design approaches. Design orientation can lead to sustainable or unsustainable wetland development.

this scenario, ecological health and landscape esthetic are mutually reinforcing each other. Designs can be introduced to achieve a particular ecological esthetic preference. The impacts of human activities will be minimized by eco-oriented approaches, as these approaches can regenerate damaged wetlands and particularly aim to improve the ecological functioning and maximize the naturalness of beauty.

Formal beauty is associated with landscapes that are dominated by human activity. The engineering-oriented design puts human activity at the forefront within these settings. Deterioration of ecological services and functions are foreseeable. The characters of engineering-oriented approaches are degenerated that may show the tendency of unsustainable wetland development.

Increasing population and urban sprawl led to reclamation of natural wetlands, habitat loss, and shifting of wetland types in the affected areas. Thus, the ecological baseline has shifted, increasing the difficulty of wetland conservation. Throughout the history of land protection, esthetic factors are given great importance [16]. That means human esthetic preference and ecological goals are aligned. Therefore, improved esthetics is a key component for ecological restoration of threatened wetlands.

2.2. Impact of man-made structures on coastal wetland

Coastal wetlands are areas where different habitat overlap, such as sea and land, river and estuary, and brackish water and freshwater. Vegetation and animals from both adjoining ecosystems overlap here, resulting in ecotones which are species rich and diverse. Increasing modification of coastal wetlands is a symptom of increasing urbanization and contemporary behaviors.

The coast offers an open ecological environment, rich in scenic beauty that provides enjoyment and contributes to the well-being of people who experience it. Populations in coastal areas are increasing, and the resulting urbanization intensifies the conflict between anthropogenic activities and the coastline. Man-made structures (e.g., seawalls and breakwaters) protect the coastal environment from the impacts of waves, tides, and storms. These structures affect the natural ecosystem and ecotones [1], undermine the coastal scenic value, and obstruct the human access to coastline.

Certain physical attributes of coastal wetlands have influenced on both the landscape esthetic and ecological health including the water body, water shore, and terrestrial vegetation. Each of these attributes constitute to the overall coastal wetland landscape. Environmental perception is derived from the environment, which physically surrounds people [17–19]. Therefore, if man-made structures were constructed, the shoreline is altered and access to the physical environment could be prohibited; this is the primary issue of man-made structures in coastal wetlands.

Characteristics of man-made structures, such as length, shape, height, slope, material, type, and location, influence the perception of the coastal wetland. The length of structures would reduce the attractiveness of the coastal landscape [20] and also decrease landscape room. Landscape room is a perceptual unit related to a visual scale. It takes into consideration the ecological patch size. The view and size of landscape room determines the degree of openness, which affects visual pleasure. Man-made structures fragment coastal wetlands and can limit the exchange of seawater with freshwater, resulting in ecotone loss.

Man-made structures can increase edge abruptness and inflict straight boundaries, both edge effects tend to decrease species movement across an edge. These structures are generally straight, hard edged, and simple and have produced monotonous and visually uninteresting coastal wetlands. The less variety of landscape elements produces a setting that fails to induce curiosity, indicated are unlikely to explore for more environmental information. Furthermore, a completely blocked view or landscaping barrier fails to go deeper to get more environmental information and thus is not favorable landscape. The height of typical man-made structures obstructs visual penetration and esthetic value [21] and reduces species movement, eco-hydrological function, and energy flows [1, 22].

The different types of man-made structures are discrepancy in location, width, height, and slope, etc. Coastal ecotones have vanished due to the wide, tall, and steep design of man-made structures [23], which limited “accessibility.” Limited accessibility has several implications which can affect perception and include the distance a person can stand from seawater, decreased visual penetration, and a decrease in obtainable environmental information. More important, species movements of both adjacent land and sea were interrupted. Each of these implications could reduce the likelihood of individuals to connect with and pursue conservation action of the coastal wetland environment. Environmental information is not readily available in the current setting; water and land are separated and the coastal wetland has lost its landscape ecology characters.

Landscape esthetic impacts	Characteristics of man-made structures	Ecological function impacts
Diminished scale of the landscape room and reduced environmental information. Altered skyline to monotonous spatial landscape.	Length	Reduced area and quality of coastal wetland, increased probability of species loss. Habitat fragmentation could reduce population size, habitat diversity, and species diversity.
Visual landscape diversity weakened, decreased availability of environmental information.	Shape	Man-made structures made the patch boundaries straight, hard, and homogeneous. The edge effect influenced the flow of nutrients, water, energy, and species movement.
Obstruction of visual penetration. Reduction in the openness of the spatial landscape. The closed setting reduces available environmental information.	Height	Ecotones vanished because man-made structures serve as barriers that divide sea and land, and restrict species movement and energy and water flows.
Water and coastline accessibility decreases with increasing slope.	Slope	The near-vertical slope of structures reduces the available inter-tidal habitat on seawalls, which could reduce species richness and abundance.
Unvaried surfaces make the spatial landscape is too tidy, uninteresting and unliving, reduction in the amount of environmental information available.	Material	The substrate is different between levee and natural ecotone and does not support species endemic to coastal wetlands.
Structures type can influence accessibility and visual variety. For example, structures upon which vegetation can grow increases the amenity of the setting.	Type	Different types of levee may receive either daily or less frequent tidal inundation which could affect vegetation and decrease or fragment coastal wetlands.
Improper structure location will affect the holistic coastal wetland, thus the size of landscape room.	Location	The location of a structure affects habitats redistribution. The sea and land both could be damaged from segmented habitats.

Table 1. The impacts of man-made structures on landscape esthetics and the ecology of coastal wetlands.

Finally, the material in which man-made structures are made from is associated with a lower probability of species colonization. The shape and material of the structure are two key factors that will influence their performance as ecosystem services providers [23]. **Table 1** details the impacts of man-made structures on the landscape and ecology of coastal wetlands.

3. Esthetic and ecology aspects of man-made structures

Many approaches have been applied to mitigate the ecological impacts of man-made structures on coastal wetlands as reported by Wiecek [24]. These mitigation efforts also need to incorporate a landscape approach to improve esthetic value [25]. Landscape esthetic preference stems from the evolutionary survival experience. In this way, landscape esthetic is aligned with ecological health. The application of landscape approaches to the development of man-made structures aims to benefit the landscape esthetic and thus ecological health coastal wetlands.

Coastal wetlands were fragmented by man-made structures. The modified coastal wetland and natural coastal wetland can be interconnected by ecological esthetic approaches. The concept of esthetic ecology introduces aspects to the man-made structures that simulate the natural landscape esthetic of coastal wetlands (**Figure 5**). If the boundary of a man-made structure does not coincide with the natural wetland boundary between both seawater and land, the landscape esthetic and ecological quality will be reduced. The concept of landscape esthetics to improve modified coastal wetland ecology involves linking the existing man-made structures and coastal wetland. The aim is to keep or restore the ecological baseline through landscape esthetics to benefit coastal wetland habitats and conservation.

Coastal man-made structures were constructed to reduce erosion and flood risk and to maintain human activities and safety. When ecological esthetics is considered, the prime objective of the man-made structure is still to protect humans and coastal stability and then fundamentally set to ensure landscape esthetics.

Landscape perception is individualistic and related to the spatial landscape composition. Esthetic appreciation indicated the perception response when people enter a landscape room. Naturalness and openness of the landscape room is highly favorable, while a unitary atmosphere also affects landscape esthetic preference. A distinctive landscape could stimulate interest and for this reason, preservation of esthetic scenery generally appreciated by many is highly important. People will stand at a spot to absorb a pleasing view; this spot is the viewing place, and the view is the target scenery. This interaction between human selection of a viewpoint and the landscape is similar to the preferable prospect-refuge character landscape setting according to habitat theory.

Man-made structures have often been constructed to truncate the landscape room. Since the visual field is bounded by the structure, the middle to background of the landscape is often subject to disappearance. The first step to improve the landscape esthetics in this scenario is to identify and preserve the optimal viewing place that provided a view of the target of scenery prior to the construction of the man-made structures. The alternative option is to preserve the target scenery and recreate new viewing places. The former ensures a unitary atmosphere and that the landscape room is not affected by the construction of man-made structures. The latter ensures that the accessibility to appreciate target scenery is not compromised by structures. The mitigated approaches of man-made structure for landscape aesthetics and ecologic health as shown as **Table 2**.

Wherever possible, minimized length of structures could moderate the impact on coastal wetland fragmentation, which is also beneficial to landscape beauty. The boundary between the sharp outline of man-made structures and the dyke foot needs to be blurred in order for the structure to integrate into the landforms of the coastal wetland. The dyke foot on the land side of the structure can be rebuilt using natural materials, such as boulders, stones, and fill soil, along with vegetation planting to make the simple boundary become various visual pictures. These settings may provide more environmental information to people than previously.

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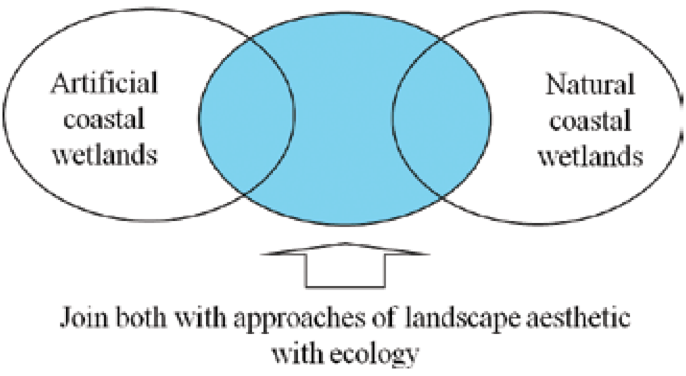

Figure 5. Framework of the integrated consideration of the esthetic and ecologic aspects of man-made structures.

Characteristics of man-made structures	Artificial structure mitigation for landscape esthetics and ecologic health
Length	Minimize, keep the landscape room is given in terms of human scale, reduce habitat fragmentation.
Shape	Use the shape of the existing shoreline for guidance in order to promote naturalness across the visual landscape and landform.
Height	Raise the viewing point and merge the structure into the existing landform. Moderating the impact from obstructed views will help renovate the landscape room.
Slope	Gentler slopes broken up with vegetation and natural materials could make the man-made structure more visually pleasing. If the change of slope is in accordance with existing landforms of coastal wetlands, it is good for environmental compatibility.
Material	Use of natural material could help blend the man-made structure with existing landforms and the overall coastal wetland landscape.
Type	Minimize size and combine the man-made structure with vegetation to create visual variety and improved accessibility.
Location	Immerse the structure into water or move away from the ecotone. The former preserves landscape room perfection, whereas the latter could reduce visual impact, as it is important to maintain the ecotone.

Table 2. The concepts of landscape esthetics and ecologic health applied to man-made structures on coastal wetlands.

The shape of man-made structures is often straight and rigid. This is due to the stability needed for coastal defense, but the visual character is tall, wide, and long and is thus perceived as arid, too orderly, and ecologically unhealthy. The setting is unattractive, and therefore, the shape of the structure must be in accordance with the coastline characteristics. This may transform the straight and hard impression left by the structure, to a gentler, more interesting, and preferable landscape. If no structural alterations can be made to improve the structure beauty, vertical lines can be applied to the surface of the structure to potentially mitigate the initial perception of a solid image.

When a structure is higher than the line of sight, the view and access to the sea is blocked. This violently decreases the landscape room and has significant impacts on landscape esthetic preference. The height of a man-made structure is the one of the most important landscape esthetic issues facing coastal wetlands, as the height with material influences visual penetration, water accessibility, sea and land ecology, and ecotones [26]. In the optimal approach, the height of a man-made structure is determined by whether people view and access the coastline from land. Moderate raises in land elevation could improve visual accessibility and decrease the influence of the man-made structure on the landscape. Other mitigation efforts, such as filling in soil on dyke foot and planting vegetation, may smooth the visual constraint imposed by the height of the structures.

When a structure is steep, the view field is narrowed and the middle to background environmental information is no longer visible. This setting is not favorable. To mitigate, decreasing the slope gradient is an option; however, this may enlarge surface area. Thus, the surface area can be divided to make the slope appear more interesting, or vegetation along with natural material can be applied to create features consistent with the adjacent landform to help visual integration with the coastal wetland. These approaches could create new target scenery, improve diversity of visual landscape, and enhance visual and water accessibility.

