

WATERBIRDS AS BIOINDICATORS OF WETLAND QUALITY:

CASE STUDY OF THE MUNI-POMADZE RAMSAR SITE.

GHANA

By

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DECLARATION

I hereby declare that this thesis is entirely the result of research work undertaken under the supervision of Prof. Yaa Ntiama-Baidu, Prof. Chris Gordon, and Dr. E.H. Owusu of the Department of Animal Biology and Conservation Science, University of Ghana. It has neither been submitted wholly nor partially to any other University for a degree. With the exception of reference to other people's work that I have duly acknowledged, all other experimental work described in this thesis was carried out by me. All forms of assistance received while working on this thesis have also been duly acknowledged.

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DEDICATION

This work is dedicated to my parents, Mr Abdourahamane Illiassou and Ms Abdourahamane Amina Moussa for their unlimited support throughout my studies. I cannot thank them enough, then just to say “JAZAK ALLAHU KHAYRAN”.

I also dedicate this work to Mrs Diakite Halima for her encouragement and advice during my studies in Ghana

May the Almighty ALLAH bless all of you and reward you with Jannat.



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ABSTRACT

The Muni-Pomadze Ramsar Site is the fifth most important coastal wetland in Ghana. It was designated as a Ramsar Site in 1992, based on its internationally important tern populations and the total population of waterbirds supported by the site. An assessment of the ecological character of the MPRS, habitat use by waterbirds, and the value of waterbirds as indicators of wetland health was carried out from September 2009 to April 2010. This involved the use of satellite imageries in order to assess changes in the habitat, monthly water quality assessment, benthic macroinvertebrates sampling, and waterbird monitoring. The habitat analysis showed that the size of the Muni Lagoon decreased by 50% from 1990 to 2007; the extent of herbaceous shrubs decreased by 99%; and the infrastructural development in the area quadrupled. The water quality analysis showed that Muni remains a shallow, hypersaline lagoon even though the salinity ranges recorded were lower compared to previous years as a result of the opening of the connection between the lagoon and the sea. A total of 1,723 individuals belonging to 10 groups of macrobenthic invertebrates were recorded. Of these, polychaetes- Annelida constituted 90.71%, bivalves- Mollusca 7.84% and the crustaceans-Arthropoda 1.45%. Muni Lagoon is clearly a stressed environment as its benthic population was dominated by a single taxon (Capitellid worms) and influenced by the fluctuating high temperature and salinity. Thirty-three waterbird species belonging to eight families were recorded, with a cumulative total sighting of 14,342 individual birds, of which waders represented 67%, terns 28% and herons 5%. The results show decreases in the species as well as populations of waterbirds when compared to the counts recorded in 1986-1998. Changes have occurred also in the waterbird community structure of the MPRS, with the Ringed Plover *Charidrius hiaticula* now being the most abundant wader species instead of the Curlew Sandpiper

Calidris ferruginea which was the most abundant in the 1986-1998 counts. The site however still supports nationally important populations of five wader species. Waterbird distribution and activity in the Muni lagoon varied not only according to feeding groups and guilds but also according to time and site. Using the Bray-Curtis similarity index, the Muni Lagoon can be categorised into three sections based on the waterbird distribution. The MPRS remains an important roosting site for the tern, which use mostly Sites C (Mid-Lagoon portion) and E (Seaward section), and foraging ground (mudflat areas A and D) for the non-breeding waders. Waders roosted mostly at Site SP (Salt Ponds). Water depth was the key factor influencing prey accessibility and availability for waterbirds and biomass of polychaetes was the only factor determining the distribution of feeding waders. The highest benthic diversity was recorded at Site E due to recruitment from the sea, while that of waterbird was recorded at Site B due to the heterogeneity of the habitat of that site. Despite the habitat losses and the general deterioration of the health of the MPRS, the site continues to support significant numbers of waterbirds, although the populations have gone down. Thus, based on the finding of this research, it may be concluded that waterbirds are not early indicators of wetland quality. Depending on what has been lost or gained, a change in the environmental health of a wetland may not result in changes in the waterbird community, such that by the time the birds stop using the altered site, the site may be degraded beyond redemption. It is suggested therefore that bird monitoring at the MPRS continues over a longer period to enable the re-assessment of its international importance. The management of the site should be reinforced and accompanied with a habitat restoration program. Changes in the waterbird community structure should also be further investigated.

CHAPTER I

INTRODUCTION

1.1 Background

Waterbirds are a key biodiversity group that depend ecologically on wetlands for feeding, roosting, and nesting. The success of waterbirds in relation to these activities, and hence their survival, is determined by the quality of wetlands they encounter on their migratory routes.

The key factors that influence habitat selection of migratory waterbirds during the non-breeding season are availability of food, safe roosting sites and the extent of disturbance (Piersma, 1994; Ntiamoa-Baidu *et al.*, 1998). With regard to foraging areas, the spatio-temporal distribution of waterbirds is largely determined by prey distribution and accessibility. These in turn, are strongly influenced by sediment type and exposition time in the case of probing waders, as well as water depth within the wetland (Moreira, 1995). The distribution and abundance in space and time of aquatic invertebrates, the primary food sources for many shorebirds, are affected by a wide range of biological, physico-chemical and hydrological variables such as water temperature variation, water quality, contaminants, and habitat alteration (Skagen and Oman, 1996; Pothoven *et al.*, 2001).

The survival of any waterbird species is highly dependent upon the quality of the wetland it inhabits during its lifetime, or at least during its annual migration cycle. The breeding success of some waterbird species is positively correlated with the abundance of resources available at the staging or stopover sites during the non-breeding period. Of the numerous threats confronting waterbirds and wetlands, studies have shown that

the greatest are habitat and biodiversity loss (Hill *et al.*, 1997; International Wader Study Group, 2003; Zöckler *et al.*, 2003).

Wetlands have been defined 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 metres”(www.ramsar.org). Globally, wetlands cover an area of 1.2 million Km² (Millennium Ecosystem Assessment, 2005), and can be classified into three main groups: (i) coastal/marine, (ii) inland and (iii) man-made. They may also be classified as (i) coral reefs, (ii) rocky marine shores, (iii) estuarine waters, (iv) intertidal mudflats, (v) intertidal marshes, (vi) intertidal forested wetlands, (vii) coastal lagoons, (viii) permanent inland deltas, (ix) freshwater lakes, (x) alpine, (xi) tundra, (xii) swamp forest wetlands, (xiii) oases, (xiv) aquaculture- farm pond, (xv) dams, (xvi) salt exploitation site, (xvii) waste water treatment areas, etc. (www.wetland.org).

Although wetlands occupy only 4% of the earth's ice-free land surface (Prigent, 2001), they are among the most valuable ecosystems for human beings as well as flora and fauna (Millennium Ecosystem Assessment, 2005). Their importance is thus not limited to direct socio-economic benefit to people through resource utilisation such as fishing and ecotourism, but also include vital ecosystem functions such as facilitating ground water recharge, nutrient retention, flood and erosion control, and sediment filtration (Simonit *et al.*, 2005; Ministry of Land, Forestry and Mines, 2007). Wetlands also represent biodiversity conservation hotspots and provide habitat not only for species that inhabit them permanently, but also for long-distance migrant birds, critical habitats during their non-breeding seasons (Ntiemoa-Baidu, 1991; Piersma and Ntiemoa-Baidu, 1995; Thompson and Polet, 2000; van de Kam *et al.*, 2004; Gbogbo, 2007a).

Wetlands are among the most threatened ecosystems in the world, being subjected to unsustainable resource utilisation, vegetation disturbance, habitat degradation, sedimentation, drainage and pollution (Millennium Ecosystem Assessment, 2005). These, mostly anthropogenic habitat disturbance factors have negative implications for the livelihood of local communities in particular (Turner *et al.*, 1990).

In Ghana, there is increasing evidence that the rate of environmental degradation has increased in recent years (Gyasi *et al.*, 1995; Ryan and Ntiemoa-Baidu, 1998) leading to habitat alteration and its associated effects, such as loss of biodiversity and change in the ecological character of wetlands, among others. Historically, wetlands have been considered as “wastelands” and therefore have been subjected to extensive degradation from (i) draining for agricultural and irrigation purposes, (ii) livestock ranching, (iii) solid waste disposal and (iv) dumping grounds for sewage and other pollutants emanating from households and industries (Ryan and Ntiemoa-Baidu, 2000). For example, in Ghana, increasing salt-winning has had a great impact on wetlands by reducing the available feeding areas for birds and limiting the abundance and diversity of macroinvertebrates that form the bulk of the prey for waterbirds (Gbogbo, 2007a).

Ghana's 550-km coastline includes over 90 lagoons (Mensah, 1979; Gordon, 1987; Gordon *et al.*, 1998; Koranteng *et al.*, 2000; Gbogbo, 2007b), many of which constitute key habitats for diverse and rare avifauna including globally important concentrations of migrants and restricted range species such as the Roseate Tern (*Sterna dougalii*). On the basis of the internationally important waterbird populations they support, and other ecological, socio-cultural and economic values that these coastal wetlands represent, the Government of Ghana as a contracting party to the Ramsar Convention, has designated six wetlands as “Ramsar sites” (i.e., wetlands of

international importance, especially for waterfowls). These include one inland wetland, Owabi Wildlife Sanctuary, in the Ashanti region, and five coastal wetlands, namely, Muni-Pomadze in the Central Region, Densu Delta, Sakumo and Songor in the Greater-Accra Region, and Keta in the Volta Region (Ntiama-Baidu and Gordon, 1991; Gordon, 1996; Armah *et al.*, 2005; Ministry of Lands, Forestry and Mines, 2007).

Since the designation of these Ramsar Sites, Ghana has invested substantial efforts and resources into wetland conservation. The Ghana Coastal Wetlands Management Strategy document (Ntiama-Baidu and Gordon, 1991) essentially outlines the importance of, and threats to Ghana's coastal wetlands, and provides recommendations on strategies to maintain the ecological integrity of these vulnerable biotopes. In 1994, the Coastal Wetlands Management Project (CWMP) was commissioned with the objective of restoring and maintaining the ecological integrity of Ghana's five coastal wetlands. This was followed by the National Wetlands Conservation Strategy and Action Plan in 1999 (revised in 2007), initiated with the ultimate goal of conservation and sustainable use of Ghana's wetland resources through documenting strategies for incorporating wetland management into activities of government, communities and individuals within the broader context of environmental management (Ministry of Land, Forestry and Mines, 1999; 2007).

In order to achieve any conservation and sustainable utilisation goal, and to understand the functioning of wetland ecosystems, there is a need for ecological monitoring and surveillance programmes which often involve the use of bioindicators or biomonitors to complement physical monitoring. Different zoological taxa, either single or combined, can be used to monitor environmental alterations, and can also be classified

as positive and negative indicators based on the information they provide. Waterbirds have been used as bioindicators in assessing the environmental status of wetlands with regard to pollution, habitat degradation, vegetation disturbance, water quality and resource productivity (BirdLife International, 2004). For example, changes in the ranges of migratory waterbirds may result from changes in habitat quality and mortality rates (Ogden *et al.*, 1987). Newton (2004) also stated that a decline in Redshank (*Tringa totanus*) populations may give indications of nesting site destruction through land drainage. Hence, for a biomonitor to be useful, it must be sensitive to changes in the variable for which it is a proxy measure, and must be predictable and easy to measure (Furness and Greenwood, 1993). Continuous long-term monitoring of the populations of bird species can therefore provide useful data in describing the implications of anthropogenic activities for habitat condition and biodiversity (Owusu *et al.*, 2002).

Waterbirds are of interest also to scientists and/or conservationists as indicators of wetland status and as parameters of restoration success and regional biodiversity. Although waterbird populations may not provide ideal early warning signals of environmental deterioration, changes in their diversity and composition in a given area over a period of time, may be indicative of the state of the habitat (Temple and Wiens, 1989). This is because generally the impact of habitat alteration on birdlife depends on what has been lost and what has been substituted in the area involved, and on the spatial relationships that exist in the new landscape, including the extent of connectivity and fragmentation. Pollutants for example, can affect birds directly through lethal and sub-lethal stress, or indirectly through habitat alteration (Furness and Greenwood, 1993). Birds accumulate heavy metals through continuous preening of contaminated materials off their feathers and by taking in food or water containing

industrial effluents, polluted agricultural run-off, as well as untreated domestic and industrial waste. When present in high concentrations, these contaminants in addition to organochlorines, can affect the breeding success of waterbirds, particularly those at higher trophic levels (i.e., piscivores, raptors, etc.) (Furness and Greenwood, 1993).

Wetland quality, from the perspective of waterbirds, can be described *inter-alia*, as the water quality, food availability, as well as roosting and nesting sites, which determine the welfare of a bird or bird community assemblages. Wetland quality is the most holistic factor that influences waterbird communities. It is determined by the ecological character of the wetland in question which is a combination of the ecosystem components, processes and benefits that characterise the wetland at a given point in time. Wetland quality in terms of its fauna, flora and hydrology also determines its socio-economic and cultural importance to humans directly and indirectly, while human activities and utilisation of the wetland could also alter the ecological character of the wetland (www.ramsar.org).

The ecological character of a wetland could be defined as “the sum of biological, physical, and chemical parametric components and their interactions which maintain the wetland and its products, functions and attributes” (www.ramsar.org). Any alteration of these attributes can either enhance or diminish the value of a site and therefore affect not only the habitat, and hence its value for humans, but also the survival of the fauna, especially the waterbirds that depend on the wetland. Changes in faunal diversity and use of the wetland by waterbirds could therefore provide an insight into the conditions of the wetland.

1.2 Justification

The Muni-Pomadze Ramsar Site (MPRS) is an important wintering and staging area for migratory and resident waterbirds with total population estimated at 20,000 (Ntiama-Baidu *et al.*, 2001). The site has been classified as a “Globally Important Bird Area” (Ntiama-Baidu *et al.*, 2000a). A total of 48 species of waterbirds (out of 88 species found on the Ghanaian coast) have been recorded on the site, comprising 29 species of waders, eight species of terns, two species of gulls, seven species of herons and egrets (ardeids), and one species each of duck and cormorant (Grimes, 1987; Ntiama-Baidu *et al.*, 2000a). The MPRS also supports at least 21 species of mammals (Ryan and Attuquayefio, 2000), 31 species of herpetofauna (13 amphibians and 19 reptiles) (Raxworthy and Attuquayefio, 2000) and 114 species of terrestrial birds (Ntiama-Baidu *et al.*, 2000b).

The MPRS faces a number of threats, particularly unsustainable resource exploitation including fishing, hunting, and fuel wood harvesting, overgrazing by cattle, and improper farming practices such as indiscriminate slash burn agriculture that lead to devastating bushfires and erosion. Sanitation and waste disposal facilities of surrounding local communities are relatively poor, resulting in high levels of pollutants in the lagoon. The degradation of the wetland has largely been attributed to unsustainable human activities and expanding dry land coverage (Amatekpor, 1994; Ryan and Ntiama-Baidu, 1998). However, the use of the site by waterbirds was found to have increased by 400% between 1986 and 1998 (Ntiama-Baidu *et al.*, 2000a). This increase was attributed to an increase in prey availability induced by inflow of sea water to the lagoon in 1994, and possible degradation of other coastal wetlands making the Muni lagoon more suitable for birds although there was no data to suggest same or

otherwise in the other coastal wetlands (Ntiemoa- Baidu, *et al.*, 2000a). It is therefore necessary to understand the extent of bird-habitat interactions and the consequences of habitat changes on waterbird populations. The need for such ecological study becomes more urgent when considered against the background of general global declines in the populations of Palaearctic-African migratory shorebirds (International Wader Study Group, 2003).

This research was intended as a continuation of 1986-1998 baseline study and the monitoring of the use of the MPRS by waterbirds and to test whether or not waterbirds provide a good indicator of the health of the Muni-Pomadze wetland ecosystem.

1.3 Hypotheses

- Waterbird populations and their use of the wetland habitat provide good indication of the quality of the Muni Pomadze Ramsar Site.
- The ecological character of the Muni Pomadze Ramsar Site has not altered in the past 15 years (1992-2007) since its designation as a Ramsar site.

1.4 Objectives

The main objective of the study was to assess the current ecological status of the Muni Pomadze wetland, the role of waterbird communities in the wetland ecosystem and the value of waterbirds as bioindicators of wetland ecosystem health. The specific objectives were to:

- determine the ecological status of the Muni Pomadze wetland with focus on physical water parameters, macroinvertebrates and waterbirds.
- compare the species diversity of waterbirds and benthic macroinvertebrates within different sections of the Muni lagoon.
- determine the spatio-temporal patterns of wetland use by individual waterbird groups for roosting and feeding.
- investigate the relationships between physicochemical water quality parameters (temperature, salinity, dissolved oxygen, conductivity, turbidity, and water depth) and benthos assemblages (abundance/density and diversity), and their influence on waterbird distribution.
- compare present habitat conditions and bird communities to past situations.
- evaluate the potential use of waterbird species as bioindicators of wetland ecosystem health.