A variable of landscape esthetic experience is the viewing distance, as it is concerned with both long- and short-range views [27]. People view an overall landscape using a long-range view, of which landscape room is an important consideration. The short-range view is more concerned with the amount of detail which can be seen. Therefore, the scale of the relationship between observer and landscape is important.

Landscapes in the short-range view require finer consideration of constructional details, such as material and the texture of surface finish design [18, 28]. An inviting foreground setting, which is part of the short-range landscape, is critical for attracting viewers and provides the starting point for the sequential visual experience of the landscape. This could entice people to explore the coastal wetland landscape further and then would relate to the long-range landscape.

Natural materials were blended with the man-made structures, which contribute to the continuance and unification of the landscape and ecology. Natural materials can be used to reconstruct local characteristics to recover the relationship between humans and the environment; visual preference could achieve by through the use of “vernacular cues to care” [15]. Soft, curvilinear boundaries constructed from natural materials at dyke foot on the land side of a structure could create micro-patches capable of providing a number of ecological benefits. This also promotes a more interesting spatial landscape providing support for more environmental information than previously available in the short-range view.

The large and fair surface of existing man-made structures make the landscape setting highly homogeneous and monotonous, thus decreasing landscape esthetic and as a result is not a visually favorable landscape. Natural materials can be associated with man-made structures

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to improve visuals could also lead to the creation of ecological corridors. Furthermore, the vegetation structure and floristics could be aligned with the adjacent habitat, possibly facilitating species movement and recolonization.

Different types of man-made structures have different impacts on coastal wetland landscapes. No matter which types of structure to create visual variety could enhance the spatial landscape attractive is primary. The surface of man-made structure is commonly flat and monotonous. If the surfaces were divided into small parts, and finished by composite materials; which landscaping approaches could create more interesting setting. Step-type dike is a good option. Planting short vegetation is an excellent way to recreate habitats at low steps, especially using native vegetation. Meanwhile, taller vegetation, such as trees, and the interaction with sunlight provide shade at the up-steps area, providing a more color-rich and interesting setting. Vegetation also varies surface structure, while clusters of trees create various heterogeneity of spatial landscape which is attractive.

The relationship between coastal wetland and man-made structures can also divide into two types. The first type is when a structure is parallel with the coastline, such as seawall or an offshore breakwater. These structures separate seawater and land, therefore destroying the ecotone and causing coastal wetland destruction. Approaches to improve the landscape esthetic and ecological quality in such cases are similar to those previously discussed. The second type is perpendicular to the coastline, such as jetty, and breakwater. These structures fragment coastal wetlands. Reducing the height of a structure can mitigate this impact and help maintain landscape room integrity.

It is well known that coastal ecotones are among the most productive ecological habitats and provide many functions which benefit humans and the environment. For this reason, man-made structures immersed in seawater are preferable as impacts on the coastal wetland landscape esthetics are minimized. An alternative option is moving the structures away from the intertidal zone to land, increasing the capacity for coastal wetland environments. This also supports the notion that increased spatial scale improves landscape room and thus preferable for landscape esthetics. However, in this case, the man-made structures may cause loss of littoral forest, and smaller patches may lead to decreased habitat diversity and the number of species. The naturalness of the landscape could become spoiled, thus impacting perception of the landscape. Coastal wetlands are important landscapes and recreational areas for local people. If man-made structures are built adjacent to communities or crowded areas, easy access to coastal wetlands must be maintained using the previously mentioned strategies to improve landscape esthetics.

Vegetation planting is the most common option to mitigate the visual impacts of man-made structures, which approaches could make structures merge into the coastal wetlands landscape. Compared with the mismanagement of modified setting, trees could be used to spatially vary the visual perception of the landscape as a greater diversity in landscape preferred by people. Structure edge can also be mitigated by adding edge vegetation of high diversity both vertically and horizontally to soften the edge and enrich landscape diversity [29]. The desirable mitigation approach encourages the diversity of habitats concurrently.

Furthermore, the sequences of landscapes are important [30]. The coherence of environmental information in the foreground and the setting is legibility that will make people feel secure. Following the complexity and mystery landscape at middle-ground to background encourages viewers to look further into the next setting to gain more environmental information. That landscape is favored by people. Furthermore, image congruity between the residential environment and the coastal wetland, promotes a sense of place attachment and landscape esthetic preference, potentially promoting conservation actions. As a result, the landscape arrangement of man-made structures accords with the local fabric, especially in coastal communities and fishing villages.

4. Conclusion

Coastal wetlands are under constant pressures resulting in habitat loss and degradation. To prevent further losses, environmentally friendly man-made structures which mimic the foreshore environment have been applied to minimize negative environmental impacts and maximize environmental value. Wetland conservation, specifically through esthetic awareness would more benefit to maintain, protect, and restore wetland habitats. However, many existing wetlands are of low environmental quality, convoluting the ecological baseline and landscape esthetics. Shifting baseline is a phenomenon where successive generations accept unknowingly the degraded quality of coastal wetlands as pristine, thus conservation action becomes less of a priority for younger generations.

The role of familiarity is important in terms of landscape preference, as it has a positive correlation with landscape preference. Consciousness of the impact man-made structures have on landscape perception of coastal wetlands may diminish over time. Will people have a continued interest on the impacts of man-made structures on coastal wetland landscape and healthy ecological functioning as familiarity of the modified or artificial coastal wetlands increases?

Environmental legislation and policy have set the protection of coastal wetlands as a priority; however, increasing economic and land use pressures continue to reclaim land from the sea, made possible by man-made structures, still impacts the coastal ecotone. The optimal scheme is for man-made structures to not only to protect the coastline but also to create high-quality landscape, through mitigation measures such as beach nourishment and artificial headland. These options can minimize disturbance to the natural coastline, while having a positive effect on the sediment downstream. Landscape esthetics can be preserved, thus limiting the negative impacts of man-made structures on coastal wetlands. Artificial reefs and submerged dikes could form underwater habitats, maintaining landscape esthetics. Offshore breakwaters can fall below the mean tidal level, ensuring that visual impacts are minimized while also achieving preferable ecological benefits [1, 22].

If there is no immediate pressure for land expansion, man-made structures should not be built or, if possible, kept away from coastal wetlands, located it on the land side. And to retain a buffer zone between ecotone and man-made structure, reasonable landscape room is required to satisfy esthetic, and this must be considered prior to determining the layout of man-made structures.

Climate change and sea level rise pose increasing coastal erosion and seawater instability risks. If wetlands are flooded, vegetation cannot grow and the edges of coastal wetlands are degraded. This makes maintaining healthy coastal wetlands even more challenging. Man-made structures are required to protect coastal wetlands; stability and safety of the coastline are the primary objective. Thus, consideration of landscape esthetics, which promotes healthy ecologic functioning, needs to be put into practice to optimize coastal wetland structures for enhanced conservation of these sensitive environments.

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Section 5

Valuation of Wetlands

Chapter 9

A Large-Scale Wetland Conversion Project in Southeastern Missouri: Sustainability of Water and Soil

Michael Thomas Aide, Byron McVay,
Indi Braden and Christine Aide

Additional information is available at the end of the chapter

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Abstract

Wetland conversion in southeastern Missouri initiated with the Little River Drainage Project (1914–1924) resulting in the permanent drainage and conversion of 5 million acres (2 million hectares) to productive agricultural land. Given that this ancestral wetland conversion has totally replaced the wetland ecosystem with prime agricultural land and with this conversion, the loss of wildlife habitat is nearly complete, the question remains what actions are now possible to restore key wetland soil pathways to support soil health and water quality. Key to any corrective practices involves agricultural producer involvement and commitment. The emerging concept of soil health supports the use of cover crops that promote soil structure development and soil carbon sequestration, each perceived as supporting farm profitability. Government programs supporting field flooding during the off-season supports migratory water fowl. Farming practices such as furrow irrigation and allied technologies for rice production limit aquifer overdraft. Edge of field technology involving riparian strips and denitrification bioreactors support downstream water quality by limiting nitrate and phosphate off-field migration. The result is that emerging technologies (i) support farm profitability and environmental stewardship and (ii) which are designed specifically to provide farming practice compatibility with the soil and water resources re-establishes some wetland mechanisms appropriate for long-term land and water resource sustainability.

Keywords: wetland conversion, hydric soils, soil health, edge-of-field technologies, cover crops

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1. Introduction

In the USA, it is estimated at 90% of the wetlands that existed prior to European discovery and settlement have been converted to other uses, most notably agricultural usage [1]. Land drainage has been extensive in many states, such as Alaska, Florida, Indiana, Minnesota, Missouri, and Wisconsin. Land drainage, both subsurface tile-drainage and surface drainage with and without diversionary earthworks, dramatically altered these ecosystems and their attendant soil and plant processes. Large scale and substantial changes in vegetation, water availability, nutrient flow, and other characteristics of these ecosystems impact the flora, negatively impact water quality, and reduce soil health, yet the economic impacts are important social restraints on returning these areas and regions to their pre-European settlement status.

Given that the return of thousands of hectares of cropland back to wetland status is not a pragmatic solution, current features of USA agriculture policy attempt to support best use methods that both support farm profitability and align sustainable agriculture production to encourage soil health, organismal diversity, and environmental stewardship.

2. Wetlands and prior converted wetlands

Our purpose in creating this manuscript is to chronical re-establishment of important wetland plant-soil interactive processes in converted wetlands to support soil health, water quality, environmental stewardship and biological diversity, while maintaining agricultural productivity. Although programs exist to re-create wetlands from agriculture land, there are pragmatic social, political and economic realities that limit their large-scale application. The application of emerging technologies and governmental policies, designed to support important soil attributes reflective of the original wetland status, provide opportunities for both environmental advancement, and agricultural profitability.

To be designated as "Prior Converted Cropland" in the USA, all the following land criteria must be validated: (i) cropped prior to December 23, 1985 with an agricultural commodity, (ii) cleared, drained or otherwise manipulated to make it possible to plant a crop, (iii) continued to be used for agricultural purposes, and (iv) does not flood or pond for more than 14 days during the growing season [1]. Vital wetland soil processes that need to be re-emphasized in converted wetlands include: (i) synthesis and subsequent maintenance of soil organic carbon, (ii) maintenance of soil biological diversity, including microbial populations, (iii) erosion abatement, (iv) unimpeded activity of nutrient cycles, especially the nitrogen cycle, (v) development of the original soil structure fabric, (vi) appropriate water transport within and among pedons, and (vii) encouragement of microbial-driven ecosystem processes that reduce excessive plant nutrients and degrade applied agrichemicals within suitable time frames.

3. The study area and the Little River Drainage System in Missouri

The study area ranges from the St. Francois River in the west to the Mississippi River in the east and ranges from the headwater diversion channel at Cape Girardeau Missouri to the

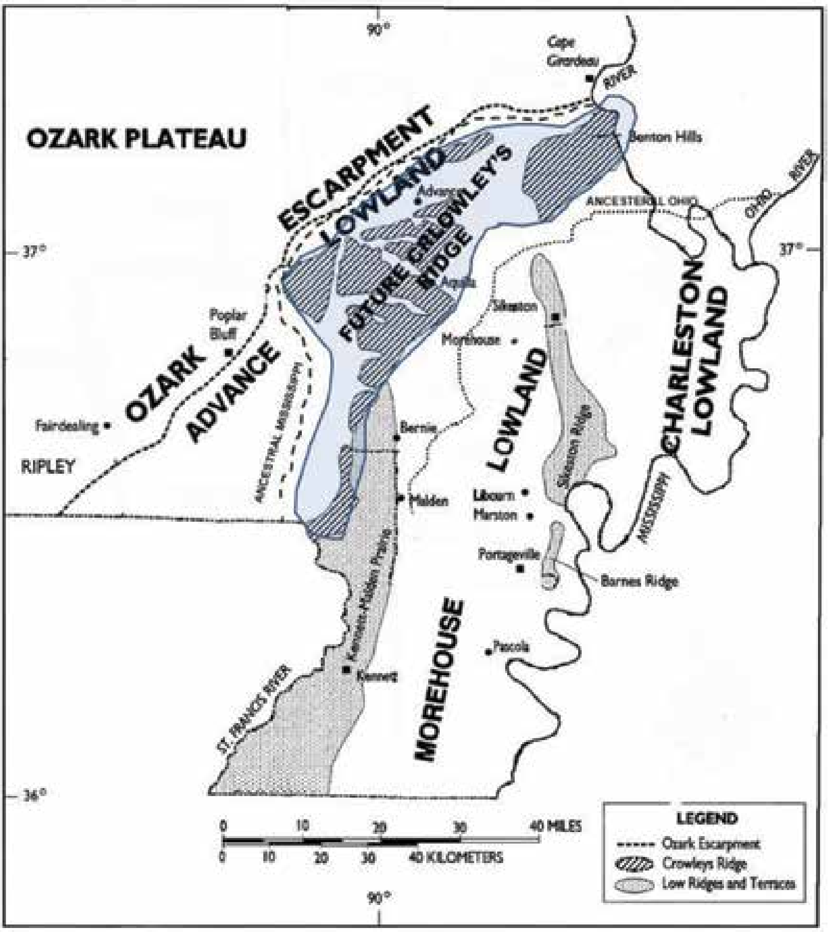

Figure 1. Landscape features of Southeastern Missouri.

Missouri – Arkansas border (**Figure 1**). Approximately 5 million acres (2 million hectares) of landscape was drained from its status as wetlands to produce an intensive agricultural setting. The entire drainage system is maintained by taxes leveed on agricultural producers [2].

A series of north to south drainage ditches (1541 km) and levees (490 km) were constructed to transport water from southeast Missouri into Arkansas and then into the Mississippi River. The Headwater Diversion Channel was constructed to intercept drainage of the Castor and Whitewater Rivers, transporting this water eastward into the Mississippi River. Dams created the Clearwater and Wappapello Reservoirs by intercepting drainages of the St. Francois and Black Rivers, respectively [2].

4. Climate of Southeastern Missouri

The climate is continental humid. The average daily January temperatures are 2 and 4°C (35 and 39°F) at Cape Girardeau and Kennett, Missouri, whereas the average summer temperatures are 25 and 26°C (77 and 79°F) at these locations. As expected, June–August are the warmest months. The growing season generally has 210-plus frost-free days. The soils are frozen only at the surface and only for brief periods of time. The rainfall is reasonably well distributed, with the total annual precipitation averaging 1.14 m at Cape Girardeau and 1.27 m at Kennett. The remnants of tropical storms from the Gulf of Mexico may provide more than 0.25 m of rainfall during a rainfall event [3].

5. Wetlands and hydric soils in the study area

We define wetlands as soilscapes transitional between terrestrial and aquatic systems that support hydrophytes and possess an undrained substratum having anoxic conditions, typically having a water table for some portion of the time when the soil temperature is above biologic zero. In our study area, a large-scale drainage management system has been implemented to produce agriculture lands; however, the ancestral wetlands would have been classified as southern deepwater swamps and riparian forested wetlands [4]. Hydric soils are defined as "soil that is saturated, flooded or ponded long enough during the growing season to develop anaerobic conditions in the upper part" [5]. Prolonged anaerobic conditions promote selected biogeochemical soil processes that modify the soil morphology, such as (i) organic matter accumulation, (ii) iron and manganese oxyhydroxide transformations/depletions/accumulations, (iii) sulfate-sulfide transitions, nitrogen transformations, and (iv) biogeochemical nutrient cycle alterations. These indicators are used to delineate hydric soils; however, in the USA the indicator criteria may vary among the major land resource areas. Wetland delineation in the USA is based on the presence of hydric soils, the local hydrology, and wetland indicator plants.

6. Landforms and vegetation

Much of the natural vegetation has been removed and replaced with agricultural enterprises. Depressional areas consisting of backswamp deposits typically supported bald cypress (*Taxodium distichum* L.), water tupelo (*Nyssa sylvatica*), sweetgum (*Liquidambar styraciflua* L.), and multiple species of canes, rushes, and grasses, whereas recent meander belt deposits have willows (Salicaceae sp.), eastern cottonwoods (*Populus deltoides* Marsh.), American elm (*Ulmus americana* L.), yellow poplar (*Liriodendron tulipifera* L.), and boxelder (*Acer negundo* L.). Mixed forest species existed on well-drained to moderately well drained soils residing on variably textured alluvium and natural levees. Mixed forest species included: southern red oak (*Quercus falcata* L.), willow oak (*Quercus phellos*), white oak (*Quercus alba* L.), swamp white oak (*Quercus bicolor* Willd.), and shagbark hickory (*Carya ovata* Mill.).

The Southeast Lowlands Groundwater Province (SLGP) is bounded on the north and west by the Ozark Plateau, with the transition from the SLGP to the Ozark Plateau called the Ozark escarpment. The eastern boundary is the modern Mississippi River and the southern boundary is the Missouri-Arkansas state border. The western boundary of Dunklin Co. is the St. Francois River (**Figure 1**).

A prominent ridge within the SLGP is called Crowley's Ridge and the Benton Hills. These elevated land masses consist on Paleozoic rocks, largely Ordovician, and are covered by Tertiary gravels and loess [6]. Crowley's Ridge and the Benton Hills bisect the SLGP, with the land mass between Ozark Escarpment and west of Crowley's Ridge and the Benton Hills called the Advance Lowlands (also called Western Morehouse Lowlands). The Advance Lowlands represent the ancestral channel of the Mississippi River and are generally composed of loamy to silty terraces and back-swamp deposits overlying glacial outwash and valley train. Conversely, the Morehouse Lowlands extend from Crowley's Ridge eastward to the Mississippi River. The Morehouse lowlands consist of terraces of varying textures, back-swamp deposits, and other alluvial environments overlying braided glacial outwash. The modern Mississippi River and its flood plain is the youngest and easternmost feature with meandering channel deposits, natural levees, silty terraces, back-swamp environments, and crevasse splay deposits that characterize the Mississippi River floodplain [7]. Sikeston Ridge and Barnes Ridge (east of Portageville, MO) are low-elevation ridges, composed of coarse-textured materials and both are in the Morehouse Lowlands. Just east of Crowley's Ridge and extending into Dunkin Co. is a terrace system of coarse-texture materials called the Kennett-Malden Prairie. The Charleston Lowland is located primarily in Mississippi Co. and consists of fine to coarse textured materials, composed to recent terrace and back-swamp environments. Between the Benton Hills and the Charleston Lowlands is the Blodgett terrace composed of coarse-textured materials and the Charleston Fan, also composed of coarse-textured materials.

7. Drainage patterns

The study area is bordered on the west by the St. Francois River and on the east by the Mississippi River. Each of these southerly flowing river systems may alternately supply floodwaters or provide surface drainage. A series of dendritic streams and rivers drain the Black River Ozark, the Inner Ozark, and the Outer Ozark Border regions, providing surface waters to the Advance Lowlands and the Morehouse Lowlands. These rivers include: the Black River, White River, Castor River, and the St. Francois River. In addition, small streams provide drainage from the Benton Hills and Crowley's Ridge, providing water to the Advance and Morehouse Lowlands [8].

8. The value of the agricultural productivity

The dominant crops in the study area include: corn (*Zea mays* L.), soybeans (*Glycine max* (L.) Merr), cotton (*Gossypium hirsutum* L.), wheat (*Triticum aestivum* L.), and rice (*Oryza sativa* L. (indica)). Other commonly cultivated crops include: potatoes (*Solanum tuberosum* L.), sweet potatoes (*Ipomoea batatas* (L.) Lam), cowpeas (*Vigna unguiculata* (L.) Walp), winter squash (*Cucurbita sp.*), sorghum (*Sorghum bicolor* L.), watermelons (*Citrullus lanatus* (Thunb) Matsum, and Nakai), peanuts (*Arachis hypogaea* L.), and a variety of vegetable crops. The study area is the most intensively cultivated region in Missouri and having the longest growing season.

The area also has the highest percentage of level and tillable land, of which 60–70% is irrigated with abundant groundwater resources. Animal agriculture is very small, consisting of a few beef cattle and horse operations.

In southeastern Missouri, there are 4133 farms [9]. Cape Girardeau County has more than 1100 farms, thus approximately one-quarter of all farms of the eight-county region are in Cape Girardeau County. In the Mid-South region, Stoddard County has the largest number of farms, many of which are smaller farms on upland hills. The range in farm size varies from small land parcels (less than several hectares) to large farming operations (greater than 5000 hectares) [9].

The study area's population is low, with 223,000 persons. To estimate the values of the agriculture production, the annual crop production by county [9] was multiplied by commodity prices for that time [9]. The annual value of the agriculture production from cropping systems is $1.27 billion (2016). The five-year (2012–2016) average value of production for the dominant crops include: (i) corn ($325 million) and (ii) soybeans ($525 million), cotton ($200 million), rice ($150 million), and wheat ($75 million). For the same five-year period, the mean crop yields are (i) corn (8844 kg/ha), (ii) soybeans (2722 kg/ha), cotton (1177 kg/ha), rice (7706 kg/ha), and wheat (3965 kg/ha).

To estimate the agribusiness sales of production inputs, the product of the county harvested acreages [9] and the University Missouri crop budgets [10] were utilized. The profitability of the agribusiness sector includes: (i) seed sales for corn ($47.38 million), soybeans ($77.94 million), wheat ($7.88 million), cotton ($29.52 million), and rice ($5.25 million), (ii) fertilizer sales for corn ($57.8 million), soybeans ($51.5 million), wheat ($14 million), cotton ($17.6 million), and rice ($19.8 million), and (iii) herbicide sales for corn ($14.1 million), soybeans ($44.3 million), wheat ($5.6 million), cotton ($14.5 million), and rice ($16.1 million).

9. The geological history of the study area

The Mississippi River embayment was initially created by an ancient down warping of the crust, presumably by tectonic forces. Confining our discussion to the Pleistocene-Holocene the ancestral Mississippi River occupied an Advance Lowland course until the late Wisconsin sub-stage [6, 11, 12]. These authors proposed that the diversion of the Mississippi River through the Bell City—Oran Gap, abandoning the Advance Lowlands and entering the Morehouse Lowlands, was initiated approximately 17,000 year BCE (before common era) and was complete by 11,500 year BCE. By 9800–9900 year BCE, the Mississippi River changed from a braided river to meandering river, passing through the Pemiscot Bayou [12]. After the diversion, the advance lowlands continued to receive sediment from the Ozark Plateau, principally from the Little Black River, the Current River, the Spring River, and the White River. Blum et al. [13] proposed that the Bell City—Oran Gap diversion into the Morehouse Lowlands occurred before 60,000 year BCE, thus placing the Bell City—Oran Gap diversion before the Wisconsin glaciation. Royall et al. [12] proposed that the Ohio River produced two braided stream terraces in the Morehouse Lowlands between Crowley's Ridge and Sikeston Ridge.

Blum et al. [13] map Sikeston Ridge as a late Wisconsin valley train having a very thin loess capping of Peoria Loess. Blum et al. [13] further attribute the Blodgett terrace as a braided terrace deposit of the Ohio River, which was entrenched within the Cache River Valley (Illinois). Based on carbon dating, Blum et al. [13] place the Charleston Fan as a Mississippi River feature formed during the creation of Thebes Gap (10,590 year BCE).

Approximately 9000 year BCE, the Mississippi River diverted through Thebes Gap and flowed east of Sikeston Ridge [12], creating the Charleston Alluvial Fan. The study area has been extensively modified by seismic activity, featuring sand blows, sand boils, clastic dikes, liquefaction, changes in stream drainages, and subsidence [14–16]. A prominent trend of earthquake epicenters has been related to deep-seated folds and igneous intrusions [16]. Loess deposition as a capping on soils in the advance lowlands shows both the stage of development and bisequal nature of these soils [17].