CHAPTER TWO

LITERATURE REVIEW

2.1 Waterbirds and Wetlands

Waterbirds belong to the orders Anseriformes, Charadriiformes, Ciconiiformes, Gaviiformes, Gruiformes, Pelecaniformes, Podicipediformes and Procellariiformes, (Bolduc and Afton, 2008). They are further categorised into three main groups: (i) Seabirds (e.g. gulls and terns), (ii) Shorebirds/Waders (e.g. plovers, sandpipers, stilts) and (iii) Waterfowl (e.g. ducks). Waterbirds are ecologically dependent on the interconnected networks of wetlands, which serve as staging or stopover sites along their migratory flyways. They use these habitats for foraging, nesting, roosting, moulting and comfort activities.

Even though wetlands are essential, dynamic biotopes that support and maintain a diverse community of birds through their seasonally-available resources during both migration and breeding periods, wetlands are limited (patchy) in their distribution on the terrestrial landscape (Duncan *et al.*, 1999; Liang *et al.*, 2002).

2.2 Importance of Waterbirds

Waterbirds have high aesthetic and cultural values. They are used as symbols by both traditional communities and environmental organisations, where they serve as flagship species, helping in the conservation of other biodiversity components. Aside their biological, cultural, and aesthetic values, waterbirds and terrestrial birds are also the focus of an ever-expanding bird watchers' community all over the world, hence

contributing to a country's economy in general and that of the local communities in particular through ecotourism. Large waterbirds, such as ducks, provide good food sources for local communities.

In order to achieve sustainable utilisation and conservation of natural resources at the local, national and international levels, there is the need to assess ecosystem conditions and changes, and to determine the necessity for active management. However, in assessing these changes, it is practically difficult and costly to assess changes within the function, properties, and values of an ecosystem. Selected components can therefore be used as proxy measures of wider conditions. The quality of these indicators depends on their inter-relationship with the others. Biological components serving these functions are called biological indicators and may be derived from any level of biological organisation.

As a key biodiversity group in wetlands, waterbirds are widely used in monitoring and surveillance programmes as indicators of wetland value and status, as well as parameters of restoration success of regional biodiversity (Scott and Rose, 1996). They represent good ecological/environmental indicators of habitats in which they occur and contribute greatly to secondary productivity (Ahulu *et al.*, 2006). The occurrence of waterbirds at the top of the food chain makes them sensitive to changes in their habitats and factors affecting those below them. Waterbirds such as seabirds are hence vulnerable to bioaccumulation of toxic chemicals and have been used as good indicators of marine pollution (Furness and Greenwood, 1993). The sensitivity of waterbirds to environmental changes and their high species turnover also provide a means to assess the ecological status of a site (Morrison, 1986; Furness and Greenwood, 1993).

Colonial waterbirds are used as bioindicators of habitat quality and condition as well as parameters of restoration success (Scott and Rose, 1996). The bioindicators derived from colonial waterbirds include contaminant burden and population numbers (egg shell quality, growth, behaviour, mortality, presence/absence, distribution, and population indices, etc.), and certain suborganismal bioindicators (genotoxicity, tissue concentration of contaminants) (Furness and Greenwood, 1993).

Waterbird abundance is indicative of abundance of prey such as fishes and macroinvertebrates they feed upon. For example, a study conducted at Densu floodplains suggested that Common Terns for example were indicators of rich fish stock in the sea and in lagoons (Ahulu *et al.*, 2006). Changes in the ranges of migratory waterbirds may result from changes in habitat quality and mortality rates (Ogden *et al.*, 1987). Increasing numbers of nesting migratory waterbirds could be as a result of the abundant food available at the foraging sites. Moreover, Norris *et al.* (2004) and Newton (2004) observed declines in the populations of Redshank as a result of nesting site destruction in summer by land drainage. Population sizes of migratory waterbird species are therefore influenced by conditions in more than one part of the world. Everything depends on what has been lost and substituted, as well as the degree of disturbance (Norris *et al.*, 2004).

As compared to other taxa used as bioindicators of wetland quality, waterbirds which are highly conspicuous and have longer life-spans, enable scientists to carry out long-term researches. This character is most important in conservation and environmental impact assessment since long-term changes in diversity, population and species assemblages of waterbirds are indicative of changes in the ecological character of their habitat. The availability of large data-bases of both survey protocols and trends in

population size and distribution of waterbirds make them better indicators of wetland condition/health. Waterbirds therefore represent an important tool in biodiversity conservation, due to their conspicuousness, abundance, high species turnover, and sensitiveness to changes occurring in their habitat. They are useful in determining the quality of the wetland in which they live (Temple and Wiens, 1989).

2.3 Wetland Quality

“Habitat”, defined as “the resources and conditions present in an area that produce occupancy, including survival and reproduction by a given organism”, is of great importance in animal ecology and conservation science. Its quality is a determinant of the type and health of the organisms living in it (Johson, 2007).

According to Hall *et al.* (1997) habitat quality is the ability of an environment to provide conditions appropriate for individual and population persistence and can be identified at three operational levels; individual, species and population. While individual-specific habitat quality tends to improve the fitness of an organism in that environment, population-specific habitat quality tends to support the largest population (Franklin *et al.*, 2000). Nonetheless, the former has a higher intrinsic rate of population growth and the latter, a higher carrying capacity. Individual birds will get access to quality food, hence maximising their chances of survival and reproduction.

An assessment of habitat quality can therefore be carried out directly by measuring attributes of habitat itself, or indirectly by measuring variables for individual birds and/or populations in different habitats to reveal variations in habitat quality or both (Johson, 2007). Nevertheless, habitat quality is not limited to only the immediate

surroundings of the birds or the resources necessary for its survival and reproduction, but also the conditions that constrain their use (Morrison *et al.*, 2006). It is thus necessary to consider the response of the birds to all factors in its environment in order to make conclusive deductions and understand the functioning of the birds' ecosystem.

Financial and time constraints coupled with the feasibility and limited staff available, contribute to the limited knowledge available on habitat use by individual waterbird species and factors affecting it. Hence, it is difficult to successfully and directly assess the quality of the habitat of that particular bird species or the whole population in the particular area. Barnes *et al.* (1995), for example, evaluated habitat quality for Northern Bobwhites (*Colinus virginianus*) by quantifying grass forage quality, food (insect) abundance and availability of cover.

Furthermore, Johson (2007) classified bird-based indicators of habitat quality into broad groups; demographic, distributional and individual condition measures to which behavioural observations could be added. He however noticed that most demographic studies were limited to abundance and reproduction, leaving the life-history of birds. Though not complete, such studies have yielded essential information on habitat quality. Some of the limitations are the fact that they did not take into consideration human disturbance, and also the fact that demographic data are difficult to obtain. They reveal only a portion of a habitat's value.

Other measures that can be used to indicate habitat quality include habitat selection when well described protocols are available. Changes in waterbird demographics or performance over a period of time or in sites displaying a gradient of a variable of interest may reveal variations in habitat quality. However, in some cases, birds may choose poor and avoid rich habitats and can further exhibit site fidelity (Rapport, 1991;

Railsback *et al.*, 2003). It is thus important to know and understand the ideal distribution of an animal and its habitat selection before using its distribution to reveal variations in habitat quality.

Individual condition measures (weight, growth, etc.) represent the third means of habitat quality assessment. However, for successful habitat quality evaluation, there should be a combination of demographic, distribution and individual condition measures. It is also important to know how spatio-temporal variations are linked and/or determined by habitat quality.

2.4 Factors Affecting Habitat Use by Waterbirds

2.4.1 Waterbird abundance and diversity

As a response of waterbird communities to the type, size and heterogeneity of the various components of wetlands, as well as migration, there is a spatio-temporal variability on the species-habitat relationship. The temporal variability is mainly driven by the influx of waterbird species during their migratory period (Page *et al.*, 1979; Shuford *et al.*, 1998; Mungula *et al.*, 2005). The presence of different habitat types in a wetland increases species richness, which varies also with seasonal changes in these habitats. In wetlands of Central-South Chile, for example, Gonzalez-Gajardo *et al.* (2009) observed spatiotemporal fluctuations in the composition of waterbird assemblages as a response to the structural heterogeneity. According to Zou *et al.* (2006), of all the wetland types, the intertidal mudflats are the richest habitats both in terms of species richness and abundance of waterbirds.

Furthermore, Paszkowski and Tonn (2000) suggested that the bigger a wetland is, the higher the number of microhabitats it may provide, hence the greater the number of species. In an urbanised area of the Targus Estuary, in Portugal, for example, Rosa *et al.* (2003) observed that species richness was positively influenced by wider mudflats and margins covered by salt marsh. Even though van de Kam *et al.* (2004) reported that migratory waterbirds are more attracted to large expanses of wetlands than the small ones, Hudson (1983) and Garay *et al.* (1991) showed that smaller wetlands could also maintain equally important waterbird densities and diversities. In Ghana, Gbogbo (2007b) found that unmanaged wetlands which are smaller than the managed ones support higher densities of ducks and cormorants, stalking herons and fishing terns, and therefore the importance and conservation of which cannot be neglected.

2.4.2 Waterbird distribution in wetlands

Waterbird distribution may be categorised as large-scale and local. Large scale patterns in shorebird distribution are mainly altered in several ways by the natural patchy occurrence of wetlands (Brown and Dinsmore, 1986; Knutson *et al.*, 1999), whereas local distribution is mainly influenced by the patchy food distribution within these wetlands (Piersma *et al.*, 1993). Furthermore, as a result of this irregular distribution of prey items in the patches within wetlands, there are variations in the distribution patterns of shorebirds, which could be also associated with the variability of the coastal landscape at a regional scale (Myers *et al.*, 1987; van Gils *et al.*, 2003). The presence of safe roosting sites within a lagoon or a wetland also determines the distribution of roosting birds in such ecosystems (van de Kam *et al.*, 1999; Blanc *et al.*, 2006).

2.4.3 Foraging birds

Waterbirds feed on a large variety of food ranging from macroinvertebrates (e.g. plovers), small fishes (Black-Winged Stilt and herons) to seeds and plants (herbivorous ducks). They consequently have different foraging habitats ranging from grasses around water bodies, shallow waters, mudflats, salt marshes, mangroves, etc. The benthic invertebrates, the primary food of waterbirds, strongly influence the feeding behaviour and distribution. Waterbird communities are therefore influenced by types, sizes and quantities of food available within wetlands (Murkin and Kadlec, 1986; Colwell and Landrum, 1993; Skagen and Oman, 1996)

According to Piersma and Ntiamoa-Baidu (1995) and Battley *et al.* (2003) waterbirds concentrate in areas with high food density. There is a strong correlation between densities of shorebirds and their prey (Meire, 1993; Kalejta and Hockey, 1994). Individual birds choose their foraging habitats based on their expected food intake rate rather than food density, though there is a strong correlation between bird distribution and food (Piersma, *et al.*, 1993). However, in the case of visual foraging birds such as Grey plovers (*Pluvialis squatarola*), Kalejta and Hockey (1994) found that their densities are better predicted by prey biomass rather than abundance.

On their feeding grounds, waterbird distribution is determined by their diet and factors likely to influence the availability/accessibility of their food, safe roosting sites and the extent of disturbance, which form the basis for habitat selection during the non-breeding period (Piersma, 1994). Furthermore, food availability of waterbirds is driven by their feeding adaptations and strategies, hydrology, and physiochemical environment of the wetland in question. Nevertheless, due to the burgeoning human

population and its impact on waterbird habitat, human disturbance is a critical factor that is likely to influence their activities.

2.4.3.1 Morphological adaptations and feeding strategies

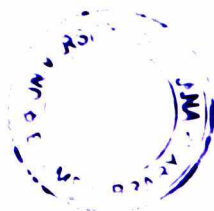
Birds have evolved several mechanisms to adapt to their niches, like specialised feeding methods, as well as bill and toe modifications. These morphological adaptations, in combination with the type of food they eat define their feeding habitat, the range of food they could prey upon and their feeding styles. For instance, habitat selection by non-diving waterbirds in terms of water depth selection depends on characteristic bill lengths, shapes, neck length, leg lengths and body sizes that allow them to feed at specific water depths. In order to facilitate the study of the feeding ecology of waterbirds, feeding guilds have been formed based on these character combinations and the way waterbirds obtained their food (Zwarts and Wannink, 1984).

Waterbirds are therefore more-or-less specialised for feeding on certain prey types, and whenever two species or sub-species were found to depend on the same resources, they both developed their restricted foraging range in order to reduce competition and maximise their food intake (Beukema *et al.*, 1993). For example, even though Waders such as oystercatcher (*Haemantopus* spp.) and knots (*Calidris canutus*) have both specialised in bivalves, they have restricted range of prey sizes and feed on prey burrowed at a limited depth in sediment. They also select habitat with higher densities of benthic invertebrates (Safran *et al.*, 1997). In situations where there is short supply of food resources due to temporal or local changes in the distribution, abundance and availability of their primary food resource, bird species can be induced to either change their feeding area or switch to their alternative preys. This switching behaviour can

therefore be considered as an optimal foraging behaviour in that it helps maximise energy intake (Beukema *et al.*, 1993).

When displaying an optimal foraging behaviour, many waterbird species select preys that give the maximum energy return for the energy invested in capture (Zwarts and Drent, 1981; Ens *et al.*, 1994). Zwarts and Drent (1981) observed oystercatchers switching from mussels to cockles when the banks of mussels had disappeared. During severe scarcity, especially after a severe winter, they either switch back to other areas with mussels or other areas with non-mussel prey (Zwarts and Wanink, 1993). Desprez *et al.* (1992) observed them to switch to other prey than cockles.

Ens (1982) however, observed that waterbirds do not always select prey that generate more energy but those whose profitability exceeds the average intake rate. He also found that oystercatchers hammering mussels did not select large mussels as a result of selectivity but profitability. They feed only on a given class of mussels and they may spend more time trying to feed on thick mussels. The ability of a bird to cope in a profitable way determines its ability to switch to its alternative prey. The success of switch further depends on the availability and density of the alternative prey (Zwarts *et al.*, 1992) and thus on the stability and fluctuation patterns of the alternative prey. If unstable as well, then for a successful switch, the alternate species should show synchronisation in their fluctuation patterns.



2.4.3.2 Prey distribution and availability

The physico-chemical environment of wetland directly or indirectly affects waterbird feeding ecology. It is direct in the sense that morphological constraints of waterbirds determine where they feed in different water depths and columns (Safran *et al.*, 1997) and indirect through its effect on the distribution of their food.

Wetland use by waterbirds is highly dependent on the fluctuating hydrology (Kingsford *et al.*, 2004) caused by unpredictable events and water chemistry (Liang *et al.*, 2002) and the species-specific depth requirement (Bolduc and Afton, 2004). Hydrological diversity within and among wetlands, often dictates where and when waterbird species can access their food. Young and Chan (1997) stated that large wading birds such as herons and egrets, for example, choose feeding sites in drained fish ponds where water is shallow and fish are more concentrated. Ntiamoa-Baidu *et al.* (1998) found that small Waders feed in habitat ranging from dry mudflats to wet mud, and shallow water of not more than 20 cm. In coastal wetlands, foraging birds are more abundant at low tide when the intertidal macroinvertebrates are more exposed. Hence the role played by the tide in the feeding ecology of waterbirds.

Gbogbo *et al.* (2009) found that waterbird species belonging to Guilds 1, 3, 4 and 5 responded significantly to decreasing water level, while those belonging to the Guild 2 responded significantly in a second order polynomial function. Their numbers were hence optimum when water levels were neither too high nor too low. Some studies have shown that waterbird density is highest at low water depth, thus the assumption cannot be generalised (Kingsford *et al.*, 2004; Holm and Clausen, 2006). Bolduc and Afton (2004) explained that there should be an optimum water depth at which

waterbirds are expected to maximise their resource utilisation, otherwise it will mean that waterbird numbers are highest when water depth is virtually zero.

Factors such as salinity, dissolved oxygen, temperature, turbidity, sediment hardness, oxygen penetrability and particle size, affect waterbird distribution through their effect on benthic macroinvertebrates. According to Leland and Fend (1998) and Mc Rae *et al.* (1998), salinity significantly affects benthic invertebrate communities as they exhibit differences in salinity tolerance levels (Euliss *et al.*, 1999). During the rainy season, the reduction in salinity changes the benthic community structure and composition (Seapy, 1981; Wilcox, 1986), while Liang *et al.* (2002) observed a decrease in small benthic invertebrates with increasing salinity.