10. The southeast lowlands groundwater province in Missouri

The Southeast Lowlands Groundwater Province in Missouri (SLGP) spans 10,142 km^2 and contains 15.2% of the State of Missouri's groundwater, estimated at 287 billion m^3. The Cretaceous age McNairy aquifer crops out (at or near the surface) on the flanks of Crowley's Ridge and the Benton Hills [18]. In Stoddard County and Butler County, the McNairy formation primarily underlies alluvial materials, whereas in Dunklin County and Pemiscot County the McNairy formation is reached by wells having a depth of 600 m. In Dunklin County and Pemiscot County where wells are in thick and clean sands, the water yields range from 570 to 2800 L min^{-1}. Overlying the McNairy formation, the Clayton Owl Creek and Porter's Creek clay formations constitute confining layers. Water from the McNairy formation in the northern regions along Crowley's Ridge are classified as iron rich, calcium-magnesium carbonate type waters, whereas waters from the McNairy formation in the southern portion of southeastern Missouri are sodium chloride type waters.

The Wilcox Group is composed largely of Tertiary-age sands, some regions having minor inclusions of lignite and clay. The Wilcox aquifer is commonly separated into the upper and lower Wilcox aquifers because of sand grain size distribution patterns. The Wilcox aquifer overlies the Porter's Creek clay and is largely absent in northern Stoddard County and attains thicknesses greater than 427 m in Pemiscot County and Dunkin County. Water yields from the Wilcox in Stoddard Co. are approximately 2900 L min^{-1} and in Pemiscot County are approximately 6400 L min^{-1}. The water composition is calcium-magnesium carbonate or calcium carbonate [18]. The Claibourn aquifer lies on the Wilcox aquifer. The Claibourn aquifer is separated in the upper, middle, and lower Claibourn aquifers, with the upper and middle Claibourn aquifers separated by a layer of thin, clayey materials that act as a confining unit (aquitard).

The Mississippi River Valley Aquifer (the Southeast Lowlands Alluvial Aquifer) consists of unconsolidated clay, silt, sand, and gravelly textured alluvium. Groundwater usage of the Mississippi River Valley Aquifer constitutes approximately 92% of the groundwater withdrawal in southeastern Missouri. These alluvial materials were largely deposited by the ancestral Mississippi and Ohio River systems, coupled some prominent deposits by the Black, St. Francois, and Little River systems. Alluvial thickness is variable, with typical thicknesses west of Crowley's Ridge ranging from 15 to 45 m, whereas the alluvial thicknesses in Mississippi, Pemiscot, and Dunklin Counties average 76 m. These unconfined aquifers are baseflow recharged annually from the Mississippi River, other prominent rivers and land drainage ditches. Water yield ranges from 3800 to 11,360 L min^{-1}; however, although water level fluctuations do occur between wet and dry seasons, no long-term depletions have been observed [18].

11. Observations of water levels in test wells in Southeastern Missouri

The study area is extensively irrigated, with many counties having center pivot, furrow and flood irrigation covering 60–70% of the landscape. Ten wells operated and continuously monitored by the United States Geological Survey (USGS) are located across the survey area [19], which sample groundwater associated in the unconfined surficial (alluvial) aquifers. The depth to the mean water table ranges from 1 to 8 m.

For example, in the community of Delta, Missouri, the USGS water level monitoring well continuously documented well water levels centered around 5–7 m below the land surface (**Figure 2**). During very dry summers, the water levels subsided to approximately 8 m and then the water levels rebounded during the winter/spring season to approximately 4.6 m from the surface. In each year and for each of the test wells, the wetter winter/spring season permitted aquifer recharge because of rainfall infiltration and baseflow.

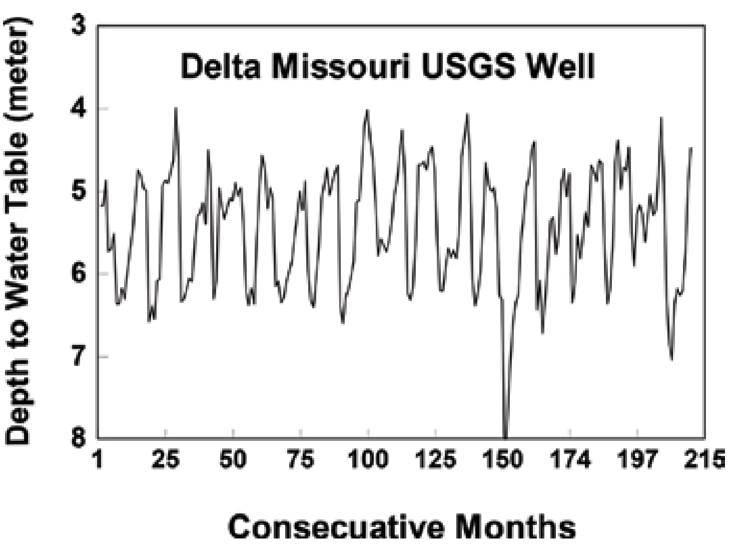

Figure 2. Water depth levels for the Delta Missouri USGS monitoring well from 2000 to 2018 [19].

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12. Soils and soilscapes

In the study area, presentative soil orders (US Soil Taxonomy) include: Alfisols, Entisols, Histosols, Inceptisols, Mollisols, and Vertisols. Landforms include: alluvial fans, splays, flood plains and backswamp deposits, ox-bows and meander channels, Holocene and Pleistocene terraces of coarse to fine-silty textures, and modern to old natural levees and constructed levees. A great portion of the landscape has been recently land-graded for furrow and flood irrigation.

For example, the Cooter-Hayti-Portageville Soil Association rests on Holocene sediment having a transitional texture from sandy alluvium to silty-loamy alluvium to clayey alluvium (**Figure 3**). The poorly drained Portageville clay series (Vertic Endoaquolls: Ap/A–Bg–Cg) exhibits soil organic matter accumulation and soil profile depletion of Fe attributed to anaerobic conditions, whereas the Cooter (Fluvaquentic Hapludolls: Ap/A-2C1-2C2) is a bisequal soil (clayey over sandy) featuring few subsurface redoximorphic features because of the quartz parent material. The fine-silty, poorly drained, non-acid Hayti series (Mollic Fluvaquents: Ap-C) developed in recent silty alluvium lacks soil profile development because of the lack of time for soil profile horizonation.

The Memphis-Loring-Calhoun-Foley Association rests in the Advance Lowlands (also called the Western Morehouse Lowlands) with the fine-silty, deep, strongly acid, well-drained

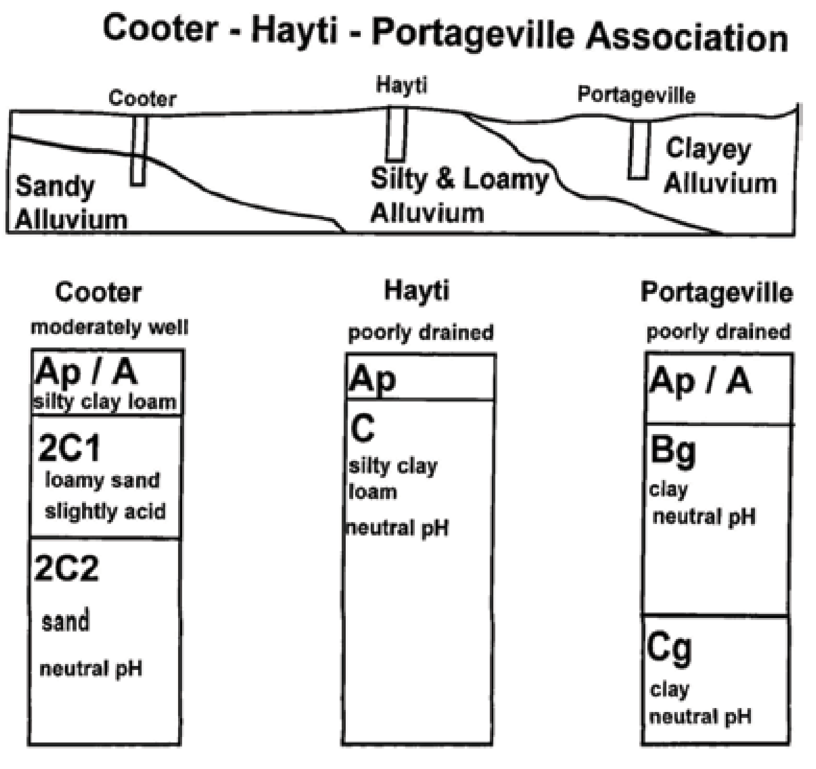

Figure 3. The Cooter-Hayti-Portageville Association.

Memphis (Typic Hapludalfs: A-E-Bt-C) resting in thick loess on Crowley's Ridge (**Figure 4**). The fine-silty, strongly acid, moderately well-drained Loring series (Oxyaquic Fragiudalfs: A-Bt-Btx-C) possesses a well-developed fragipan whose surface represents a transition from Pleistocene silty alluvium to the overlying loess. The poorly drained, fine-silty Calhoun (Glossaqualfs (Ap-Ed-Btg-BCg) and Foley (Natraqualfs: Ap-Eg-Btng/E-Cg) series rest on Pleistocene terraces and may have thin loess mantles. The presence of argillic horizons in these soils indicate their relative more mature age when compared with the previous Holocene soils.

The Sharkey-Alligator Association is a commonly occurring association in the Morehouse lowlands (**Figure 5**). The soils of the Sharkey series consist of very deep, poorly and very poorly drained, very slowly permeable soils formed on level to nearly level backswamp positions along modern and former channels of the Mississippi River. The Alligator series consists of very deep, poorly drained, very slowly permeable soils formed in clayey alluvium in backswamps and sloughs.

The very-fine textured Sharkey soils (very-fine, smectitic, thermic Chromic Epiaquerts) have an Ap - Bssg - Bssyg - Bssg soil horizon sequence. The soil colors range from dark and very dark grayish brown in the silty clay to clayey Ap soil horizon to dark gray and gray in the clayey cambic horizon. The near surface horizons are slightly acid to neutral and deeper soil

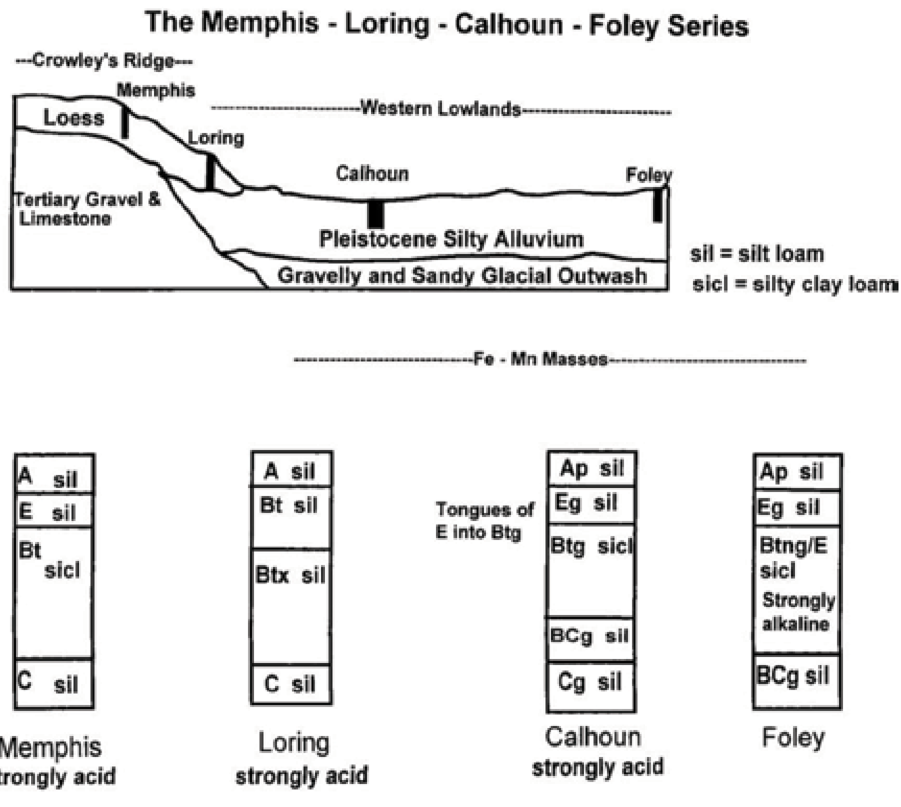

Figure 4. The Memphis-Loring-Calhoun-Foley Association.

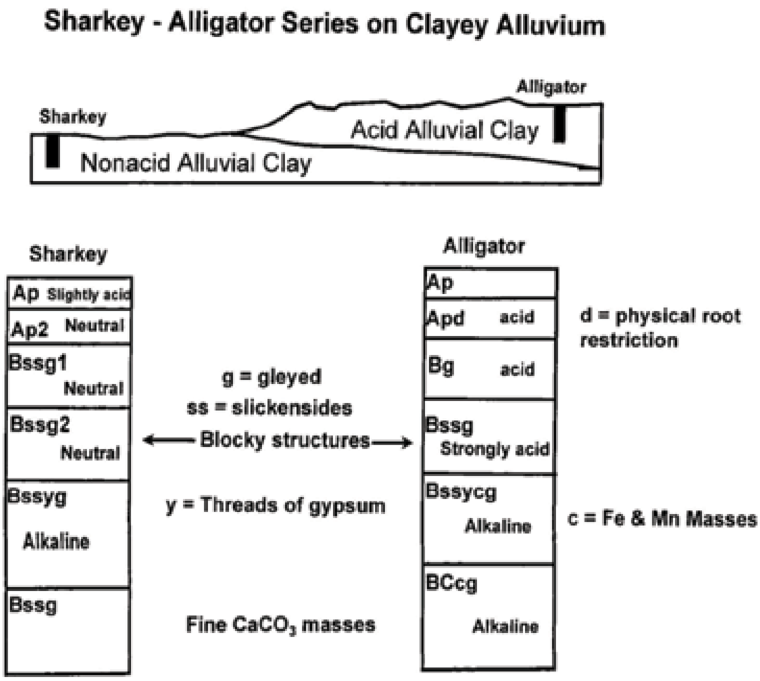

Figure 5. The Sharkey-Alligator Association.

horizons are neutral to moderately alkaline. The cation exchange capacity (CEC) is generally high, attributed to the abundance of smectic clay. The soils of the very-fine Alligator series (very-fine, smectitic, superactive, thermic Chromic Dystraquerts) present an A - Bg - Bssg - Bssycg soil horizon sequence, with all horizons having a clayey texture. These soil horizons are commonly very strongly acid. The grayish brown Bssg and Bssycg horizons have coarse wedge-shaped structures with grooved slickensides on their surfaces. The CEC is generally high, attributed to the abundance of smectite clay.

With the advent and continued maintenance of the Little River Drainage Project the region's hydrologic conditions have been irreversibly altered towards achieving agricultural productivity [20]. Because the Little River Drainage System and its extensions are relatively new and given that soil changes are a function of time and no previous soil baseline data exists prior to land drainage, it is difficult to quantify soil changes because of regional land drainage. Yet soil evolution has been altered and the expected macro-soil changes likely include: (i) loss of accumulated soil organic matter because of oxic soil conditions, (ii) soil acidification coupled with nutrient leaching, (iii) deeper soil water tables resulting in fewer near-surface alternating episodes of soil oxidation-reduction, (iv) loss of soil structure attributed to tillage, land grading, and loss of soil organic matter, (v) changes in the microbial communities, (vi) changes in the invertebrate and vertebrate populations, and (vii) acceleration of mineral weathering intensities, particularly alteration of smectites to kaolinite and apatite dissolution. Because of agriculture, fertilization practices have increased the phosphorus and potassium soil test values.

Horizon	Texture	Color	pH	CEC	SOM	ESP
A	Silt loam	Grayish brown	6.0	14.9	4.3%	<1%
E	Silt loam	Light brownish gray	5.1	12.8	1.9%	<1%
BE	Silty clay loam	Pale brown	5.2	14.5	1.9%	<1%
Btg	Silty clay loam	Light brownish gray	5.4	26.8	1.7%	2.0%
Btgn	Silty clay loam	Grayish brown	7.3	18.0	1.4%	18.0%

CEC is cation exchange capacity (cmol kg^{-1}); SOM is soil organic matter (%).

Btg – argillic horizon (Bt) that is gleied (g or low chroma colors).

Btgn – Btg horizpon that has natric characteristics (high exchange sodium percentage (ESP).

Table 1. The essential properties of the Overcup soil series in an old growth natural forest.

All soil evolution is a complex interplay between horizonation (development of diagnostic soil horizons) and haploidization (the phenomena of organisms and vegetation altering the soil profile to reduce the expression of soil horizons). Land drainage should support the intensity of soil processes to create and maintain soil horizons, particularly albic and argillic horizons. Conversely, loss of soil organic matter will alter mollic (high base saturation and high soil organic matter) and umbric (low base saturation and high soil organic matter) epipedons to orchric (low organic matter) epipedons. Wetlands are commonly acknowledged to purify surface waters and facilitate surface water transfer to shallow aquifers. There is growing concern that land drainage and the associated agriculture will promote nutrient migration and support fresh water eutrophication. Installed levees prevent river flooding in selected areas, leading to greater flooding elsewhere on lands not levee protected. Irrigation may lead to aquifer overdraft; however, this issue is not apparent in this study area.

For example, the Overcup soil series from the Advance Lowlands (fine, smectitic, thermic Vertic Albaqualfs) are very deep, poorly drained, very slowly permeable soils that formed in alluvium. Soil analysis by the authors of the Overcup soil series in both long-term deciduous forest settings and modern rice production fields (unpublished) demonstrate that considerable soil organic matter contents are evident in the forest settings (**Table 1**), whereas the production fields have diminished near-surface soil organic matter contents. The Overcup soil series shows considerable gray color patterns because of seasonal or fluctuating soil water tables within the solum. Soil acidification is evident in the upper argillic horizon, a feature attributed to base removal by leaching. The lower argillic horizon shows a neutral to alkaline pH with a considerable exchangeable sodium presence because restricted drainage has not permitted base leaching, especially including exchangeable sodium. Thus, the placement of cover crops in rice production fields should re-establish soil organic matter contents in the near-surface soil horizons.

13. Policies and practices supporting soil and water sustainability

Currently Federal, State, and agricultural producer partnerships are creating policies and farming practices that support ecosystem health and farm profitability [21]. The goal is to support sustainable and profitable agriculture, while identifying farming practices that are wetland suitable, even when the wetland has been altered by previous land drainage projects. A corollary is attempting to identify wetland benefits and reinstitute practices to return or augment wetland benefits to these altered landscapes while preserving agriculture productivity. One key initiative includes "soil health". Soil health (soil quality) is defined as the continued capacity of soil to function as a vital living ecosystem that sustains plants, vertebrate and invertebrate animals, microorganisms, and humans [22]. This definition speaks to the importance of managing soils to optimize living organisms that contribute to maintaining soil structure, soil organic matter, and functioning nutrient soil and plant connectivity. Considering soil as a living ecosystem reflects a fundamental thinking shift towards nutrient management for plant growth, supporting the soils ability to absorb and hold rainwater for use during dryer periods, filter and buffer potential pollutants from leaving fields, and provide habitat for soil microbes to flourish and diversify. This website [22] provides an annotated bibliography with citations of current literature on soil health initiatives that support water availability, soil structure improvement, soil organic matter optimization (including promotion of active carbon contents), nutrient availability, and limited nutrient transport of nutrients from farm fields to fresh water resources.

A key land practice associated with soil health is the establishment of cover crops. We define cover crops as grasses and legumes cultivated to provide cropland vegetative cover during the off-season to support soil carbon accumulation, improved soil structure (including reduced soil compaction), improved water availability, and substantial reduction is both water and wind induced soil erosion. Our cover crop programs frequently rely on establishment cereal rye (*Secale cereale* L.), crimson clover (*Trifolium incarnatum* L.), and canola (*Brassica napus* L); however, many producers and extension services support other plant compositions. In early spring, the cover crops will receive chemical burndown with the new crop established with a no-till grain drill/planter into the existing cover crop residue.

USA has established the Mississippi River Basin Healthy Watersheds initiative across 13 USA states [21] to limit the Mississippi River's nutrient and sediment loads. The initiative supports direct payments to agriculture producers to establish erosion and nutrient migration mitigation, primarily through the Environmental Quality Incentives Program (EQIP) and the Agriculture Conservation Easement Program (ACEP). Nutrient reduction strategies are tailored to individual states. Wetland restoration is a key and central provision wherein marginal land is returned to a wetland status.

Southeast Missouri State University and the United States Department of Agriculture—Natural Resources Conservation Service have partnered to address nutrient transport from production agriculture. The development of Edge of Field Technologies is gaining producer acceptance and has witnessed the establishment of denitrification bioreactors to intercept tile drainage effluent to render the effluent comparatively free of NO_3-N. From 2015 to the present, the denitrification bioreactor at the David M. Barton Agriculture Research Center effectively reduced nitrate-N concentrations from between 10 and 100 mg L^{-1} NO_3-N to less than 10 mg L^{-1} NO_3-N [23]. Currently, Southeast Missouri State University and the United States Department of Agriculture—Agriculture Research Service has been active in pumping nitrate and phosphate bearing tile drainage effluent into off season water retention basins to reapply the water as an irrigated source during the growing season. The goal is to reduce aquifer depletion. This research is also investigating whether the stored off-season water may be passed through a denitrification bioreactor and then returned to the aquifer, thus limiting aquifer overdraft with high quality water replacement. Rice production is an important crop in the study area. Recently, arsenic uptake has become an issue. Aide et al. [24–26] investigated different irrigation practices and determined that furrow irrigation would provide similar yields, substantial limit transference of arsenic to paddy rice and reduce water application rates and aquifer overdraft.

Observed and perceived carbon-cycle changes attributed to the wetland conversion project to the atmosphere-plant-soil continuum include: (i) wetland forest vegetation replaced by annual monocot and dicot agricultural plantings resulting in reduced carbon sequestration, (ii) carbon loss because of grain harvesting and because of enhanced soil oxidation by the combined effects of land drainage and tillage, and (iii) increased soil temperatures [1]. Current technologies practices recently implemented to favor restoring soil carbon levels include: (i) improved residue management and the conversion to reduced tillage practices, (ii) off-season cover crop establishment, and (iii) restricted (controlled) drainage technologies and winter irrigation to preserve organic soil carbon. Winter irrigation also provides over-wintering nesting sites for migratory water fowl.

14. Conclusion

A large-scale agriculture region in Missouri was converted from its wetland status to cropland in the 1920s. The loss of hardwood forest and associated wildlife habitat was profound. At the time of the wetland conversion, the benefits of wetland ecosystems were both not understood or appreciated. Approximately 100 years later, we realize the need to reinstate agriculture practices that restore soil health and water quality that the wetland ecosystem provided. We are progressing with best management practices that improve soil carbon replacement, soil structure repair, improving microbial diversity, and appropriate nutrient flux. Plant diversity is still impaired, resting with agriculture monocultures. Wildlife restoration is a far-future goal and flood control and restoring the natural river flow are still critical areas for improvement.

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Chapter 10

Assessing the Drivers of Wetland Changes in Areas Associated with Wildlife-Based Tourism Activities in Zimbabwe

Thomas Marambanyika and Mbulisi Sibanda

Additional information is available at the end of the chapter

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Abstract

The study assesses wetland land cover changes associated with high wildlife densities and tourism activities in Dete vlei, located in Sikumi protected forest, adjacent to Hwange National Park, Zimbabwe. The vlei is used for photographic safaris and is associated with high number of tourists visiting the wetland to see a variety of wildlife species congregated in it. On-screen digitization and analysis of SPOT images for the period of 1984–2013 was used to determine land cover changes in the wetland. Field data were collected through observations, measurements and semi-structured interviews with key informants. The results of the study showed that the spatial extent of bare areas increased in the lower section of the vlei after the establishment of salt licks and watering points meant to attract many wild animals during the dry season. In contrast, wetland conditions have been expanding in the upper section of the wetland without artificial salt licks and watering points. Tourists' footpaths, road culverts, unplanned vehicles' roads, to mention a few, contribute to erosional features evident in the wetland. The study recommends the introduction of wildlife-based tourism management strategies in seasonal wetlands to minimise degradation and possibly loss of wetlands.