Dissolved oxygen is one of the physical parameters that determined the health of a water body. It gives an indication on the capability of a given water body to support biological processes. The optimum range for dissolved oxygen is 5-10mg/L; when it is below 2mg/L it becomes unfavourable to fishes and other fauna. High temperature values and salinity lead to environmental stress in benthic communities, whereas high water turbidity inhibits waterbird vision when feeding. High water temperature also limits light penetration that is necessary for photosynthesis of algae and phytoplankton that serve as primary producers in aquatic ecosystems and also represent food sources for benthic organisms.

Another factor such as sediment exposition time also influences bird distribution patterns through food availability (Kalejta and Hockey, 1994). There is high positive correlation between the number of Dunlin (*Calidris alpina*), Common Redshank (*Tringa totanus*) and Grey Plovers (*Pluvialis squatarola*), among others, and the width of the intertidal area in wash estuary. Physiognomy probably influences food

availability in that wider sectors will have consequently longer exposition time and consequently allow longer feeding periods (Goss-Custard and Yates, 1992; Yates *et al.*, 1996).

Several studies have shown the importance of habitat physiognomy and preference in wetlands. Blanco *et al.* (2006) found that shorebird distribution was heterogeneous in the Buenos Aires (Argentina) coastal zone with estuarine saltmarsh landscape having higher use and providing better foraging condition. Sanderlings (*Calidris alba*) prefer sandy beaches where they occur in patches in areas where the waves have just receded (Morrison and Ross, 1989; Petracci, 2002). This is known to increase penetrability of their beak. Shorebird distribution and abundance in the Central Valley of California, for instance is simultaneously influenced by species habitat preferences, seasonal and geographic variation in habitat availability and north to south variations in climatic conditions (Suford *et al.*, 1998). They also found that in some cases, habitat and physical variables could be used as predictors of feeding conditions.

2.4.4 Roosting waterbirds

Roosting behaviour in waterbirds is a complex phenomenon (Koffijberg *et al.*, 2003). They gather in large and dense flocks, mostly in mixed species, in order to minimise the risk of predation for individuals, but also save energy when staying close together at roost. In intertidal areas, this phenomenon is mostly observed at high tide. However, the attendance of high tide roosts depends on actual water tables, distance to the nearest feeding areas, the degree of human disturbance and species specific behaviour.

2.5 Migration in Waterbirds

Waterbird species can be classified as sedentary, nomadic or migratory. The migratory species experience seasonal, daily or irregular movements that may cover short distances (microgeographic migration) or thousands of kilometres (macrogeographic migration) (Linzey, 2001). Black-Crowned Cranes (*Balearica pavonina*) migrating from southern Senegal to wetlands in Guinea-Bissau, and the Artic Tern (*Sterna paradisea*) which travels a distance of 50,000 km between Antarctica and northern Scandinavia, are examples of short and long-distance migrants respectively. The Artic Tern has a record of covering a distance of 50,000 km in a single year (Jourdain *et al.*, 2007).

These periodic movements or migration of a population or part of a population of animals away from a region and their subsequent return to that same region is caused by the local depletion of harvestable prey and high cost of maintenance during unfavourable conditions (Ens *et al.*, 1994). In some species with two distinct populations (discrete migratory and sedentary) such as the Eurasian Spoonbill (*Platalea leucorodia*) which occurs in Africa, only part of the population migrates during winter (Dodman and Diagana, 2007). It is also known that the cost of travelling by medium-sized to small waders such as Knots (*Calidris* spp.) from temperate regions to the tropics is much lower than the cost of living in the temperate regions during winter (Piersma *et al.*, 1991). Knots travelling to Mauritania, Banc d'Arguin in West Africa save relatively about 45% of an average maintenance energy requirement spent if they had stayed in Europe during the northern winter.

2.6 Waterbirds in Coastal Wetlands of Ghana

Coastal wetlands in Ghana are located in three habitat types; (i) estuaries with sandy shores (e.g. Esiama), (ii) brackish water lagoons/ salt pan complexes (Elmina, Pambros), (iii) brackish lagoons with shallow margins and mudflats (Muni, Sakumo, Songor and Keta). Moreover, Ghana's coastal wetlands are effectively not tidal systems but rather lagoons with surrounding flood plains (van de Kam *et al.*, 2004). Of these lagoons, two types can be distinguished: (i) open lagoons associated with large rivers which have a permanent connection to the sea, and (ii) closed lagoons with sand dunes that can break intermittently depending on the water levels caused by heavy rains (Mensah, 1979; Gordon, 1987). The open system is ecologically more stable than the unpredictable closed ones (Finlayson *et al.*, 2000).

Coastal wetlands are important areas for resident and wintering Palaearctic migrant waterbirds that use the East Atlantic and Mediterranean flyways (Smit and Piersma, 1989; Van de Kam *et al.*, 2004). They receive significant numbers of waterbirds from different breeding ranges than most wetlands in West Africa, more specifically other countries along the Gulf of Guinea (Altenburg *et al.*, 1983; Tye and Tye, 1987). The value of the coastal lagoons of Ghana as staging and wintering grounds for migratory waterbirds has been documented in several publications based on data that has been gathered since 1986.

Out of approximately 100 wetlands found along the coastline of Ghana, eight, namely Esiama Beach, Elmina Beach, Densu Delta, and Muni, Sakumo, Keta and Songor Lagoons, have been recognised to be of importance in terms of total populations and bird species. Data gathered from these sites informed the process of designating five of

Migration is a complex phenomenon (Bairlein *et al.*, 2002; Rees *et al.*, 2005) affected by series of factors ranging from general, seasonal environmental, ecological to species-specific biological and physiological factors. Thus the migration strategies vary according to physiological/biological characteristics of the birds and unfavourable climatic/environmental conditions affecting them through limitations of food availability. The rainy season in Africa, winter in the northern hemisphere, wind speed and direction are major environmental variables that trigger migration. The effect of climate change on timing changes migratory behaviour at the scale of populations. As a result of a combination of factors, waterbirds use different migration strategies, the extents of which are related to their differential evolutionary fitness (Ens *et al.*, 1994).

There are different migration strategies by waterbirds, some of which have been observed in waders moving from coastal West Africa to sub-Arctic breeding grounds. Birds that require local or regional interspersed habitats after every short distance flight, like Turnstone (*Arenaria interpres*), usually hop from closely interspersed habitats. Others such as Dunlin (*Calidris alpina*) and Redshank (*Tringa totanus*) skip, or fly long distances without stopping, whereas Red Knot (*Calidris canutus*) and Bar-tailed Godwit (*Limosa lapponica*) jump from one part of the hemisphere to another.

Despite all this complexity of factors attributed to migration, many species of waterbird prefer to migrate than to stay in the northern hemisphere during winter and further exhibit site fidelity behaviour which can be expressed as a result of various selective pressures that favour individuals which have an intimate knowledge of their environment. Detailed knowledge of migration and its routes and stopover sites was derived from years of bird ringing (using metal rings and individual colour-marks) and recently from satellite tracking (some species) (Davidson *et al.*, 1999).

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them (Muni, Densu, Sakumo, Songor and Keta) as Ramsar Site (Ntiama-Baidu, 1991; Ntiama-Baidu and Gordon 1991).

Migrant waterbirds begin to arrive on the Ghanaian coast from the end of August to early September and remain until April. The peak numbers however usually occur from September to December. Waterbirds are thus most abundant during the dry season when water levels in lagoons are falling and the shallow waters and exposed mudflats offer favourable condition for foraging. More than 50 species of waterbirds occur in coastal lagoons (Piersma and Ntiama-Baidu, 1995; Gbogbo, 2007a,b; Gbogbo *et al.*, 2009).

Migrant waders arrive in coastal Ghana in August and reach a peak in November/December and most (50%) of them leave in January leaving a small portion which remain until final departure in April. They are observed to use more than one wetland during their stay on the coast of Ghana; hence the evidence of local movements within sites and between habitat zones (e.g. Sanderlings ringed in Essiama were found in Muni lagoon). Furthermore, there is a site preference character exhibited by some waders; Sanderlings and Oystercatchers prefer sandy beaches whereas curlew sandpiper, spotted redshank and avocet prefer mudflats. They are consequently found to be more abundant in one habitat zone than the other.

The importance of these coastal wetlands is not restricted to only their international important populations of birds but also the socio-cultural and economic values they provide to local communities. They are used for fishing, farming and ecotourism.



2.7 Threats to Ghanaian Coastal Wetlands

The Ghana coast is not an exception to the globally increasing urbanisation and industrial development that affect coastal regions worldwide. Attuquayefio and Gbogbo (2001) stated that of the threats affecting coastal wetlands, land ownership, fishing, hunting of waterbirds, sewage disposal and pollution, cutting of mangroves, farming and habitat reclamation and commercial salt production greatly contribute to the degradation of wetlands and loss of biodiversity in Ghana. According to Gordon *et al.* (1998), wetlands such as Korle, Teshie and Kpeshie, Fosu and Benya lagoons are so polluted that they can no longer support life. Another prominent threat to coastal wetlands and therefore waterbirds in Ghana is sea level rise, which may reduce the intertidal mudflats areas used by waders. Habitat could be lost through erosion, fragmentation, pollution and even inundation, and consequently waders belonging to Guilds 2, 3, and 4 are most at risk, more especially against the background of the globally decreasing number of waterbird species.

Most studies on waterbirds and wetlands in Ghana were carried out in the Keta-Songor complex, the largest wetland, and few others on Sakumo, Densu, Laiwi and Mukwe lagoons (Gbogbo, 2007a, b; Gbogbo *et al.*, 2009) in the Greater Accra region. Muni-Pomadze Ramsar site is the smallest of all the five coastal wetlands in Ghana.

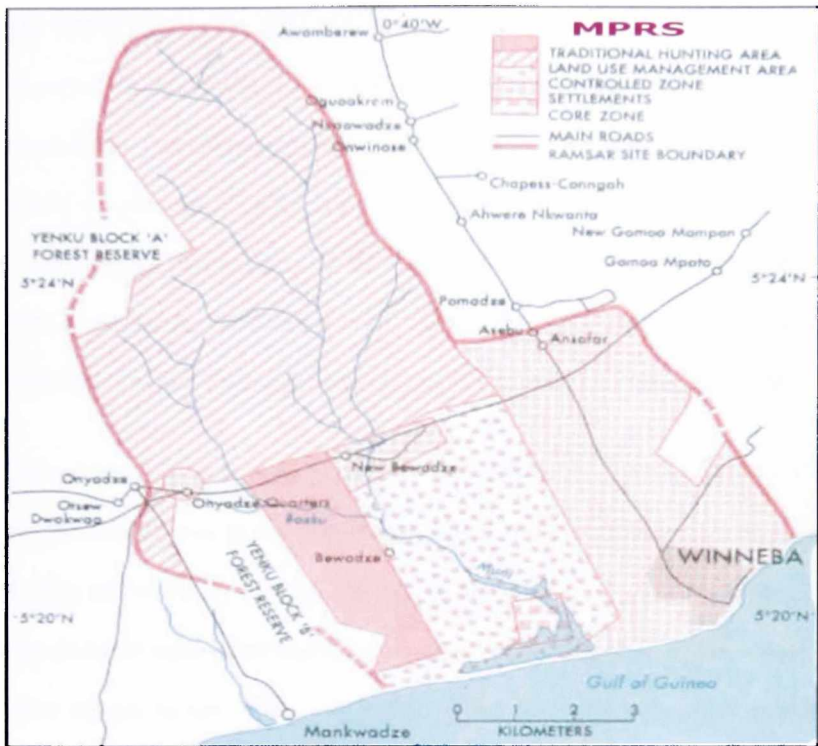
CHAPTER III

MATERIALS AND METHODS

3.1 Study Area

3.1.1 Location and size

The Muni-Pomadze Ramsar Site (5°19'-5°27' N, 0°37'-0°41' E) is located south-west of the coastal town of Winneba in the Central Region of Ghana (Gordon *et al.*, 2000; Koranteng *et al.*, 2000) approximately 55 km west of Accra. This site was designated as a Ramsar Site in August 1992. It covers an area of 95 km², which is the entire catchment area of three seasonal streams, the Pratu, Goaku and Muni which drain into the Muni Lagoon. The area of the shallow, saline lagoon is estimated at 3 km² (Ntiemoa- Baidu and Hollis, 1988) but may expand across the surrounding floodplain to 6 km² during the rainy season. The lagoon is separated from the sea by a sand bar during the dry season which gets breached intermittently during the rainy season. This Ramsar Site also encompasses two forest reserves (Yenku A and B, and salt pans) (Figure 3.1).



MUNI-POMADZE RAMSAR SITE

Figure 3.1: Muni-Pomadze Ramsar Site

3.1.2 Climate and vegetation

The MPRS experiences a bimodal rainfall pattern with two distinct peaks and troughs occurring in March/April-July and September-November and December-February respectively. Annual precipitation is about 850 mm (Gordon *et al.*, 2000). The mean annual temperature of MPRS is around 27°C with the relative humidity as measured at 15:00 hours GMT averaging 70-80% for most part of the year and dropping to below 65% in the dry season (Gordon *et al.*, 2000). These climatic conditions show that the MPRS falls within the coastal savanna zone (Wuver and Attuquayefio, 2006).

Two meteorological stations are found at the MPRS. The first is located at Winneba to the east and the second at Apam to the west. The mean rainfall was highest in January 2010 in all the stations (Figure 3.2) and the tide level was highest in October 2009. The highest mean water depth was however observed in February 2010 as compared to the mean rainfall values. There was a drop in mean rainfall values in December and a gradual decrease from October to December 2009 and February to April 2010 (Figure 3.2). The open Muni Lagoon is thus influenced by both rainfall and tidal level in the area.

Fifty-three percent of the Ramsar Site is classified as natural vegetation, 32.5% as agricultural land, and the remainder as built-up areas (Gordon *et al.*, 2000). The natural flora of the site is divided into four main types (Oteng-Yeboah, 1994): (i) flood plain vegetation (mangrove and wetland vegetation), (ii) dune vegetation, (iii) riverine vegetation, and (iv) terrestrial vegetation on elevated ground (combination of grasslands, thickets, and eucalyptus plantations) (Gordon *et al.*, 2000). The floodplain vegetation surrounding the Muni lagoon is consistent with the vegetation found at the

shoreline of other coastal lagoons in Ghana (e.g. *Sesuvium portulacastrum*, *Paspalum virginicum* and *Sporolobus virginicus*).

3.1.3 Geomorphology and soils

The geomorphology of the MPRS can be categorised into four sections (Gordon *et al.*, 2000):

- Yenku Hills with a maximum height of 290 m and the Egyasimanku Hills (max. Height = 205 m) spread from the north to the north-east and the south-west portion of the Ramsar Site.
- A complex of estrusive and hyperbyssal rocks (greenstones) from the upper Brimian that serves as a basement to the western part of the catchment area.
- Lagoonal deposits of sand located in the central portion of the catchment.
- Outcrops of Tarkwaian quartzschists and biotite-hornblends, with grey and pink phenocrysts in the eastern part of the MPRS. The sand bar separating the lagoon from the sea is formed by the strong long shore littoral drift of sand from the west to east direction.

Amatekpor (1994) stated that “soils of the MPRS vary with the elevated and undulating areas, as well as the lagoon; with higher ground on the slopes being occupied by the Adzintam-Mankoadzi complex (37% of the catchment), and the summits and steep slopes of the hills in the area, occupied by a shallow clay/ loam soil, the Yenku-Adzimtan (12%). The fourth type of soil, the Oyibi-Muni association (11%) is also a poorly-drained soil with clear stratification in the sands and clays. It is the typical soil found in coastal lagoons in Ghana. The last types of soils, the Osibi-Bumbi association (39%) are low land soils with poorer drainage”.

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The topography of the Muni-Pomadze Ramsar Site has a slope of not more than 5-20% grades. If left bare, the moderately to highly erodible soils on slopes of 2-25% could yield up to 16 tons per hectare per year of sediment which would eventually end up in the lagoon.

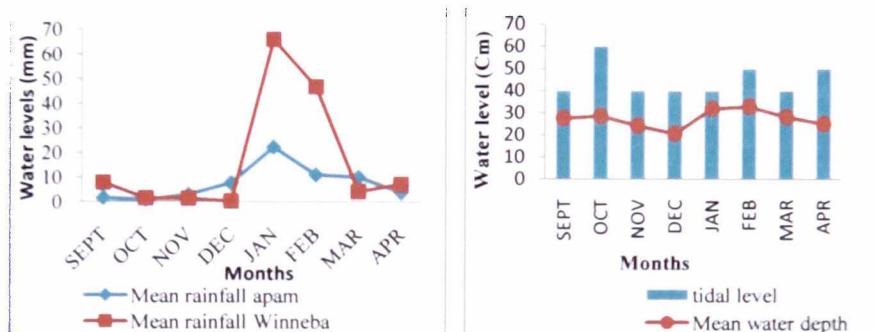


Figure 3.2: Rainfall Patterns, Tidal Level and Water Depth at the Muni Lagoon during the Study Period (Sept-09 to Apr-10).

3.1.4 Demography and socio-economics

The nine major settlements within the MPRS constituted a population of 32,000 people in 1984. This went up to about 47,327 by the year 2000 (Ghana Statistical Service, 2005). The population growth rate is less than 1% annually (Gordon *et al.*, 2000). The dominant ethnic groups are the Fanti, Ewe and migrant Fulani herdsmen. The management of the site is vested jointly in two local or traditional authorities. The Effutu have responsibility for issues concerning the Muni Lagoon, while the Gomoa manage the wetlands. Chief Fishermen are responsible for fishing-related issues in their respective traditional communities.

The primary occupations of the local inhabitants are fishing and farming (e.g. maize and cassava), but there are other minor activities like hunting (Effutu people), cattle grazing, sand, clay and gravel mining, salt winning and charcoal production. The MPRS is home to the annual “Aboakyir” festival, during which “Asafo” companies compete to capture a live bushbuck (Wuver and Attuquayefio, 2006).

3.2 Methods

3.2.1 Selection of study sites

The Muni lagoon was stratified into six sampling sites: (i) Site A (Mudflat I), (ii) Site B (Inland Water), (iii) Site C (Lagoon Mid-Section), (iv) Site D (Mudflat II), (v) Site E (Seaward Site), and (vi) Site SP (Salt Pond) (Figure 3.3). The extents of these sampling sites were defined using physical references of the landscape. Sites A, C; and D are located in the eastern part, centre of the lagoon, Site E in the southern part, and Sites B and F in the northern part of the lagoon.

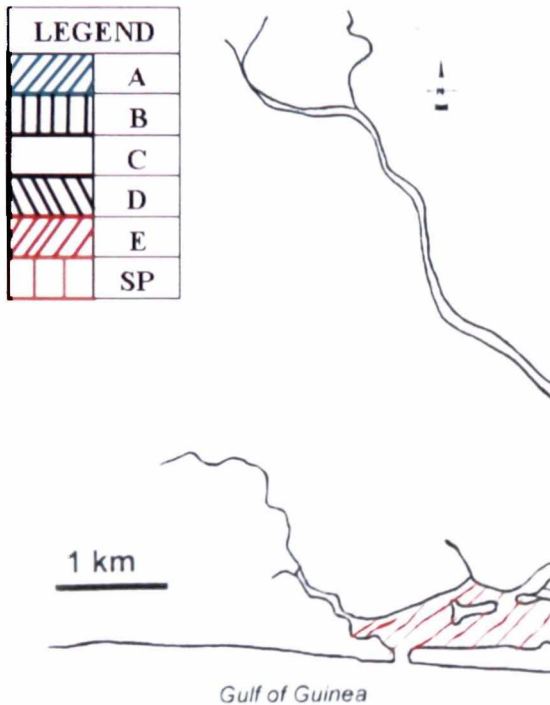
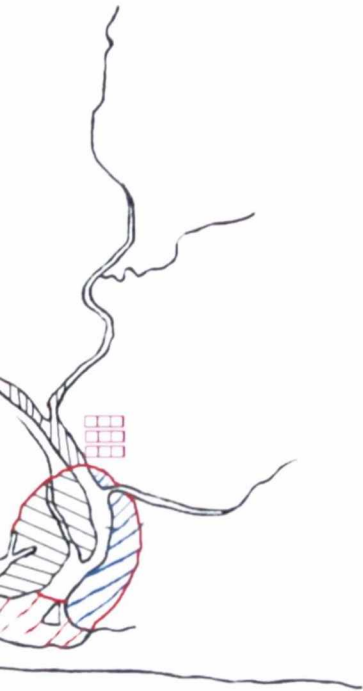


Figure 3.3: Sampling Sites in the Muni Lagoon



3.2.1.1 Site A (Mudflat I)

This site is close to Akosua village, with portions of it serving as a waste dump site for the local population. There is flooding at high tide and water depth is the lowest compared to the other sites (Figure 3.4). The site is also linked to a small stream which becomes prominent during heavy rains. The vegetation is dominated by *Sesuvium portulacastrum* and other grasses. This site has the highest soil clay percentage (17.54%) of all the sites (Table 3.1).

3.2.1.2 Site B (Inland Water)

This site receives freshwater inflows from the Pratu, Boaku and Muni rivers. During the dry season, there is high salinity due to high evaporation and low freshwater inflow. The soil composition is sand (84.8%), clay (13.6%) and silts (1.6%) (Table 3.1).

3.2.1.3 Site C (Lagoon Mid-Section)

This site represents the mid-section of the Muni lagoon, where fishing activities are more prominent than at the other sites except Site E. Dead seashells are also most abundant at this site. Sandy soil dominates (84.9%) in the area.

3.2.1.4 Site D (Mudflat II)

This is also a mudflat bordered by mangroves to the west. Water depth is affected by the tide levels of the sea, resulting in flooding of the site during high tides.

3.2.1.5 Site E (Seaward Site)

This site connects the lagoon to the sea, and is thus the deepest portion of the lagoon (Figure 3.4). There is fishing at the site, which had the highest soil silt composition (2.6%) of all the sites.

3.2.1.6 Site SP (Salt Pond)

This site is the most hypersaline of all the sites due to salt production using water pumped occasionally from the sea. There is no direct influence of increasing or decreasing water levels in the lagoon, but rather by the frequency of sea water pumping into the salt ponds. Wind force and direction influence the mixing of the water column in the salt ponds because of the enclosed nature of the site.

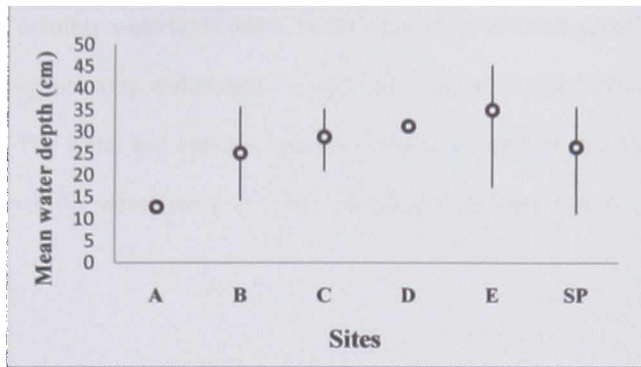


Figure 3.4: Spatial Variations in the Mean Water Depth at the Muni Lagoon over the Study Period (Sept 09-Apr 10).

Table 3.1: Soil and Sediment Percentage Composition at the Muni Lagoon

	SILT	SAND	CLAY	TOTAL
A	1.54	80.92	17.54	100
B	1.60	84.8	13.60	100
C	2.22	84.87	12.91	100
E	2.59	80.90	16.51	100

3.2.2 Sampling design

At each site, monthly waterbird counts, benthic macroinvertebrate sampling and water quality assessments were undertaken for a period of eight months (September 2009 to April 2010). The water and benthos sampling points at each site are shown in Figure 3.5 and the GPS coordinates of the sampling locations are in Appendix I.

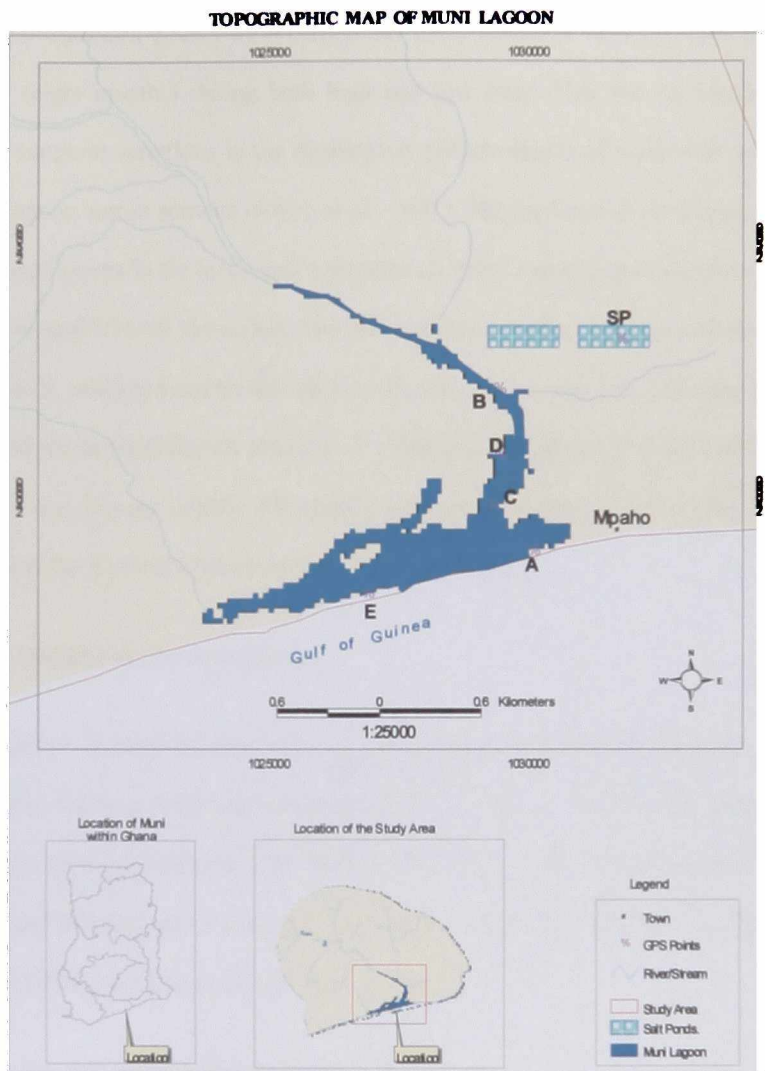


Figure 3.5: Sampling Points in the Muni Lagoon (Source CERSGIS)

3.2.3 Avifaunal survey

Multiple waterbird counts were carried out every month at each site over the study period (eight months) during both high and low tides. This was to determine the spatio-temporal variations in the distribution and abundance of waterbirds within the Muni lagoon across seasons (Bibby *et al.*, 1993). The total counts were done on foot, from fixed points in the landscape with good visibility along transects using a 20 x 60 telescope and 10 x 40 binoculars. The surveys always began from the eastern bank of the lagoon, moving towards the inlets of the two rivers, and then returning to cover the southern banks (Ntiamoa-Baidu *et al.*, 2000a). Bird species identification followed Borrow and Demey (2004). All species seen were counted and the total for each species at the six sites were pooled at the end of the count.

3.2.4 Habitat use by waterbirds

The number of birds feeding, roosting (sleeping, standing) and undergoing comfort (preening, bathing, stretching) activities were estimated during low tide counts every month at each site using the scan method. Waterbirds were categorised into Waders, Terns and Herons, and further put into guilds according to Piersma and Ntiamoa-Baidu (1995), and Ntiamoa-Baidu *et al.* (1998).

3.2.5 Benthic macroinvertebrate sampling

At each of the sampling sites, five sediment cores were taken using a hand held soil PVC corer, with a core area of 0.177m² in order to examine the benthos. The core depths ranged from 5-15 cm. The benthos was then carefully removed from each corer separately and extracted by wet sieving using a sieve of 300 microns mesh size. Drops of Rose Bengal were added to the collected substrate to improve identification

of the specimens which were then fixed in 4% formalin and sent to the laboratory. Benthic species identification by means of a microscope followed Day (1967a, b). Species richness and numbers of individual benthic macroinvertebrate prey were estimated as well as the biomass at each section. After identification, the macroinvertebrates were dried at 60°C in an oven for two days, and the dry weight recorded. The benthos was grouped into polychaetes, bivalves and crustaceans.

3.2.6 Measurement of environmental parameters

Environmental factors such as salinity, temperature, dissolved gases, turbidity and water depth, are key factors determining the activity, abundance and distribution of waterbirds as well as the macrobenthic community. These environmental parameters of the Muni lagoon were measured *in-situ* over the study period.

- **Mean Water Depth**

Mean lagoon water depth measurements were taken by measuring the depth of the lagoon at three different points in each site every month, using a fixed graduated pole.

- **Water Quality**

Water temperature, salinity, turbidity, dissolved oxygen (D.O.), pH and conductivity were measured monthly at each site. Salinity, water temperature and D.O. were measured by means of a D.O. meter, Hanna oxy-Check H19147. Turbidity was measured using a HaCH 2100P Turbidimeter 46500-00. Conductivity was evaluated with a HaCH conductivity meter.

- **Habitat Change**

Satellite images of the MPRS were obtained from the Centre for Remote Sensing and Geographic Information Services (CERSGIS), University of Ghana for the years 1997, 2000, 2007.

3.3 Analysis of Data

Waterbird and benthic macroinvertebrate diversity were estimated using Primer v6 software. Shannon-Wiener, Simpson reciprocal (1-D), Pielou's evenness and Magarlef richness indices were used to assess species diversity for each site (Clarke and Warwick, 1994; Clarke and Gorley, 2006).

Feeding activity of each waterbird group (waders, terns, and herons) was defined as the number of feeding birds divided by the total number of birds observed in at that site (Liang *et al.*, 2002).

Similarity among sites based on the feeding activities and abundance of waterbirds, was assessed using Cluster Analysis based on the group average agglomerative dendrogram of Bray-Curtis similarity index in Primer software.

Primer software was also used to analyse benthic community assemblages by means of the Bray- Curtis similarity index. A fourth root transformation was applied to all biological data to minimise the effects of the dominant species prior to the similarity test. When dealing with spatial data, the samples were standardised by dividing them by the total. This was done because the sites were of different sizes. All hydrological data were $\log(x+1)$ -transformed to normalise them. The SIMPROF test at $p < 0.05$

was carried out to determine the presence or absence of structure, that gives statistical meaning to the groupings in the dendrogram. Based on the abundance and biomass of polychaetes found at each site (A, B, C and E) during the study period, a comparison was carried out using Abundance Biomass Comparison curves in Primer v6 (Clarke and Warwick, 1994; Clarke and Gorley, 2006).

Non-parametric tests were used when the data were proven not to be normally distributed. Here, only hydrological variables were $\log(x+1)$ -transformed. Spearman's correlation in SPSS 16.0 was used to determine bivariate correlations among hydrological variables; waterbirds (wader, heron, and tern) abundance and activities (roosting, feeding and comforting activities) and macrobenthic communities. The first part of the bivariate analysis took into consideration all sites apart from Sites A and D which were the only sites that were not sampled during the period December 2009 to April 2010. In order to analyse the difference in the impact of each factor at the different sites, data from Sites A, B, C, E and SP were used separately.

Another non-parametric test, Kruskal-Wallis, was used to determine the existence of any significant differences in the number of species of waterbirds among the sampling sites using PracticStat Software (Ashcroft and Pereira, 2003). Whenever significant, a post-hoc test, the Fisher Least Significant Difference (LSD) test was carried out to determine the sites that were significantly different from each other in terms of waterbird species.

Chi-square analysis (Payne *et al.*, 2006) (using GENSTAT 9th edition) was used to determine whether there was any association between sites and time with regards to bird abundance and their activity (roosting and feeding).

Multivariate analysis using Canonical Correspondence Analysis (CCA) CANOCO v 4.45 (Ter Braak and Smilauer, 2002) was carried out to test for any interrelationship between hydrological and benthic assemblages; and hydrological variables and waterbirds abundance and activity. Sites that had been sampled during the whole study period (Sites B, C, E) were selected for the CCA. Sites A, D and SP were therefore excluded, as well as variables like conductivity and D.O. since they were not sampled for all the eight months.

CHAPTER IV

RESULTS

4.1 Water Quality at the Muni-Pomadze Ramsar Site

4.1.1 Characteristics of the Muni Lagoon

Water samples were collected from the Muni lagoon over a period of eight months and analysed for the following parameters; water temperature, pH, turbidity, conductivity, salinity, and dissolved oxygen; the spatio-temporal variations of which are represented by the graphs below (Figure 4.1 to 4.6).