Keywords: land cover changes, wildlife population, photographic safaris, wetland ecosystem, tourism

1. Introduction

Wetlands cover 1.8% of Zimbabwe's total surface area [1]. The most common type of wetlands in Zimbabwe are vleis also known as dambos [2], described as seasonally waterlogged valleys or depressions with herbaceous vegetation, mainly grasses and sedges, and devoid of trees [3]. About 60% of wetlands are found in communal and resettlement areas [1], and are predominantly used for cultivation and livestock grazing [4]. Several researches in Zimbabwe have been focusing on communal and resettlement areas to understand the impact of the aforementioned agricultural practices on various wetland ecosystem components such as vegetation, hydrology, geomorphology, soils and water quality [5, 6].

Estimates show that more than 50% of the original wetlands have been lost world over [7]. In Zimbabwe, estimates show that wetlands declined by almost 50% over the past three decades. In the 1980s, wetlands covered 3.6% of the total country area [3] compared to 1.8% in 2015 [1]. Despite previous research on severe wetland degradation in communal, resettlement and urban areas of Zimbabwe [2, 8], there is a dearth of information on how wetlands located in demarcated or state-protected forests (which accounts for part of 40% of wetlands in Zimbabwe) are affected by the existing land uses that are different to those of well-studied communal and resettlement areas, dominated by agriculture. Lack of information on wetlands condition may compromise effective management of these ecosystems in protected forest areas.

In Zimbabwe, demarcated forests are primarily established to manage catchment areas located on fragile Kalahari soils [9]. These forests are managed by a statutory body, the Forest Commission. The major focus of this statutory body is to ensure protection of the forests; hence, there is no deliberate policy to manage wetlands found in the demarcated forests. Wetland ecosystems in protected forests are managed as part of the forest ecosystem, with the primary objective being to protect the forest. However, these wetlands have different human threats to that of forests, a situation that may result in unnoticed wetland degradation and loss. Therefore, there is need to understand the ecological as well as the geomorphological conditions of wetlands in protected areas in light of the presence of potential degrading agents such as high number of tourists and wildlife densities. Some studies have shown that wetland degradation in game reserves is possible, although the rate and causes may vary spatially [10–13].

Dete vlei is primarily used for photographic safaris since it is adjacent to Zimbabwe's largest wildlife sanctuary, Hwange National Park. As a result, different wildlife species graze and drink water in the vlei during the dry season. Wetlands are known to provide forage for herbivores in the African savannah ecosystem over the dry season and during droughts [11, 14]. However, lack of wildlife management within the carrying capacity can lead to high grazing pressure and ultimately wetland degradation [11, 14, 15]. On the other hand, the population of wild herbivores may be threatened by widespread degradation of wetlands [11, 12]; hence need to explore ways of sustainably managing wetlands with different drivers of change.

Meanwhile, salt licks and watering points were established to attract wildlife for game viewing in Dete vlei. The importance of salt licks and watering points as attractants for game viewing has been studied [10, 16]. However, the link between wildlife-based tourism activities and wetland conditions has not been well studied in Zimbabwe regardless of the fact that wildlife pressure is known to have the most damaging outcomes to the world's natural environment, including wetlands [17]. This study, therefore, assesses the potential impact of wild animals controlled by watering points and salt licks as well as the associated tourism activities on Dete vlei's cover during the dry season when more wildlife is attracted to the area. This study aims to provide baseline information that can be used to manage wetlands mainly used for wildlife-based tourism activities in protected areas.

2. Material and methods

2.1. Study area

Sikumi forest has several depressions, with Dete and Zingeni vleis forming the main drainage system of the forest. The study was carried out in Dete vlei found in Sikumi forest (27°10′E; 18°45′S), located in Hwange district of Zimbabwe (**Figure 1**). Sikumi forest is a demarcated forest area that occupies about 55,700 ha [18]. Dete vlei occupies about 903.1 ha, that is, approximately 1.6% of the total forest area. The forest shares boundaries with communal areas, large commercial farms and Hwange National Park (**Figure 1**).

Rainfall in the area is low, variable and unpredictable. The average rainfall for the past 5 years is 500 mm [9]. The rain season normally stretches from October to April. The average minimum

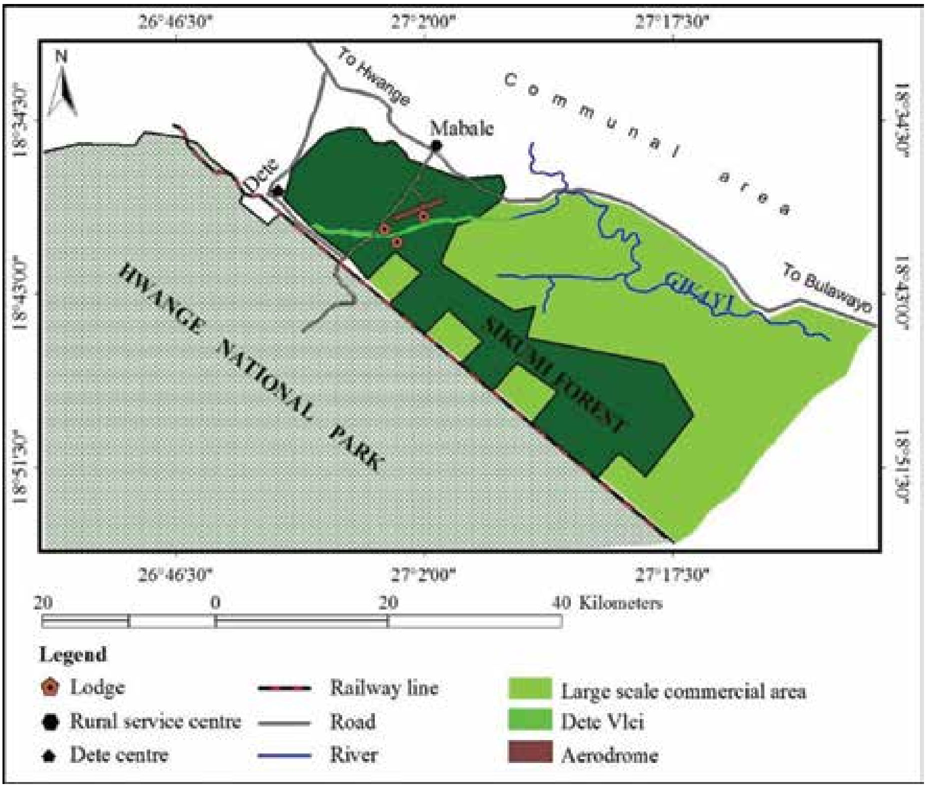

Figure 1. Map showing the location of Dete vlei in Sikumi Forest, Hwange district of Zimbabwe.

and maximum temperatures are 13 and 29°C, respectively, with occasional frost experienced in the depressions [9]. The forest reserve provides commercial timber and wildlife. The dominant soil type is the Kalahari sand associated with the endemic dominant *Baikiaea* genus tree species [18]. The depressions and gentle areas in the forest area are associated with pale sands.

The common grass species in the vlei are *Aristida*, *Sporobolus*, *Eragrostis*, *Pogonarrhia, Perotis and Hyperrhenia* as well as sedges such as *Cyperus*. Due to the forest's proximity to Hwange National Park, the vlei has abundant and wide diversity of game. The common wildlife during the dry season includes elephants, buffaloes and different type of plains game. About 7500 ha of forest land, including the vlei, is leased to private operators for photographic safari business. Therefore, the vlei provides grazing and water to wildlife during the wet and dry seasons and at the same time sustains photographic safari activities. This means the impact of these wildlife-tourism-based business ventures on the vlei needs to be understood in order to come up with appropriate wetland use and management strategies as promoted by [19].

2.2. Hydrogeomorphic characteristics of the vlei

The study compares land cover changes in the upper and lower sections of Dete vlei. The lower section is primarily used for photographic safaris, whereas the upper section has been set aside for uncontrolled wild animal grazing. The whole vlei resembles the features of an unchanneled valley bottom [20]. The vlei has a gentle, longitudinal slope (approximately 1.5%), and no clearly defined stream. The wetland is located at the head waters of a stream that drains into Gwaai River. The predominant source of water for the vlei is direct precipitation, although subsurface inflows can be experienced from the protected forest area that occupies the entire upstream catchment area of the wetland due to the presence of dense vegetation and Kalahari sand soils with a high hydraulic conductivity and infiltration capacity. The use of the wetland for wildlife-based photographic safaris influenced by artificial watering points and salt licks has potential to change wetland cover and possibly degradation of the resource.

2.3. Wetland mapping procedures

Spatial and temporal changes of wetland ecological conditions in relation to erosion and vegetation cover changes were assessed by comparing SPOT satellite imagery for years 1984, 2007 and 2013 to determine the spatial extent of impact of various photographic safari activities. The upper section of the vlei has no artificial watering points and salt licks; hence was compared with the lower section characterised by watering points and salt licks established to influence game viewing. In this study, the upper section of the vlei, across the road to the western side (**Figure 2**), was used as a baseline condition to show an area grazed by wild animals without the influence of watering points and salt licks. Selection of the years for satellite imagery analysis was influenced by availability of high spatial resolution imagery. The image acquired in 1984 image was used as a baseline imagery since it pre-dates the establishment of artificial watering points and salt licks in the entire vlei. The areas that have no grass due to wildlife grazing, salt licking and watering points, which facilitate a high concentration of wild animals on the same spot more frequently, were digitised on screen and classified as bare areas. The spatial extent occupied by bare areas, water and grass within the vlei was

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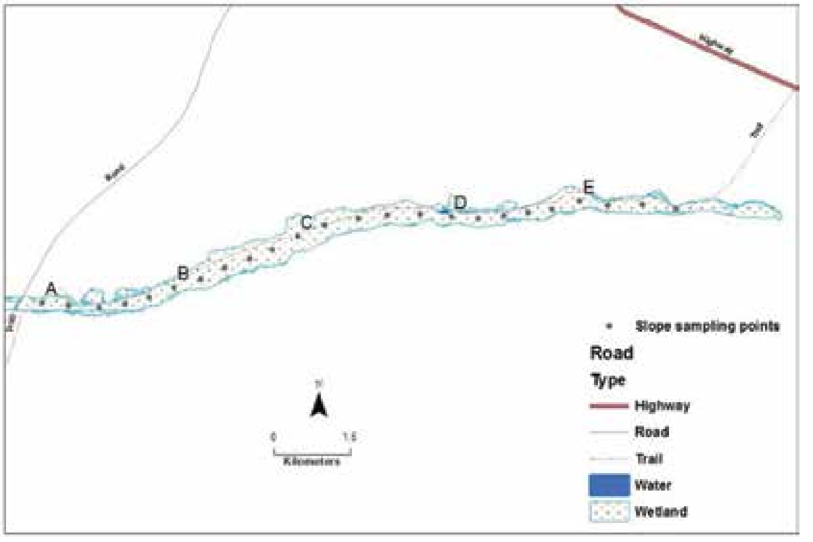

Figure 2. Slope sampling points on the lower section of the wetland.

computed in GIS environment. Image characteristics such as tone, texture, shape, colour and contextual traits as well as locations noted during the field surveys were used in characterising the land cover types within the vlei.

2.4. Field data collection methods

A field survey was carried out during the dry period between November and December 2016. Field observations were carried out to identify evidence of erosion in the form of rills and gullies, sediment deposition and grass cover loss within the vlei. The locations of these erosional sites were noted and used to facilitate their characterisation during the on-screen digitising procedure. Observations were further done, through transect walks, to ascertain if movement of wild animals and associated tourist activities was influencing vegetation cover changes and soil erosion within the vlei. Wildlife paths, pressure on grazing, gravel roads condition, salt licks and watering points were observed, noted and described.

The slopes of the vlei depression were measured at 500 m intervals, from the main road going eastwards (**Figure 2**; **Table 1**). The slope was measured since it influences water erosion, although the rate of erosivity depends on a combination of factors including rainfall amount and intensity, soil type, to mention a few [21]. The identified erosional features such as gullies' slope, depth, width and length were measured. Gully depth and width were measured using a tape measure, whereas the length was measured using a measuring wheel (Bosch GWM 32 Professional). Slope was measured and expressed in percentages.