Table 4.1 presents a summary of the physico-chemical characteristics of the Muni lagoon, and also indicates the number of times (N) the water samples were analysed for the various parameters.

Table 4.1: Summary of Statistics of Physico-chemical Characteristics of Water Samples

WATER VARIABLES	MEAN VALUE ($\pm$ S.D.)	MIN	MAX	N
Water temperature (°C)	32.12 $\pm$ 3.67	25.20	39.50	35
Salinity (‰)	40.82 $\pm$ 2.95	20.20	106.00	35
Conductivity (mS/cm)	58.43 $\pm$ 0.73	38.60	74.80	30
Dissolved Oxygen (mg/L)	7.06 $\pm$ 1.40	5.50	10.60	26
pH	8.21 $\pm$ 0.28	7.43	8.70	35
Turbidity (NTU)	27.49 $\pm$ 20.06	5.04	89.30	35

N = number of samples analysed; S.D. = Standard Deviation

The mean temperature of the Muni lagoon recorded during the sampling period was 32.12°C (S.D. 3.67°C) with the minimum value recorded at Site E (25.2°C) in September 2009 and the maximum at Site B (39.5°C) in March 2010 (Table 4.1). The temperature was always above 25°C. A fall in temperature was however observed in November at all the sampled sites. Sites E and SP on one hand and site C and B on the other, had similar patterns in temperature variations from December to January (Appendix II).

The values of pH varied between 7.43 (SP in April) and 8.7 (B in November and E in December) and the calculated mean value was 8.21 (Table 4.1). A fall in pH was observed in Sites B, C, E and SP in January 2010. There was a similarity in the spatio-temporal variation pattern of pH in the Sites C, SP and B, between December 2009 and April 2010. A positive correlation ($r = 0.0489$, $P < 0.05$) was found between pH and temperature (Table 4.16). In Sites B and C, temperature was found to be highest when water depth was lowest in March and vice-versa in November in B (Appendix II).

Turbidity ranged from 5.02 NTU (in Site C in March 2010) to 89.3 NTU (in Site A in December 2009) with a mean value of 27.49 NTU. There was an increase in turbidity in November 2009 in all the sampled sites and a decrease in March 2010.

Mean salinity values of the Muni lagoon were high, with the highest salinity recorded as 106‰ in SP and the lowest, 20.20‰ in Site A (November). The mean salinity values were very high at Site SP. A drop in salinity was also observed in November in all sites sampled.

Conductivity of the Muni lagoon ranged from 38.6 mS/cm in Site E in December to 74.8 mS/cm in Site SP (December); with a mean value of 58.43 mS/cm. Conductivity values were high in Sites SP and B. There was a positive correlation between salinity and conductivity ($r = 0.524$, $P < 0.05$) (Table 4.2).

The mean dissolved oxygen was 7.6 mg/L. A peak was recorded in March at all sites sampled except site B whose D.O. rather increased in April. Site E and SP showed similar patterns in temporal variations of D.O. from December to April. The lowest D.O. values were recorded at Sites A and SP in December and the highest was recorded in Site B in April. Salinity and D.O. exhibited a negative relationship ($r = -0.448$, $P < 0.05$; Table 4.2).

Table 4.2: Interrelationships among Water Variables

Interacting variables	r	P
Salinity * D.O.	-0.448	0.048
Salinity * Conductivity	0.524	0.018
pH* Temperature	0.489	0.028

4.1.2 Spatio-temporal variations in physico-chemical parameters of the Muni Lagoon

The mean turbidity varied significantly among sites, with minimum mean value recorded at Site C (18.32 NTU) and the maximum at Site A (61.88 NTU). The temporal mean turbidity also varied among months, and ranged from 9.27 in March 2010 to 54.52 NTU in November 2009 (Figure 4.1).

The spatial variation in the mean temperature (Figure 4.2) was very little and ranged from 30.55°C (Site A) to 33.34°C (Site C). The temporal mean temperature values

varied among months with the maximum (36.12°C) in March 2010 and minimum (27.16°C) in September 2010 (Figure 4.2). A drop was observed in mean temperature in November 2009.

The spatio-temporal variations in the mean pH were very small, although a drop was seen in January 2010 (Figure 4.3).

The spatio-temporal variation in mean salinity showed that the lagoon is hypersaline. The lowest mean salinity values (34.88‰) were recorded in Site A and the highest values, 60.1‰ in Site SP. The mean temporal salinity values ranged from 30.55‰ in November 2009 to 53.25‰ in April 2010 (Figure 4.4).

Little variation was observed in the spatio-temporal mean values of dissolved oxygen at the Muni lagoon. Nevertheless, the spatial mean values ranged from 6 mg/L in D to 7.83 mg/L in C, and the temporal mean values ranged from 5.9 in September 2009 to 8.375 mg/L in April 2010 (Figure 4.5).

Like the mean salinity values, the mean conductivity was highest in Site SP. The month of January 2010 scored the highest temporal mean conductivity values during the sampling period (Figure 4.6).

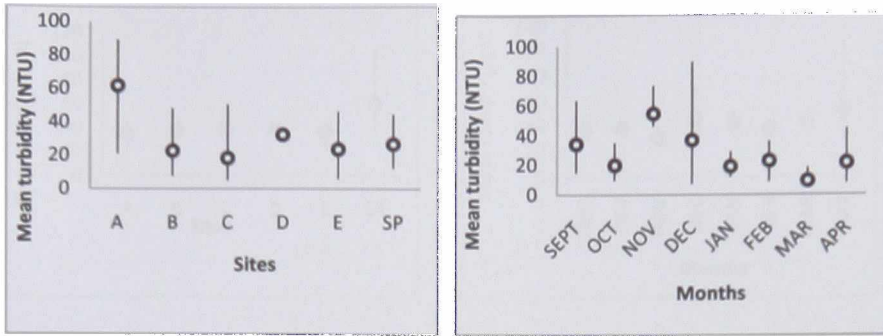


Figure 4.1: Spatio-temporal Variations in Mean Turbidity (NTU) at the Muni Lagoon over the Study Period (Sept 09-Apr 10)

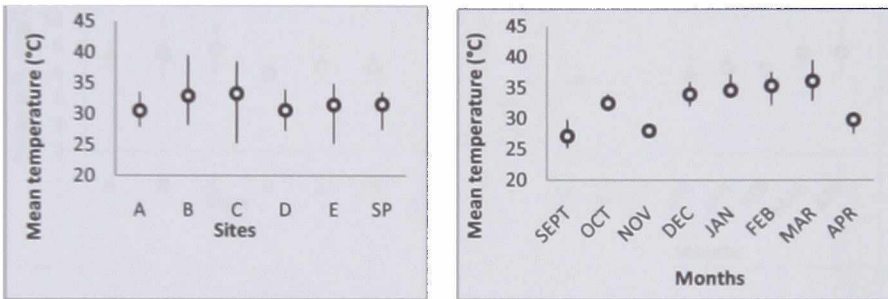


Figure 4.2: Spatio-temporal Variations in the Mean Temperature (°C) at the Muni Lagoon over the Study Period (Sept 09- Apr 10)

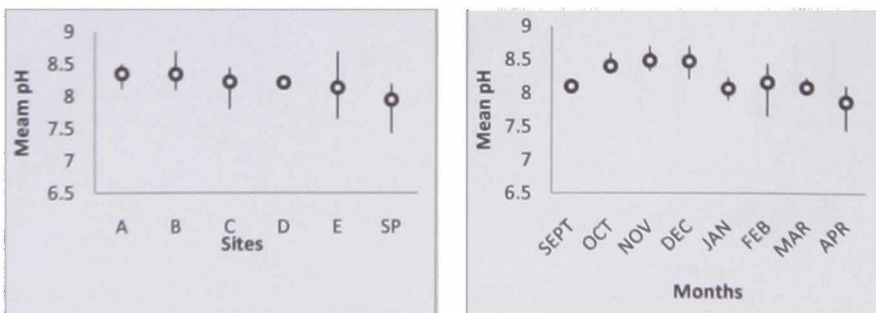


Figure 4.3: Spatio-temporal Variations in the Mean pH at the Muni Lagoon over the Study Period (Sept 09- Apr 10)

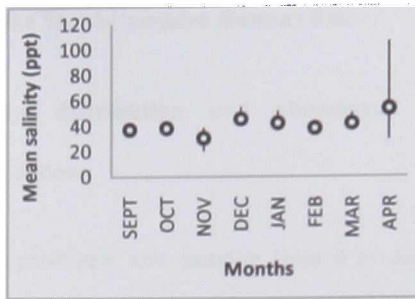
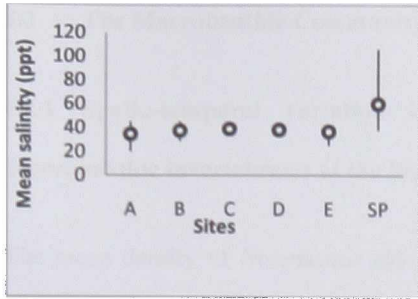


Figure 4.4: Spatio-temporal Variations in the Mean Salinity (%) at the Muni Lagoon over the study Period (Sept 09- Apr 10)

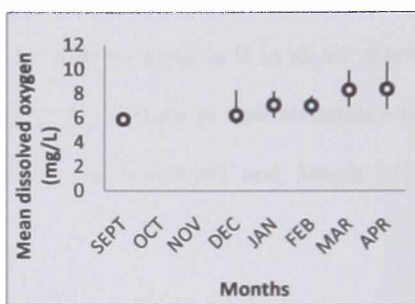
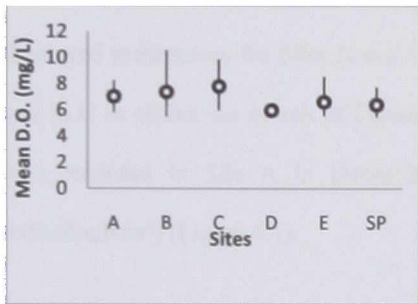


Figure 4.5: Spatio-temporal Variations in the Mean Dissolved Oxygen (mg/L) at the Muni Lagoon over the Study Period (Sept 09- Apr 10)

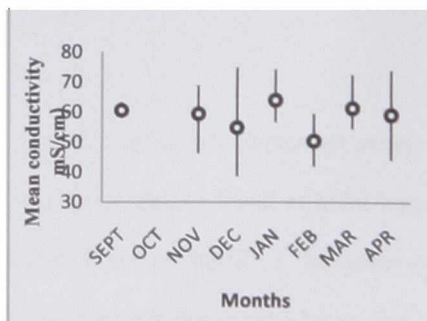
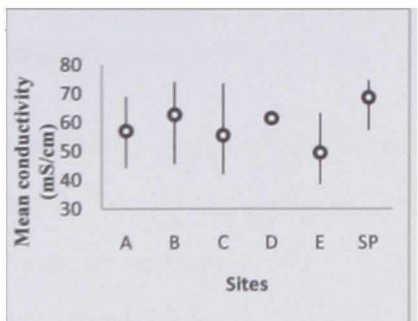


Figure 4.6: Spatio-temporal Variations in the Mean Conductivity (mS/cm) at the Muni Lagoon over the Study Period (Sept 09-Apr 10)

4.2 The Macrobenthic Community at the Muni-Pomadze Ramsar Site

4.2.1 Spatio-temporal variation in the distribution and abundance of Macrobenthic invertebrates at the Muni Lagoon

The mean density of *Notomastus* spp. was relatively low, ranging from 0 to 5381 individuals/m² during the study period; however, a peak was observed in most Sites (B, C, and E) in January 2010 (Figure 4.7). A peak in density was recorded earlier in November 2009 at the Site B. Apart from Site A, the density of *Notomastus* spp. in all other sites decreased in the month of October 2009. Furthermore, *Notomastus* spp. displayed preferences for Sites B and C in that they occurred in B in all the months and in C in all but the month of February 2010. An increase in their abundance was also recorded in Site A in December (3171 individuals/m²) and March (1219 individuals/m²) (Figure 4.7).

Capitella spp. occurred most in Sites A, B and C, but were most abundant in Site C (25,928.71 individuals/m²). A significant increase in density of *Capitella* spp. was observed in sites A (10,580 individuals/m²) and C (12,043 individuals/m²) in March and in site B in April (4817 individuals/m²). Like the *Notomastus* spp., a decrease was observed in October (Figure 4.8).

Leiochrides africanus in the Muni lagoon occurred at all sites in September except in Site A. This species in addition to all the remaining species found at Muni lagoon except the opportunistic *Capitella* spp. were not found at site SP. *L. africanus* was most abundant in Site B (2499.26 individuals/m²) where it also occurred most (six out of eight) (Figure 4.9).

Nephtys spp. were not encountered in October and March in all sites. Though very few at all sites, they were still most abundant in Sites A (534 individuals/m²) and B (727 individuals/m²) (Figure 4.10).

The spatio-temporal variation in the abundance and distribution of the *Neiris* spp. at the Muni lagoon was characterised by a shift in abundance from Site D (76 individuals/m²) in September to Site A (76 individuals/m²) in December 2009, E (125) in January 2010 and Site B (76) in March 2010. They were however very low in abundance (Figure 4.11).



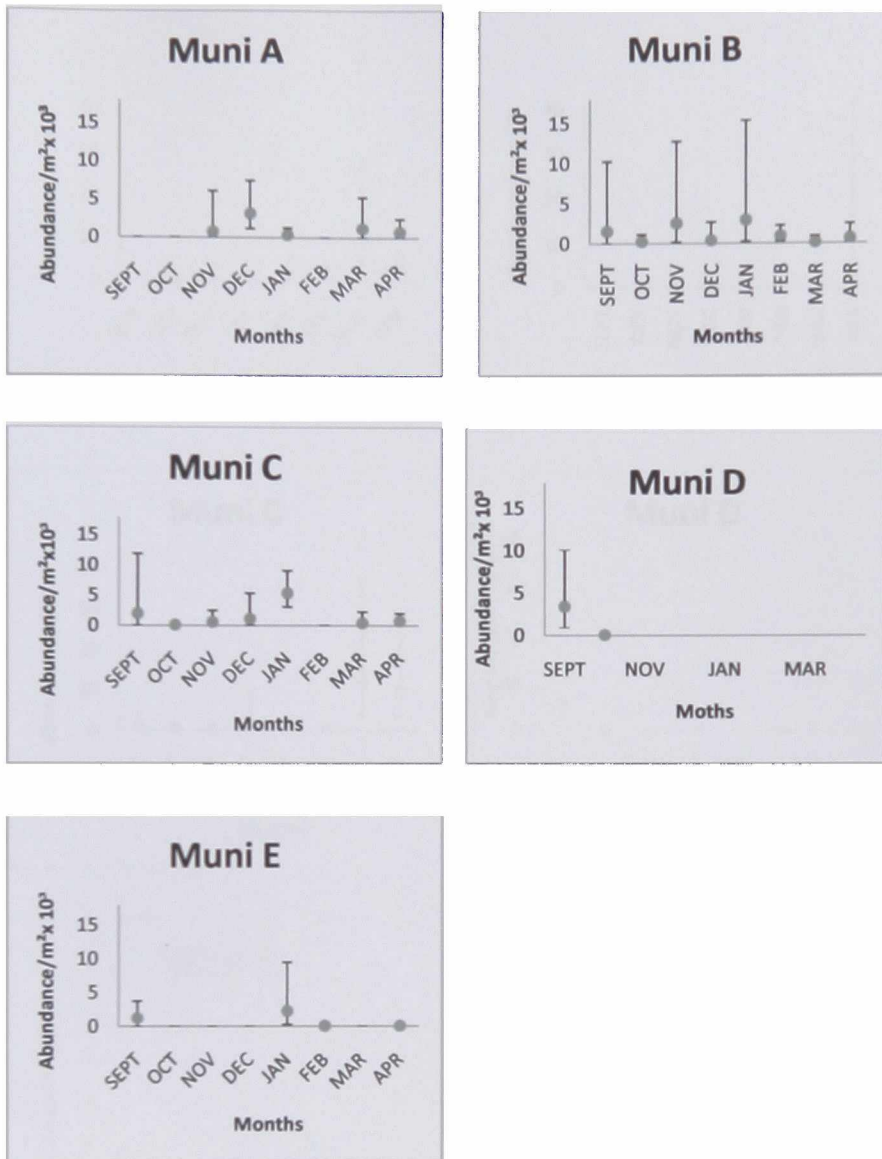


Figure 4.7: Spatio-temporal Variations in the Density of *Notomastus* spp. at the Muni Lagoon over the Study Period (Sept.09-Apr.10)

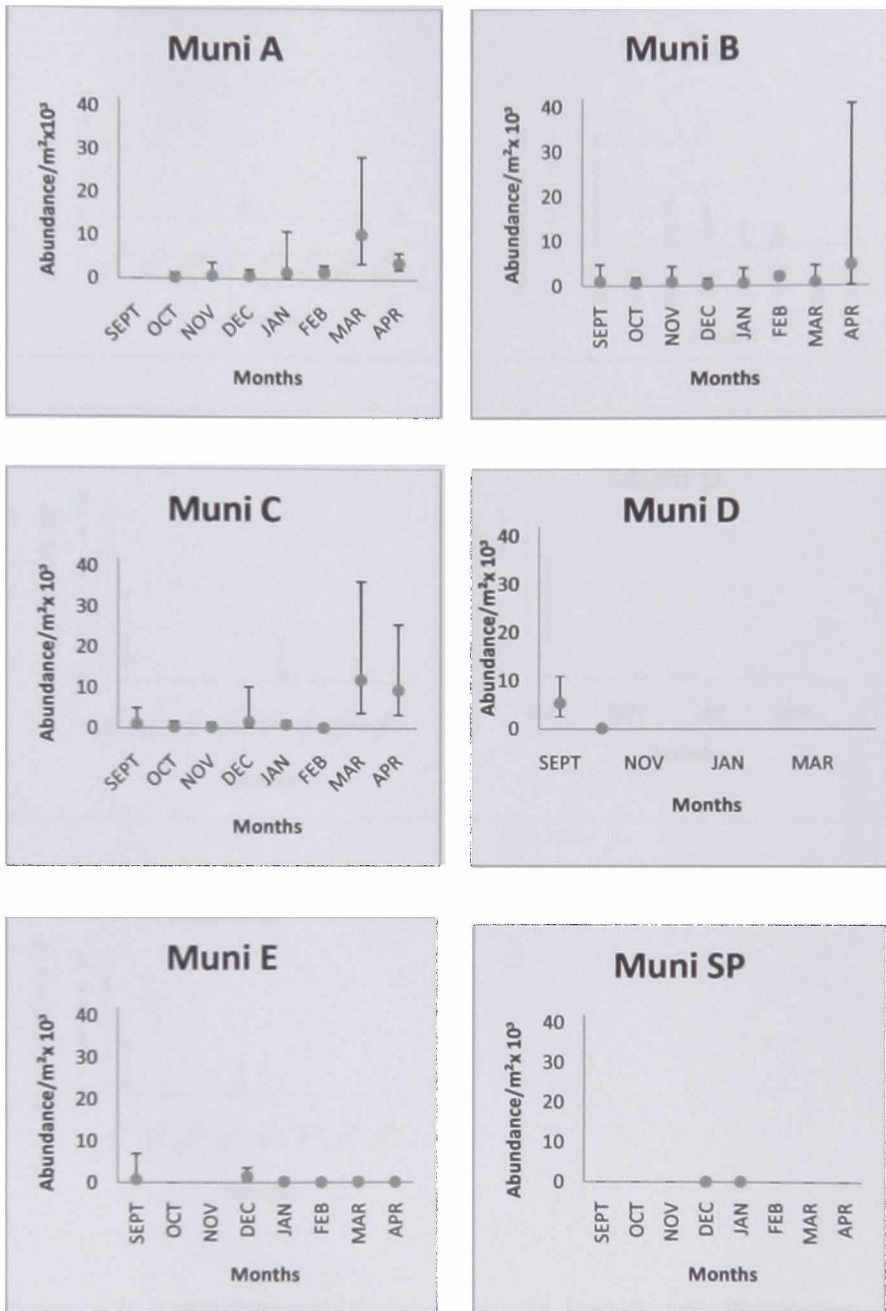


Figure 4.8: Spatio-temporal Variations in the Density of *Capitella* spp. at the Muni Lagoon over the Study Period (Sept. 09-Apr.10).

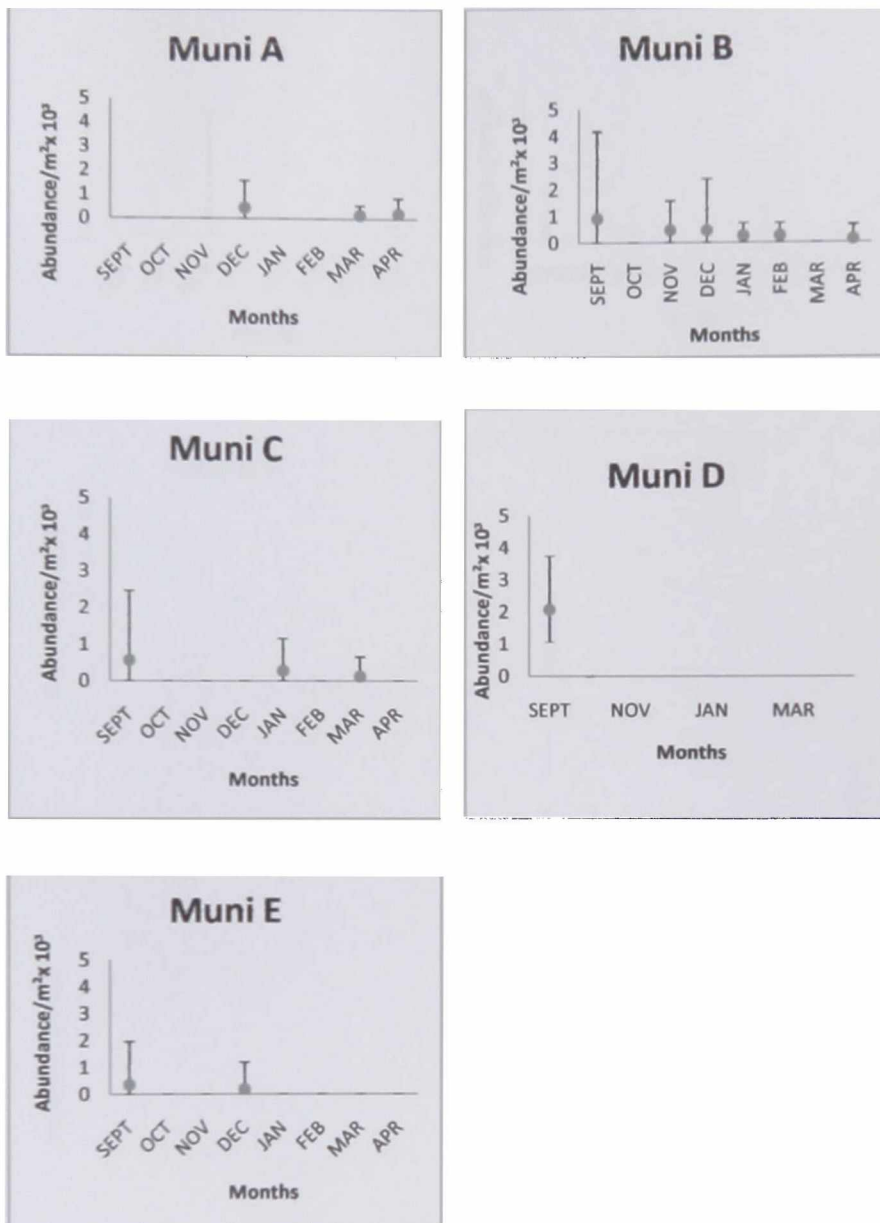


Figure 4.9: Spatio-temporal Variations in the Density and Distribution of *Leiochrides africanus* at Muni Lagoon over the Study Period (Sept.09-Apr.10)

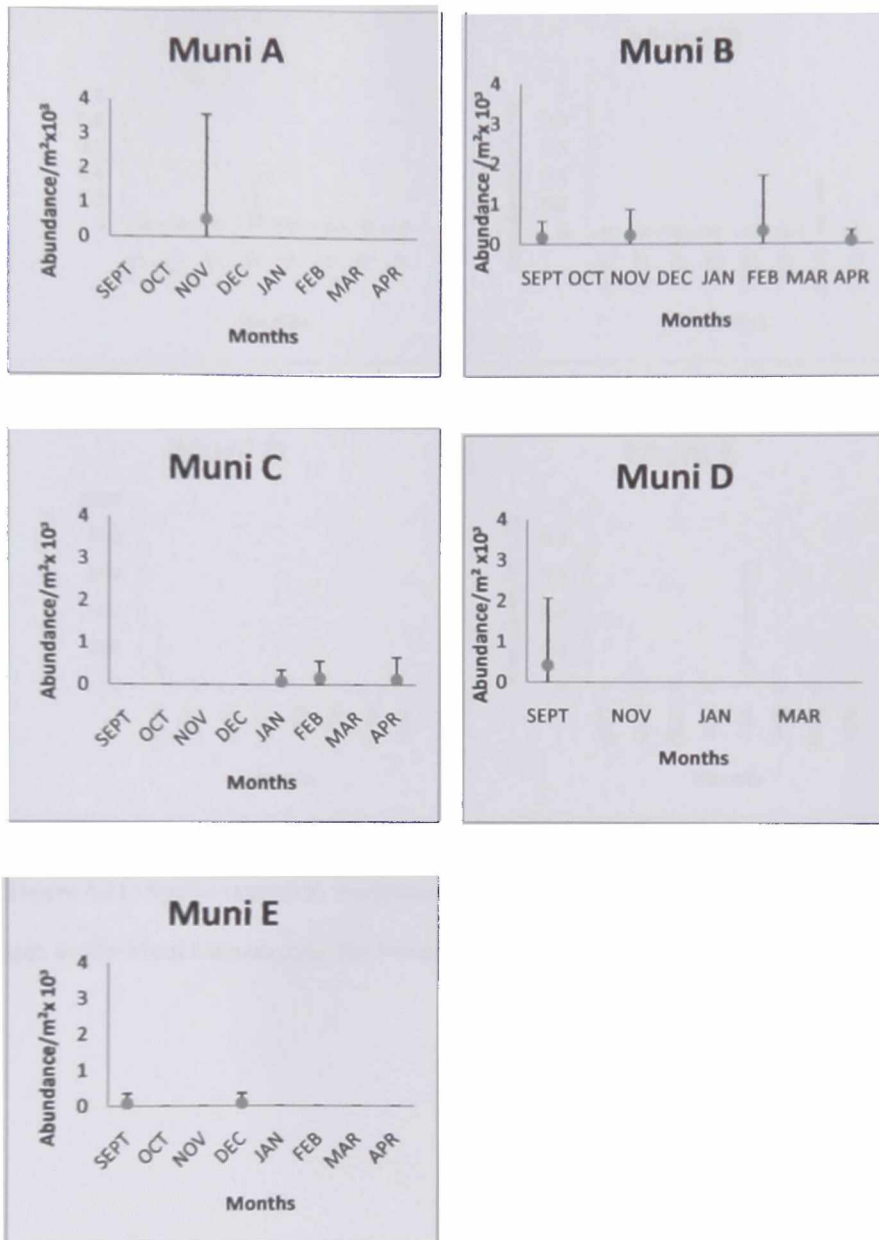


Figure 4.10: Spatio-temporal Variations in the Distribution and Density of *Nephtys* spp. at the Muni Lagoon over the Study Period (Sept.09- Apr.10)

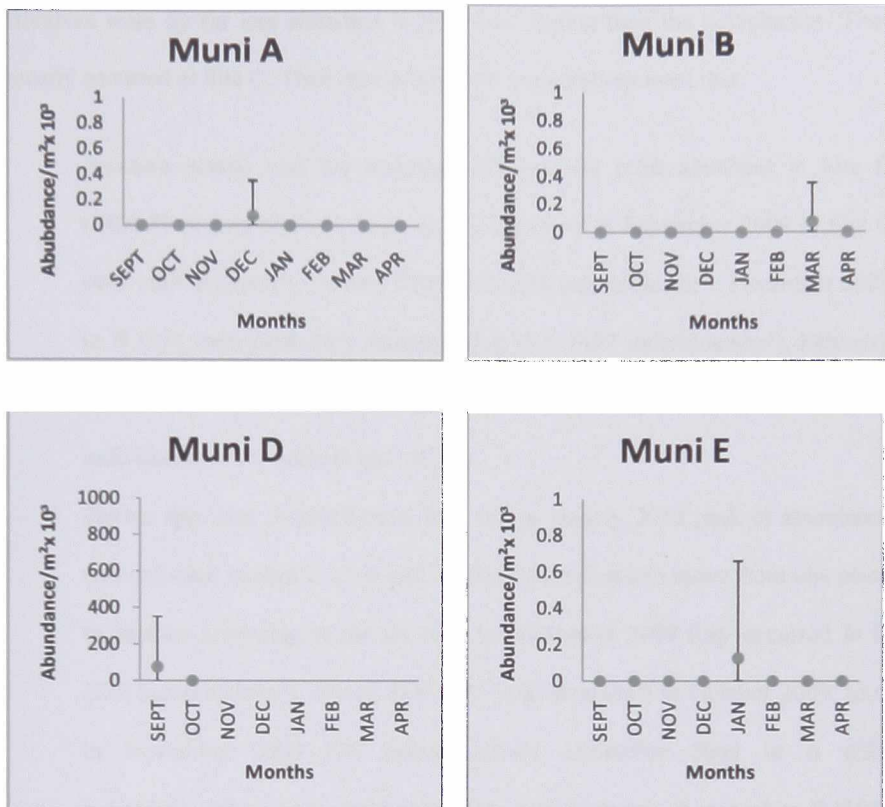


Figure 4.11: Spatio-temporal Variations in the Distribution and Density of *Neiris* spp. at the Muni Lagoon over the Study Period (Sept.09-Apr. 10)

Bivalves were by far less abundant in the Muni lagoon than the polychaetes. They mostly occurred at Site C. Their spatio-temporal variations showed that:

- a. *Anadara senilis* was the dominant bivalve and most abundant in Site C (1401.89 individuals/m²). They were found most in September 2009 in Site C (445 individuals/m²), October 2009 in D (378 individuals/m²), December 2009 in B (399 individuals/m²), January 2010 in C (497 individuals/m²), February 2010 in E (281 individuals/m²) and March, April 2010 in C (194 individuals/m² for both) (Figure 4.12).
- b. *Tellina* spp. also occurred most in C and a January 2010 peak in abundance was recorded in sites E, C, B and A. They were found to move from one place to another according to the months. In September 2009 they occurred in D (220 individuals/m²), moved to B (125 individuals/m²) in October 2009, to C in November 2009 (76 individuals/m²), December 2009 in A (220 individuals/m²), January 2010 in E (744 individuals/m²), then back to C (194-220-163) from February to April 2010 (Figure 4.13). Other bivalves were also encountered in B in January and February 2010 (Figure 4.14).
- c. The Amphipods were recorded only a few times in the Muni Lagoon, especially at Sites A and B. They were most abundant in Site A (263) in February and B (243) in March 2010 (Figure 4.14). Other crustaceans such as the *Uca* spp. were sampled from February to April 2010 in Sites A (twice) and C (once) (Figure 4.15).