The diameter and depth of the observed salt licks pits near watering points in different parts of the wetland were also measured to determine how the wetland morphology or landscape was altered by salt licking. Annual rainfall data (1962–2016) and annual mean daily minimum and maximum temperatures data (1962–2010) were obtained from the Meteorological

Point name	X coordinate	Y coordinate
A	26.98	−18.66
B	27.00	−18.66
C	27.02	−18.65
D	27.05	−18.64
E	27.07	−18.64

Table 1. Location of the selected slope sampling points.

Services Department of Zimbabwe. Local climate data were used to assess the possible effect of local climate variability on wetland vegetation condition and geomorphic processes such as erosion and deposition.

Historical information of the wetland's geomorphic condition was obtained from purposively sampled key informants targeted for semi-structured interviews. In this case, a template with open-ended questions was prepared to guide face-to-face discussions. The key informants were selected from organisations that are involved in photographic safari ventures, management of the forest areas located in the catchment area of the vlei or individuals who had knowledge of the area stretching over several decades. The key informants were the Safari Operators, former Forest Commission Divisional Manager for Indigenous Forests, Matabeleland North Forestry Commission Provincial Manager, Sikumi Forester and the Parks and Wildlife Management Authority Ecologist. Records of wildlife population changes and number of tourists were also obtained and reviewed.

2.5. Data analysis

Rainfall and temperature data obtained from the Meteorological Services Department of Zimbabwe used for determining trends were subjected to regression analysis performed in Microsoft Office Excel 2007. Trend analysis was done to determine if there was change in mean annual temperature (minimum and maximum) and annual rainfall totals, since temperature and rainfall amount influences vegetation cover and geomorphic processes such as erosion and deposition. Qualitative data generated through semi-structured interviews (on perceived changes in wildlife numbers, vlei's condition and climate trends) were analysed using thematic analysis method [22]. Wildlife population density was calculated basing on average game counts done by Forestry Commission in 2016 and the vlei size measured in ha.

3. Results

3.1. Local climate trends

Figures 3 and **4** show a graphical representation of annual rainfall totals and temperature. Generally, the vlei area experiences low rainfall and high temperatures. The average annual rainfall total is 544.38 mm. The total annual amount of rainfall has been decreasing between 1962 and

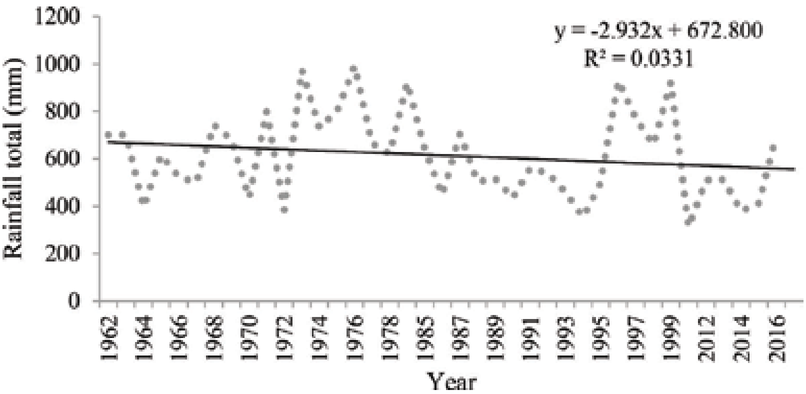

Figure 3. Annual rainfall totals for Dete Vlei, Hwange district (1962–2016).

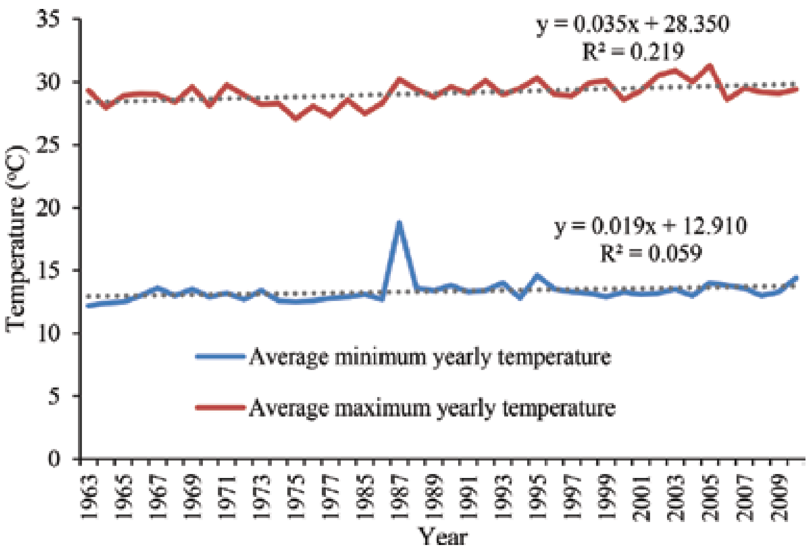

Figure 4. Annual minimum and maximum temperature for Dete Vlei, Hwange district.

2016 ($y = -2.932x + 672.800$; $r^2 = 0.033$). The highest (975.9 mm) and lowest (311.9 mm) amount of rainfall were received in 1973 and 2000, respectively. In contrast, mean maximum yearly temperature ($y = 0.035x + 28.350$; $r^2 = 0.219$) and the mean minimum yearly temperature ($y = 0.019x + 12.910$; $r^2 = 0.059$) have been increasing between 1962 and 2010. The mean maximum yearly temperature is 29.1°C, whereas the mean minimum yearly temperature is 13.3°C (**Figure 4**). The former Forest Commission Divisional Manager for Indigenous Forests attributes the reduction in rainfall to changing climate accompanied by frequent droughts that intensified from the year 2000.

3.2. Wildlife population changes

The common wildlife species found in the vlei are elephants, buffaloes, baboons, sables, impalas, kudu and warthogs (**Figure 5**). Generally, small and large predators account for relatively

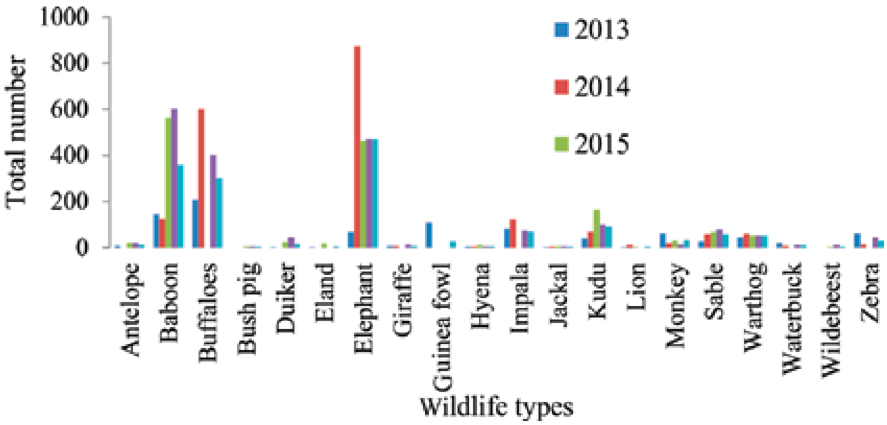

Figure 5. Wildlife species population between 2013 and 2016 (Source: Forestry Commission [9]).

few of the total number of animals found in the vlei as they are significantly outnumbered by herbivores, categorised as grazers, browsers or both. Large mammals such as elephants and buffaloes frequent the wetland for water and/or grazing. The elephant population is the highest, with an average of 469 over the last 4 years (2013–2016), whereas buffaloes have the second highest number of large mammals, with an average of 302.

Key informants indicate that plain game species such as impala, kudu, duiker, waterbuck and eland were also a common feature of the wetland landscape. Although game count statistics obtained from the Forestry Commission show that the population of different wildlife species has been fluctuating over the past years, in contrast, baboon populations have been increasing. The interviewed Ecologist and Forester indicated that the present wildlife population exceeds the carrying capacity of the area as evidenced by grazing pressure in some parts of the wetland. The elephants destroy trees on the edges/fringes of the wetland indicated by dominance of trimmed trees. The pressure on grazing has resulted in bare areas (**Figure 6**).

3.3. Wetland erosion linked to wildlife

Figure 6 shows that the wetland's lower section has erosional features such as developing gullies and is losing vegetation in areas surrounding artificial watering points and salt licks.

Spatio-temporal analysis of land cover shows that there is no bare area in the upper section of the vlei between 1984 and 2013. In contrast, the areas devoid of vegetation as a result of wildlife trampling and erosional features occupy 5% of the overall extent of the lower section of the vlei (**Table 2**). The bare area around artificial watering points and salt licks increased by 58.56% between years 2007 and 2013. They now cover 4.6% of the lower section of the vlei used for photographic safaris. Erosional features such as gullies, not present in the whole wetland during the previous years, occupied about 0.82 ha in the lower section of the wetland in 2013 (**Figure 6**; **Table 2**); a sign that geomorphological disturbances such as erosion were taking place. The overall spatial extent of the upper section of the wetland increased by 41.2%, whereas the lower section with artificial water points and salt licks shrunk by −2.3% between 1984 and 2013.

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Year	Upper	Lower	Total vlei size	Bare area in the lower section of the vlei		Erosional features in the lower section of the vlei	
				Area (ha)	%	Area (ha)	%
1984	316	467.6	783.6	0	0	0	0
2007	348.8	429.7	778.5	13.32	3.1	0	0
2013	446.2	456.9	903.1	21.12	4.6	0.82	0.002

Table 2. Proportion of bare area (ha) and erosional features in the vlei.

Salt licks are evident in the lower section of the vlei and are characterised by several pits and areas devoid of vegetation (**Figure 7**). Other than pumped water, artificial salt licks appear to be attracting many different wildlife species during the dry season. This explains why high numbers of large mammals such as elephants, buffaloes and plains game are found in the lower section of the vlei throughout the year. This situation is in sharp contrast with the upper section of the vlei without salt licks where evidence of bare areas was not noted at all based on the SPOT satellite images (**Figure 6**). On average, salt licks of 3 m diameter and 35 cm depth are found in several parts of the lower wetland section. Field observation results showed that vegetation cover was completely lost in areas as wide as 800 m^2 around the salt licks (**Figure 7**). The deepest salt lick pit was 83 cm, whereas the widest pit was 8 m in diameter. Interviews with key informants revealed that some safari operators apply salt, especially in areas near the artificial water sources, in order to attract more wildlife for game viewing. Salt licking is assumed to have started in

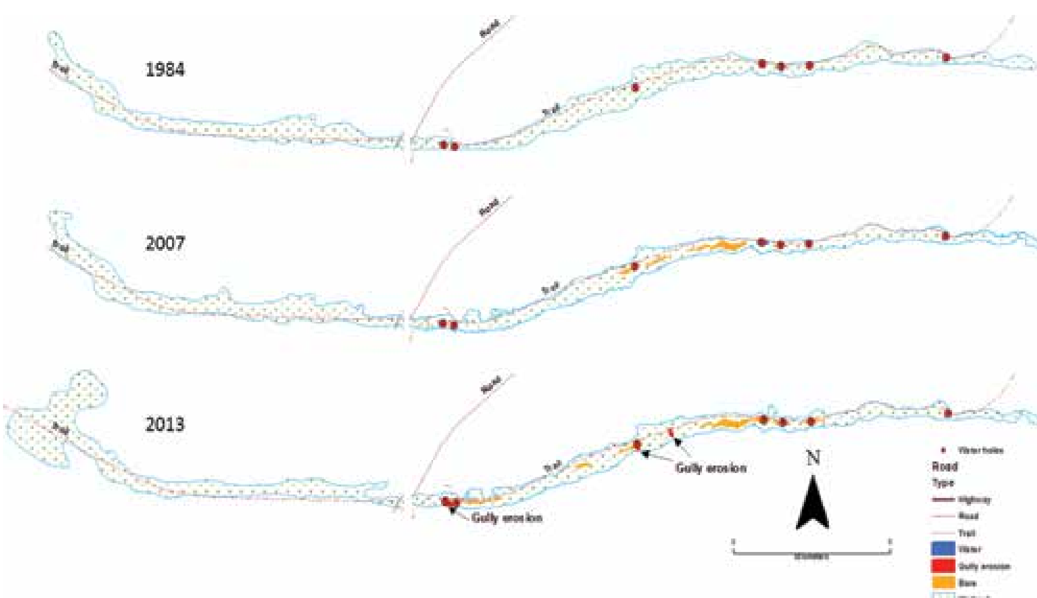

Figure 6. Saptio-temporal changes in erosional features and bare area linked to water holes in Dete vlei (1984–2013).