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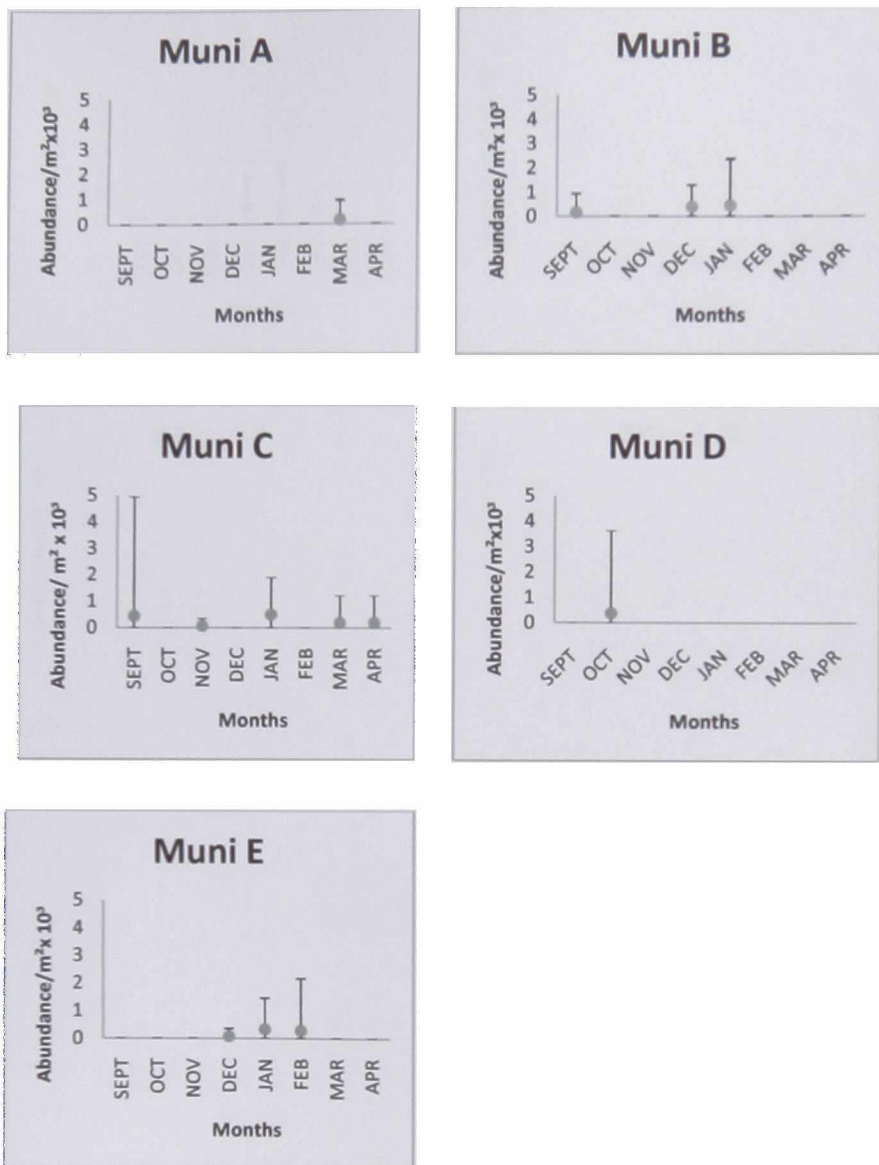


Figure 4.12: Spatio-temporal Variations in the Distribution and Density of *Anadara senilis* at the Muni Lagoon over the Study Period (Sept.09-Apr.10)

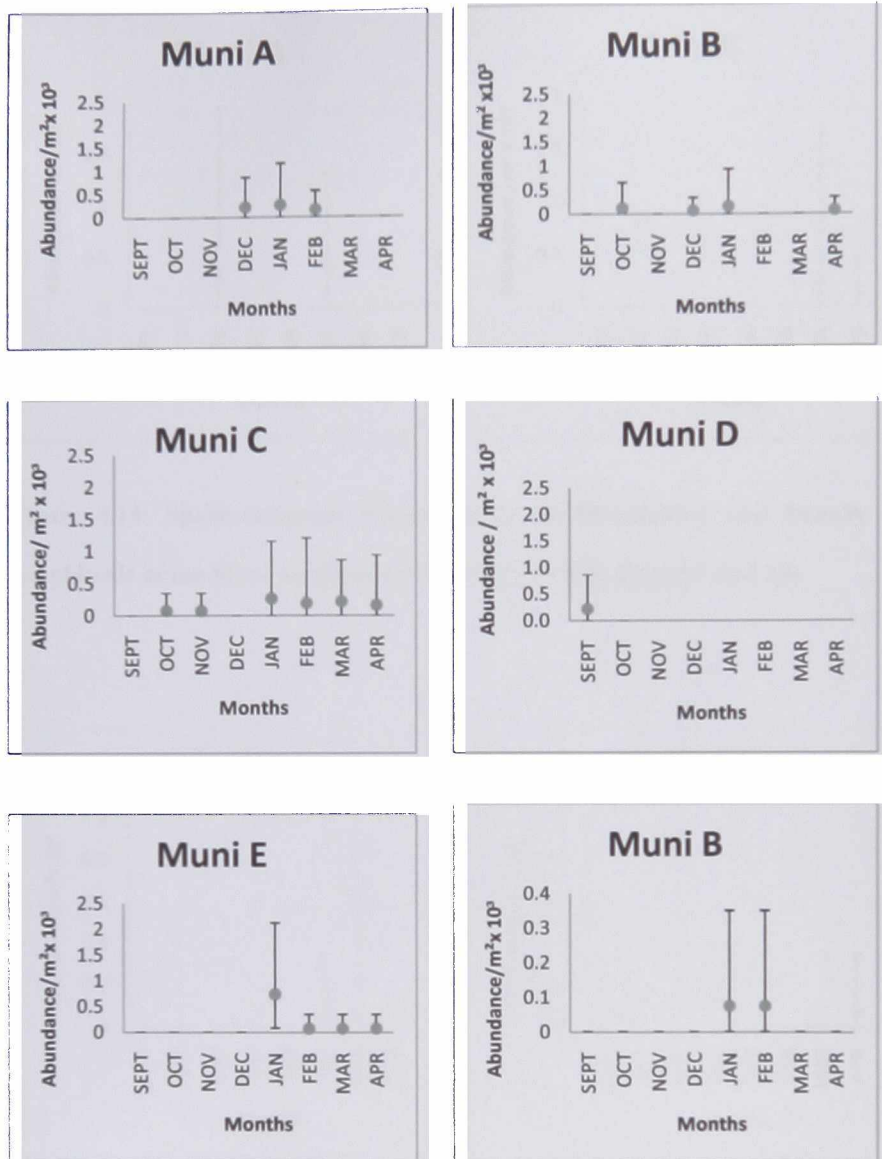


Figure 4.13: Spatio-temporal Variations in the Distribution and Density of *Tellina* spp. and other Bivalves (last figure on the right) at the Muni Lagoon over the Study Period (Sept.09-Apr.10).

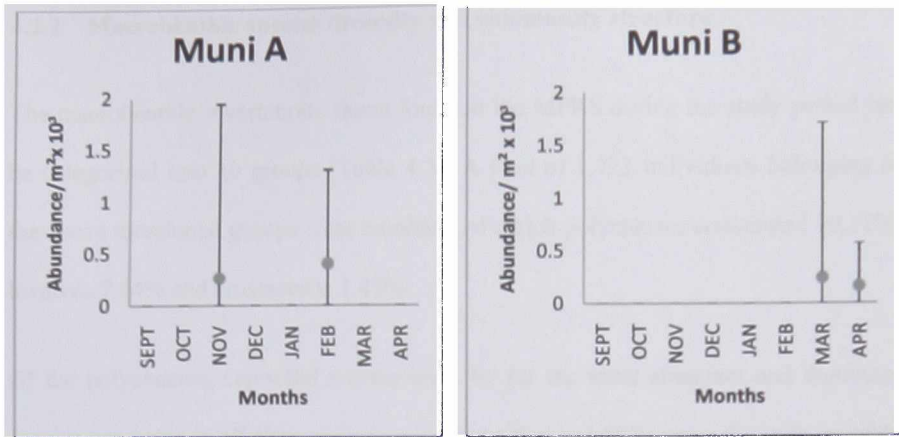


Figure 4.14: Spatio-temporal Variations in the Distribution and Density of Amphipods at the Muni Lagoon over the Study Period (Sept.09-Apr.10).

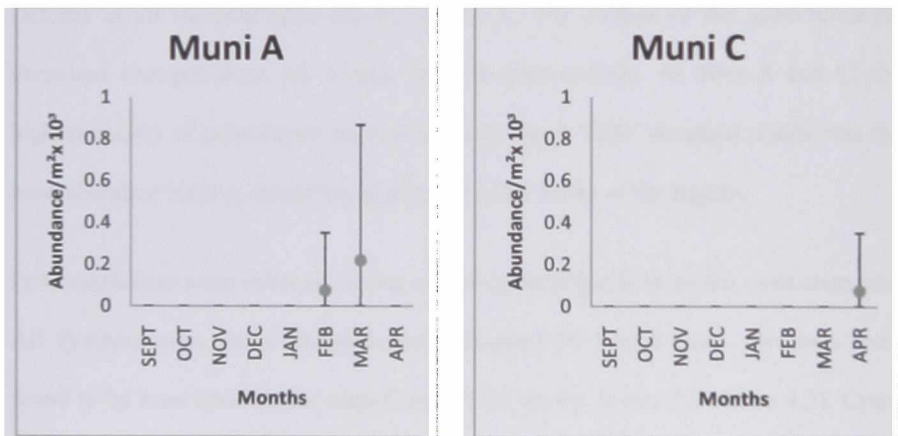


Figure 4.15: Spatio-temporal Variation in the Distribution and Density of *Uca* spp. at the Muni Lagoon over the Study Period (Sept.09-Apr.10).

4.2.2 Macrobenthic species diversity and community structure

The macrobenthic invertebrate fauna found at the MPRS during the study period can be categorised into 10 groups (Table 4.3). A total of 1,723 individuals belonging to the above mentioned groups were recorded, of which polychaetes constituted 90.71%, bivalves 7.84% and crustaceans 1.45%.

Of the polychaetes, capitellid worms were by far the most abundant and dominant species occurring in all sites, even in conditions that would be normally unfavourable for macrobenthos. *Capitella* spp. was the only species found at Site SP where the mean salinity was 50.6‰ and could reach up to 106‰. Some species were absent in specific areas. For example, *Nephtys* spp. were not found in Site A, and *Neiris* spp. were not found in the C. Polychaetes of the Muni lagoon were seen to decrease in October at all sampled sites except in Site A. The pattern of the spatiotemporal variations changed from site to site and was time-specific. At Sites A and C, the highest density of polychaetes was recorded in March 2010. *Anadara senilis* was the most abundant bivalve, occurring most often at the banks of the lagoon.

Few crustaceans were collected during sampling, amphipods being the most abundant. All *Tympanotonus* shells sampled were inhabited by hermit crabs. Bivalves were found to be most abundant at Sites C and B but absent at Site SP (Table 4.3). Crabs (e.g. *Callinectes* spp, etc.) were observed at the Muni Lagoon except *Uca* spp. Crustaceans were only sampled in Sites A, B and C.

Family diversity, richness and evenness were estimated using Shannon (H') and Simpson ($1-D$), Magarlef (d'), and Pielou's evenness (J') respectively. Spatially, these indices (Table 4.4) showed that Sites A (0.78) and B (0.79) were the richest in terms of family (d'). Sites E ($j' = 0.75$) had the most evenly-distributed benthic community and A ($j'=0.47$) the least. Shannon and Simpson diversity indices showed that Site E ($H'= 1.45$, $1-D= 0.71$) was the most diverse, followed by Site B. Only *Capitella* spp. were recorded in Site SP ($H' = 0.0$) making it the least diverse site.

Table 4.4: Diversity Indices of Macrobenthic Invertebrates at the Muni Lagoon

Diversity indices	Sampling Sites					
	A	B	C	D	E	SP
Number of species (S)	9	9	7	7	7	1
Abundance (Mean/m²)	29119.4	26226	40230.5	12266	9349.22	250.65
Magarlef richness (d)	0.78	0.79	0.57	0.64	0.66	0
Pielou's evenness (j')	0.47	0.62	0.51	0.69	0.75	0
Shannon- wiener (H')	1.04	1.37	0.99	1.34	1.45	0
Simpson (1-D)	0.5	0.67	0.51	0.68	0.71	0

Table 4.5 indicates the temporal variations in macroinvertebrate faunal assemblages of the Muni lagoon. Family richness was higher in February ($d' = 0.9$) and lowest in October ($d' = 0.38$). Benthic invertebrates were most evenly distributed in October ($J' = 0.8$) and the least in March ($J' = 0.26$). Shannon diversity index showed that macrobenthic community was the most diverse ($H' = 1.45$) in February whereas Simpson index showed that macrobenthic community was rather more diverse in September ($1-D = 0.68$). The lowest diversity was recorded in March ($H' = 0.55$; $1-D = 0.22$).

Table 4.5: Temporal Variations in Macrobenthic Community Structure

Sampling periods (2009-2010)	Diversity Indices					
	Number of species (S)	Abundance (mean /m ²)	Magarlef richness (d)	Pielou's evenness (j')	Shannon-wiener (H')	Simpson (1-D)
SEPT	7	22063.1	0.6	0.67	1.31	0.68
OCT	4	2509.52	0.38	0.8	1.11	0.6
NOV	7	7528.31	0.67	0.67	1.3	0.64
DEC	7	11145.7	0.64	0.65	1.26	0.65
JAN	8	18052.5	0.71	0.57	1.19	0.58
FEB	9	7233.13	0.9	0.66	1.45	0.63
MAR	8	27010.8	0.69	0.26	0.55	0.22
APR	8	21898.9	0.7	0.31	0.65	0.29

Macrobenthic invertebrates were abundant at the Muni lagoon (Figure 4.16). The highest density was recorded at Site C and the lowest at Site SP. The density varied greatly with time. It decreased drastically in October and February, and increased in January and March (when it reached its highest peak, Figure 4.16).

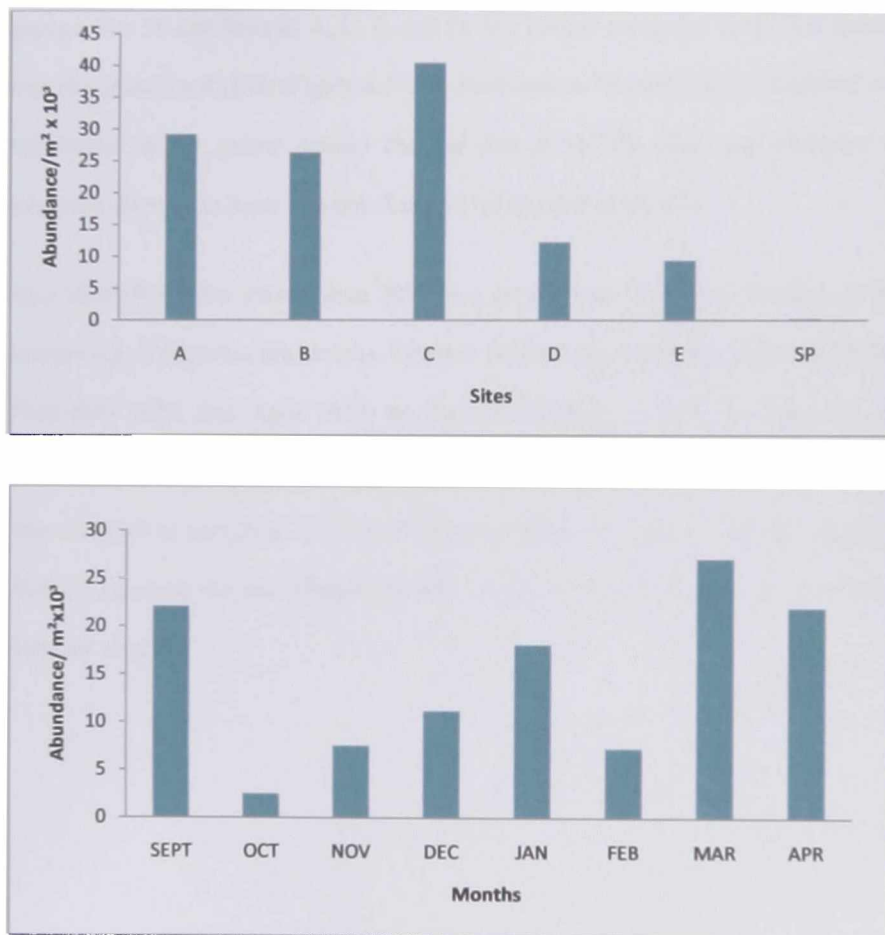


Figure 4.16: Spatio-temporal Variations in the Density of Macroinvertebrates at MPRS over the Study Period (Sept.09-Apr.10).

4.2.3 Macrobenthic invertebrate community structure

The agglomerative dendrogram based on the spatial density of macroinvertebrates found at the Muni lagoon over the study period indicated that at a similarity index greater than 20% the different sections of the Muni lagoon can be categorised into two groups, Site SP and Sites B, A, C, E, and D. The second group can further be divided into two (BAC) and (ED) (Figure 4.17.A). Furthermore The SIMPROF (PRIMER v6) test based on the spatial density showed that at 16.25% there was evidence of structure, though the same was not observed for the split at 80.62%.

At a similarity index greater than 70%, two groups were identified in terms of the seasonality of macrobenthic density, October 2009 on one hand and (September 2009, November 2009, and April 2010) on the other (Figure 4.17.B). Nevertheless, the SIMPROF test ($p < 0.05$) showed that at 62.3% similarity, $\pi > p$, indicating that there was evidence of structure. It however failed to show any evidence for the apparent division based on the macrobenthic faunal density at 74.02% Bray- Curtis similarity between months.