Figure 7. Section of the vlei altered by wildlife during salt licking.

the mid- to late 1990s in order to control the introduced Presidential Elephants, protected under a presidential decree of 1990 but instead the licks are attracting different wildlife species.

During the dry season when there is water shortage in the area, safari operators pump ground water from the wetland into open waterholes. These artificial watering points replenished by boreholes were established in the late-1980s across the lower section of the wetland, in proximity to lodges (**Figure 3**), to attract more wildlife for game viewing. Trampling is evident within 15 m around watering points as a result of large number of wild herbivores that drink water from these sources.

Due to high movement and frequency of wild animals in the vlei, mainly for water and salt licking, a number of wildlife trails or paths have been formed. Wild animals' trails are possibly facilitating the formation of several rills across the wetland area, especially in the lower section. The wildlife paths facilitate concentrated water flow, hence promoting soil erosion and siltation or sediment delivery into the wetland. Pressure on grazing by wild herbivores is a common phenomenon mainly around salt licks and watering points. Key informant interviewees attributed the grazing pressure to high wildlife populations and the presence of grazers around water points. This is more pronounced during the dry season due to physical water scarcity in the natural pans dotted around the forest area and in the adjacent Hwange national game park.

Meanwhile, elephants have been destroying trees, predominantly *Acacia* and *Terminalia* species along the edge of the wetland, forming a transitional zone between the vlei and the protected forest area. The elephant density is estimated at 0.01/km^2 in the whole Sikumi forest and 0.52/km^2 in the wetland area. This has affected vegetation density as exhibited by broken trees in the areas adjacent to watering points.

3.4. Soil erosion linked to tourist activities

Normally, tourists walk closer to wild animals at watering points to take pictures or films. The habituated elephants (by Allan Elliot since 1974) commonly found in the vlei area are not vicious to humans. Given the fact that each of the lodges receives more than 200 visitors per fortnight during the dry season, some walking paths often used by tourists to get closer to the watering points are gradually developing into rills and gullies and there is evidence of deposition in the vlei as shown in **Figure 8**. Gully development seems to be further influenced by fairly steep gradient, which on average is 4% (**Table 3**). Point D with the steepest slope has a more pronounced gully.

Moreover, two weirs were excavated in the lower section of the wetland in the late 1980s to impound more water for photographic safari activities. Coincidentally, weir construction and boreholes drilling started at the same time when additional lodges were established. However, there is evidence of soil erosion on some of the weirs. Erosion is as a result of overtopping of the weirs during the rainy, especially during the years when high rainfall was received in the area. The eroded sediments are likely to increase sediment yield downstream, which may be severe if high rainfall persists, since rainfall in the area is highly variable (**Figure 8**).

There is a gravel road constructed along the vlei to facilitate easy game drive by tourists around the wetland area. The gravel road has drains which control and discharge runoff into the vlei at certain points. Channelized flow is discharged into the wetland resulting in gully erosion, especially where vehicles use unplanned drive ways to cross the vlei during the dry season. Unplanned drive ways in the vlei result in vegetation loss and defined channels

Figure 8. Gully development along tourists' foot paths.

Slope point	Slope angle
Point A	2.4
Point B	4.1
Point C	3.9
Point D	5
Point E	3.8
Average	3.8

Table 3. Slope gradient at selected interval along the vlei, in percentages.

for concentrated flow, a condition that enhances geomorphic process such as water erosion linked to surface runoff. The most pronounced developing gully is on average 3 m wide, 24 cm deep and 45 m long.

There are also unplanned roads that are used for game drive by tourists into the forest area surrounding the vlei. Some of the roads are developed following fairly steep gradients (on average 6% slopes) on the margins of forest area that forms the catchment area of the vlei into the wetland. Despite the fact that the predominant soil type is Kalahari sand (with high infiltration capacity), there is evidence of soil erosion on these roads as runoff is enhanced by the steep slope and channelized flow. Some of the sediment ultimately gets into the vlei, a situation likely to alter the ecological characteristics of the wetland due to enhanced sediment delivery.

4. Discussion

Results of this study indicated negative changes in the area occupied by the lower section of the wetland by a magnitude of 38 ha between 1984 and 2007 and by about 11 ha between 1984 and 2013. Meanwhile, the upper section incurred significant increase of about 32 ha between 1984 and 2007 and 130 ha between 1984 and 2013. The significant decreases in the area occupied by the lower section of the wetland could be attributed to the high concentrations of wild animals in the salt pans and water points as well as the high intensities of anthropogenic activities. On the other hand, the limited number of wildlife concentrations in the upper section could explain the intact and increases in the areal extent of the wetland covered by grass. Our results are supported by those of a study carried out in South Africa which also indicated that the creation of artificial water points in Kruger national park on the upland section of the park caused a high concentration of wild animals [23].

A gravel road stretching along the northern fringe of the vlei is likely to disrupt normal sediment mobility and deposition in the wetland area, a situation also observed by [23]. Erosion in the wetland is initiated from concentrated flow starting from culverts established to divert runoff from the road. This has resulted in rills and gullies in some parts of the lower wetland section where water is discharged into unplanned roads (**Figure 8**). Therefore, the effect of road construction through culverts on the vlei's erosion is evident. This result concurs with observations that roads tend to disrupts wetlands functioning through erosion and sedimentation [24].

Salt licking was observed as one of the main wildlife-related causes of wetland landscape alteration as indicated by existence of open pits surrounded by bare areas (**Figure 2**). This concurs with previous studies that salt lick areas are mostly devoid of vegetation as a result of heavy trampling from large herbivore which including those elephants and sables [10, 24]. This explains why in countries like Malaysia salt licks and land in its immediate vicinity are protected against disturbance of soil and vegetation [10]. Loss of vegetation cover generally exposes soil to erosion by either water or wind [25]. Despite the fact that the total amount of rainfall received per season has been declining over the past four decades as shown by linear regression results ($y = -2.932x + 672.8$; $r^2 = 0.033$), the occasional high rainfall occurrences noted in this study could also be attributed to excessive erosion activities. Arid conditions worsened by increasing minimum and maximum temperature also may expose the bare areas around salt licks to wind erosion. Therefore, the wetland is susceptible to both water and wind erosion given the changing climate in the area. Wind speed in the vlei may be high since the depression is predominantly grassland surrounded by forests, which could make it a trough for wind passage.

Wildlife grazing is also influencing the alteration of the wetland's landscape. High grazing intensities by plain game species were mostly observed around watering points and were almost devoid of vegetation. The pressure on grazing has the potential to enhance soil erosion by exposing the soil facilitating surface runoff. Some previous studies revealed that the effect of cattle grazing around watering points is low [15] whereas that of wildlife was found to be high, characterised by absence of vegetation [10, 16, 17]. This explains why large-scale commercial farmers which occupied vleis in the early days limited the use of the vlei to late dry season grazing to avoid heavy grazing which resulted in erosion throughout the year [2]. In the case of the lower section of Dete vlei, pressure of wild grazing is high since grazing is continuous during the dry season while wild animals are attracted by watering points and salt licks to a central point. Therefore, strategies should be considered to regulate grazing around watering points and salt licks in order to mitigate soil erosion considering that bare conditions were not a common phenomenon in the upper section of the vlei which has no watering points and salt licks.

Wildlife trampling which is well pronounced within a 15 m radius of watering points also results in top soil loosening and loss of vegetation cover, making the soil susceptible to erosion and possibly siltation of the existing water points. This finding concurs with [10] who acknowledged that the visibility of wild herbivores trampling around watering points results in vegetation cover reduction. According to [13], the continuous trampling by wild animals in a forage land accelerates the reduction of vegetation cover and ultimately exposes the soil to erosion agents.

Wildlife vegetation destruction, especially *Acacia* trees by elephants, exposes soil to water erosion along the wetland fringes, facilitating increased sediment input into the wetland given its fairly steep gradient (**Table 2**). The effect of high elephant densities on vegetation and the environment in general is well documented [26]. This was complemented by findings by [27] that high elephant population results in severe environmental damage, loss of biodiversity and increased competition for scarce resources. In the case of Dete vlei, the elephant density is estimated at 0.01/km^2 in the whole of Sikumi forest (and 0.52/km^2 in the wetland area). According to the Ecologist and Forester, the current elephant population is beyond the optimum carrying capacity of the area. Considering that there are various wildlife species frequenting the vlei as well as a result of the presence of water during the dry season (**Figure 4**), the ecological carrying capacity of the vlei could have been severely exceeded as different wild animals compete for grazing; hence vegetation loss and the potential of soil erosion being accelerated in the vicinity of watering holes.

Furthermore, artificial salt licks attract more wildlife for photographic safaris, resulting in more tourists visiting the area at the detriment of the wetland. In this case, safari operators, by applying salt, are more concerned with the economic gains associated with the influx of game viewers at the expense of the vlei's ecological condition which is the basis for the existence of these economic activities. The tourists have also been contributing to soil erosion as evidenced by erosional features such as rills and gullies developing along footpaths around watering points in the lower section of the wetland. Therefore, instead of simultaneously harmonising environmental and economic considerations to achieve wise use of the wetland, these two objectives are treated as discrete entities by safari operators, a situation with potential to cause vlei degradation and loss and ultimately loss of business in the long run for the safari operators.

Unplanned and poorly designed drive ways have potential to worsen the rate of erosion despite the reduction in rainfall amounts received in the area. This is more evident where some roads from the catchment surrounding the vlei were established following fairly steep gradients, a situation likely to accelerate the rate of soil erosion due to the effect of concentrated flow and possibly increased sediment yield. This may suggest that vehicle movements if not well planned and monitored have great potential to cause soil erosion in the wetland and tourist areas.

Although the gullies noted in this study are relatively small (a depth of 24 cm) when compared with those reported in other studies [2] which exceed 50 cm in depth, they are still of major concern. This implies that intervention strategies to mitigate soil erosion should be considered so that the vlei does not develop big gullies as those noted by Whitlow in the communal areas of Zimbabwe. These gullies are a growing threat to the socio-economic benefits linked to wetland utilisation. This is grounded on the findings of this study which illustrated that there is a temporal increase in the spatial extent of bare areas in the lower section of the wetland and overall reduction in the wetland size. In contrast, the upper section of the wetland without watering points and salt licks is increasing in size (**Table 1**). This suggests that if photographic safari activities, watering points and salt licks, in particular, are not well regulated, degradation of the wetland is likely to be more pronounced.

5. Conclusions

The study assessed wetland land cover changes associated with high wildlife densities and tourism activities, using Dete vlei in Hwange district as a case study. Results show that bare conditions have been increasing around watering points and salt licks resulting in the reduction in wetland conditions of the lower section. In contrast, the upper section remains without bare cover and the wetland conditions are expanding. Based on these findings, we conclude that photographic safari activities such as wild animals grazing and trampling around artificial salt licks and watering points, vehicle movements and tourists paths are contributing to vegetation loss and erosional features. Therefore, there is need for deliberate policy and strategy to control wetland degradation in protected used for photographic safaris. The strategy should involve all stakeholders (private players and public institutions) in order to achieve sustainable wetland-based photographic safari business.

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Conflict of interest

No conflict to declare.

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Wetlands include mangroves, peatlands and marshes, rivers and lakes, deltas, floodplains, rice fields, and even coral reefs. It is known that wetlands are ecologically sensitive systems and the most vulnerable of habitats. Anthropogenic activities (urbanization, water uses, land cover changes, industrial activity, pollution, climatic change, etc.) have direct and indirect effects on wetlands. The evaluation of wetlands with a multidisciplinary perspective in environmental sciences and social sciences provides efficient results. Each chapter takes a crucial look at different approaches to the solution by analyzing wetland problems in the laboratory or in the field and collecting data. The purpose of this book is to help researchers, scientists, and decision-makers utilize a methodology appropriate for a specific problem.

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