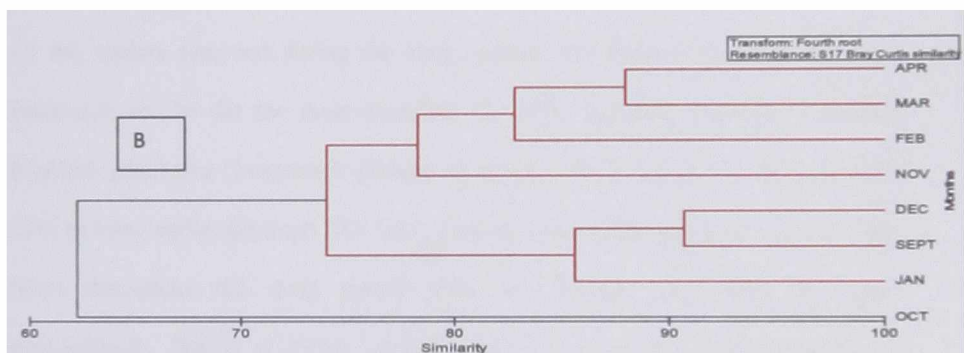
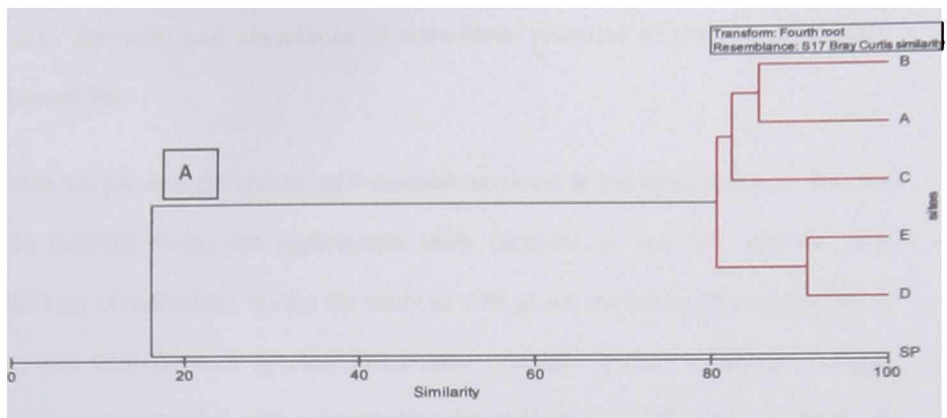


Figure 4.17: Group Average Agglomerative Dendograms of Bray-Curtis Similarity of Macrobenthic Faunal Abundance Data for the Six Sampling Sites over the Eight Months Study Period. (A) indicates dendrogram based on sampling sites whereas (B) indicates the one based on monthly variations. SIMPROF test, $p < 0.05$.

4.3 Waterbirds at the Muni-Pomadze Ramsar Site

4.3.1 Diversity and abundance of waterbirds recorded at the Muni-Pomadze Ramsar Site

Table 4.6 presents the species of waterbirds recorded at the Muni-Pomadze Ramsar Site (MPRS) during the eight-month study (Sept-09 to Apr.-10), and the total sightings of individuals during the study as well as the maximum count recorded at any one time for each species. Thirty-three waterbird species belonging to eight families were recorded, with a cumulative total sighting of 14,342 individual birds, of which waders represented 67%, terns 28% and herons 5%.

Of the waders recorded during the study period, the Ringed Plover (*Charadrius hiaticula*) was by far the most abundant (2, 517). Together with the Sanderling (*Calidris alba*) and Greenshank (*Tringa nebularia*), the three species accounted for 60% of total wader sightings. The least common species (encountered less than five times throughout the study period) were the Senegal Thick-Knee (*Burhinus senegalensis*), Dunlin (*Calidris alpina*), Spotted Redshank (*Tringa erythropus*), Kittlitz's Plover (*Charadrius pecuarius*) and Collared Pratincole (*Glareola pratincola*).

The commonest terns were the Sandwich (*Sterna sandvicensis*) and Royal (*Sterna maxima*) Terns, representing 70.86% of the tern sightings. Although the Roseate Tern (*S. dougalii*) was not recorded during the regular survey periods, seven individuals were sighted at a high tide roost on the site on one occasion.

Other waterbirds commonly present on the site were Little Egrets (*Egretta garzetta*) and Reef Herons (*Egretta gularis*). The maximum numbers of the different groups of

waterbirds recorded at any one time on the site were 1,070 waders (January 2010 at low tide and February 2010 at high tide), 881 terns (October 2009) and 145 herons (September 2009). The maximum numbers of Ringed Plover, Sanderling and Greenshank individuals recorded at any one time were 413, 308 and 187 respectively. Of the terns and herons, those with the highest counts at any one time were the Common Tern (*Sterna hirundo*) (676) and Little Egret (51).

Based on the maximum counts recorded at any one time, none of the waterbird species occurred in internationally-significant numbers. Nonetheless, five of the commonly-sighted species at Muni occurred in nationally significant numbers: Spur-Winged Plover (*Vanellus spinosus*), Whimbrel (*Numenius phaeopus*), Wood Sandpiper (*Tringa glareola*), Common Sandpiper (*Actitis hypoleucos*), and the Sanderling. These species represent 5% or more of the peak total Ghana coastal count (Ntiamoa-Baidu *et al.*, 2000a).

Table 4.6: Waterbird Species and Relative Abundance at the Muni Lagoon (September 2009/April 2010).

Common name	Scientific name	Total sightings	Maximum recorded at any one time at Muni
Waders			
Senegal Thick-Knee	<i>Burhinus senegalensis</i>	4	2
Collared Pratincole	<i>Glareola pratincola</i>	66	52
Spur-Winged Plover	<i>Vanellus spinosus*</i>	159	36
Ruddy Turnstone	<i>Arenaria interpres</i>	57	14
Kittlitz's Plover	<i>Charadrius pecuarius</i>	8	3
Common Ringed Plover	<i>Charadrius hiaticula</i>	2517	413
White-Fronted Plover	<i>Charadrius marginatus</i>	63	13
Grey Plover	<i>Pluvialis squatarola</i>	255	39
Whimbrel	<i>Numenius phaeopus*</i>	262	34
Wood Sandpiper	<i>Tringa glareola*</i>	116	56
Common Sandpiper	<i>Actitis hypoleucos*</i>	280	57
Common Redshank	<i>Tringa tetanus</i>	44	5
Dunlin	<i>Calidris alpina</i>	1	1
Bar- Tailed Godwit	<i>Limosa lapponica</i>	36	7
Little Stint	<i>Calidris minuta</i>	612	113
Sanderling	<i>Calidris alba*</i>	1748	308
Curlew Sandpiper	<i>Calidris ferruginea</i>	934	212
Black-Winged Stilt	<i>Himantopus himantopus</i>	933	103
Spotted Redshank	<i>Tringa erythropus</i>	11	5
Common Greenshank	<i>Tringa nebularia</i>	1478	187
Marsh Sandpiper	<i>Tringa stagnatilis</i>	39	10
Herons, Egrets, & others			
Grey Heron	<i>Ardea cinerea</i>	51	12
Squacco Heron	<i>Ardeola ralloides</i>	1	1
Western Reef Egret	<i>Egretta gularis</i>	208	32
Little Egret	<i>Egretta garzetta</i>	383	51
Great Egret	<i>Egretta alba</i>	6	2
Green-Backed Heron	<i>Butorides striata</i>	3	1
Long-Tailed Cormorant	<i>Phalacrocorax africanus</i>	72	68
Terns			
Common Tern	<i>Sterna hirundo</i>	1000	676
Sandwich Tern	<i>Sterna sandwicensis</i>	1376	386
Royal Tern	<i>Sterna maxima</i>	1455	270
Little Tern	<i>Sterna albifrons</i>	72	62
Black Tern	<i>Chlidonias niger</i>	92	50
* Nationally significant numbers			

4.3.2 Tidal influence on waterbird abundance at the Muni-Pomadze Ramsar Site

Between December 2009 and April 2010, two counts were undertaken on each day of the survey at low and high tide to assess the best time for carrying out waterbird counts at the MPRS. Table 4.7 shows the total sightings of each individual species of waterbirds recorded during the low and high tide counts. The total sightings of waterbirds were higher at high tide from December 2009 to February 2010 then lower in March and April 2010.

Some species like the Common Tern and Kittlitz's Plover were recorded only at low tide. Others such as the Sanderling, Curlew Sandpiper (*Calidris ferruginea*), Royal Tern, Spotted Redshank and Grey Heron (*Ardea cinerea*) were most abundant at high tide, while the Ringed-Plover and Little Stint (*C. minuta*) were more abundant at low tide.

Table 4.7: Tidal Influence on the Relative Abundance of Waterbirds at the Muni Lagoon (December 2009/April 2010).

Common name	Scientific name	Total sightings at low tide	Total sightings at high tide
Waders			
Senegal Thick-Knee	<i>Burhinus senegalensis</i>	2	2
Collared Pratincole	<i>Glareola pratincola</i>	55	11
Spur-Winged Plover	<i>Vanellus spinosus</i>	77	38
Ruddy Turnstone	<i>Arenaria interpres</i>	31	8
Kittlitz's Plover	<i>Charadrius pecuarius</i>	6	0
Common Ringed Plover	<i>Charadrius hiaticula</i>	1431	736
White-Fronted Plover	<i>Charadrius marginatus</i>	36	14
Grey Plover	<i>Pluvialis squatarola</i>	73	111
Whimbrel	<i>Numenius phaeopus</i>	95	117
Wood Sandpiper	<i>Tringa glareola</i>	72	39
Common Sandpiper	<i>Actitis hypoleucos</i>	123	71
Common Redshank	<i>Tringa tetanus</i>	20	12
Bar- Tailed Godwit	<i>Limosa lapponica</i>	17	10
Little Stint	<i>Calidris minuta</i>	370	158
Sanderling	<i>Calidris alba</i>	592	926
Curlew Sandpiper	<i>Calidris ferruginea</i>	254	633
Black-Winged Stilt	<i>Himantopus himantopus</i>	396	315
Spotted Redshank	<i>Tringa erythropus</i>	1	10
Common Greenshank	<i>Tringa nebularia</i>	572	599
Marsh Sandpiper	<i>Tringa stagnatilis</i>	14	9
Hérons, Egrets, etc.			
Grey Heron	<i>Ardea cinerea</i>	15	36
Western Reef Egret	<i>Egretta gularis</i>	82	78
Little Egret	<i>Egretta garzetta</i>	161	154
Great Egret	<i>Egretta alba</i>	2	2
Green-Backed Heron	<i>Butorides striata</i>	1	2
Terns			
Common Tern	<i>Sterna hirundo</i>	36	0
Sandwich Tern	<i>Sterna sandvicensis</i>	502	459
Royal Tern	<i>Sterna maxima</i>	468	729
Little Tern	<i>Sterna albifrons</i>	4	3

Figure 4.18 shows the relationship between high and low tide wader counts, with significant linear correlations for Guild 3 ($r = 0.53$, $P < 0.05$; $df = 17$) and Guild 4 ($r = 0.84$; $P < 0.05$, $df = 15$), and an insignificant correlation for Guild 2 ($r = 0.706$, $P > 0.05$, $df = 47$). It therefore means that waterbird species belonging to Guild 2 displayed significant tidal preferences at the MPRS, as their numbers increase at low tide.

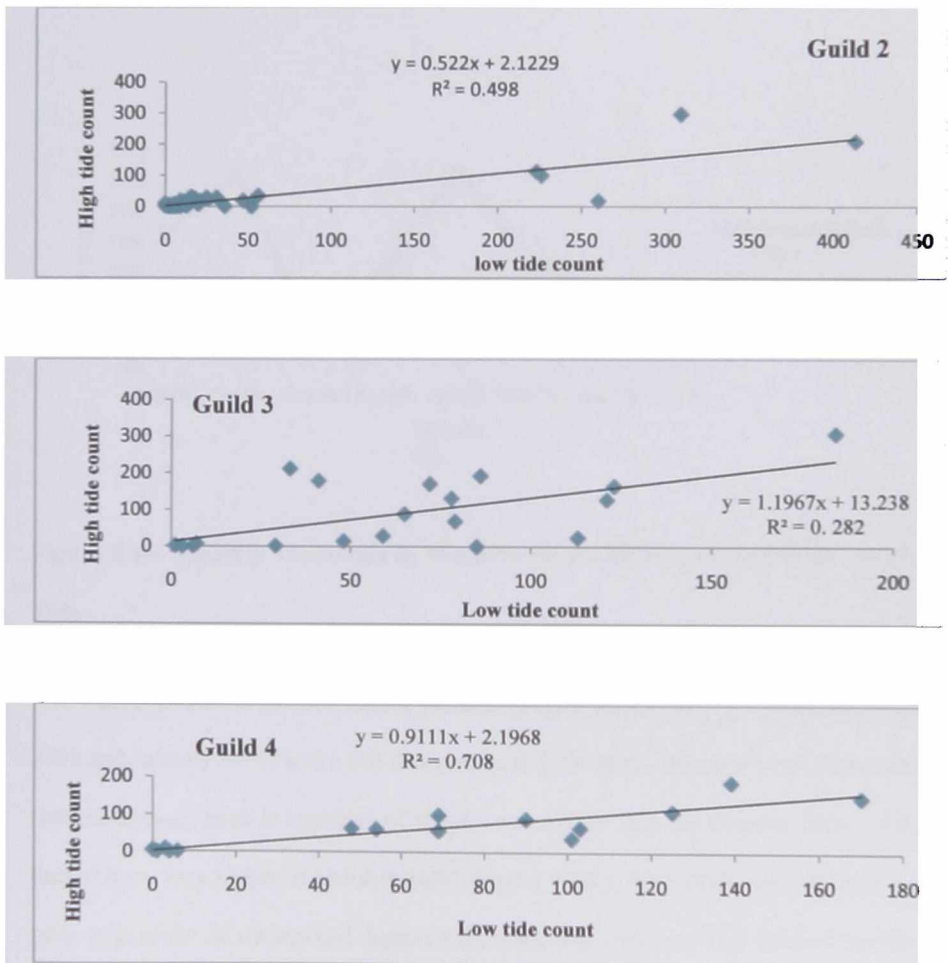


Figure 4.18: Correlation between High and Low Tide Counts of Waterbirds found at MPRS belonging to Guilds 2, 3 and 4

4.3.3 Seasonal patterns in waterbird occurrence at the Muni-Pomadze Ramsar Site

Figure 4.19 shows the monthly counts of waterbirds at the Muni Pomadze Ramsar Site. Waterbird numbers increased from September 2009, reaching a peak in October 2009; dropping in November 2009 and rising again in December 2009 to reach a second peak in January 2010.

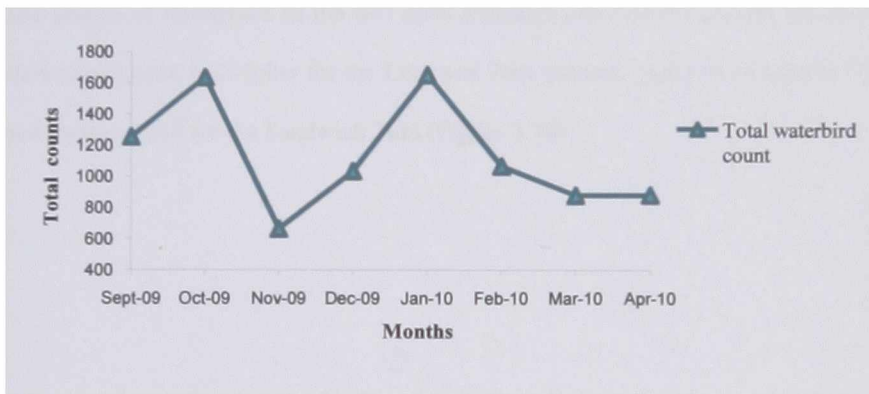


Figure 4.19: Monthly Variations in Waterbirds Abundance at the MPRs at Low Tide

The waders and terns showed similar patterns in their occurrence; two peaks (October 2009 and January 2010) and a fall in numbers in November. It was however observed that the January peak in numbers of waders was higher than the October peak, while that of terns was higher in October 2009 (Figure 4.20a). By March, over 30% of the peak population of waders had departed from the site with less than 20% of the tern population left at that period. The population of herons remained fairly stable

throughout the study period; the highest numbers counted were always less than 100, and there was no clear pattern in their occurrence (Figure 4.20c).

Similar seasonal patterns were shown by the most abundant waders (Ringed Plover, Sanderling and Greenshank) and terns (Sandwich and the Common Terns) at the MPRS. Ringed Plover and Sanderling numbers peaked in October and January (Figure 4.20a). The population of Greenshanks peaked in January from a decrease in December, and fell in numbers in March 2010.

The pattern of abundance of the two most abundant terns (Common and Sandwich) showed one peak in October for the Common Tern and two peaks in November 2009 and January 2010 for the Sandwich Tern (Figure 4.19b).

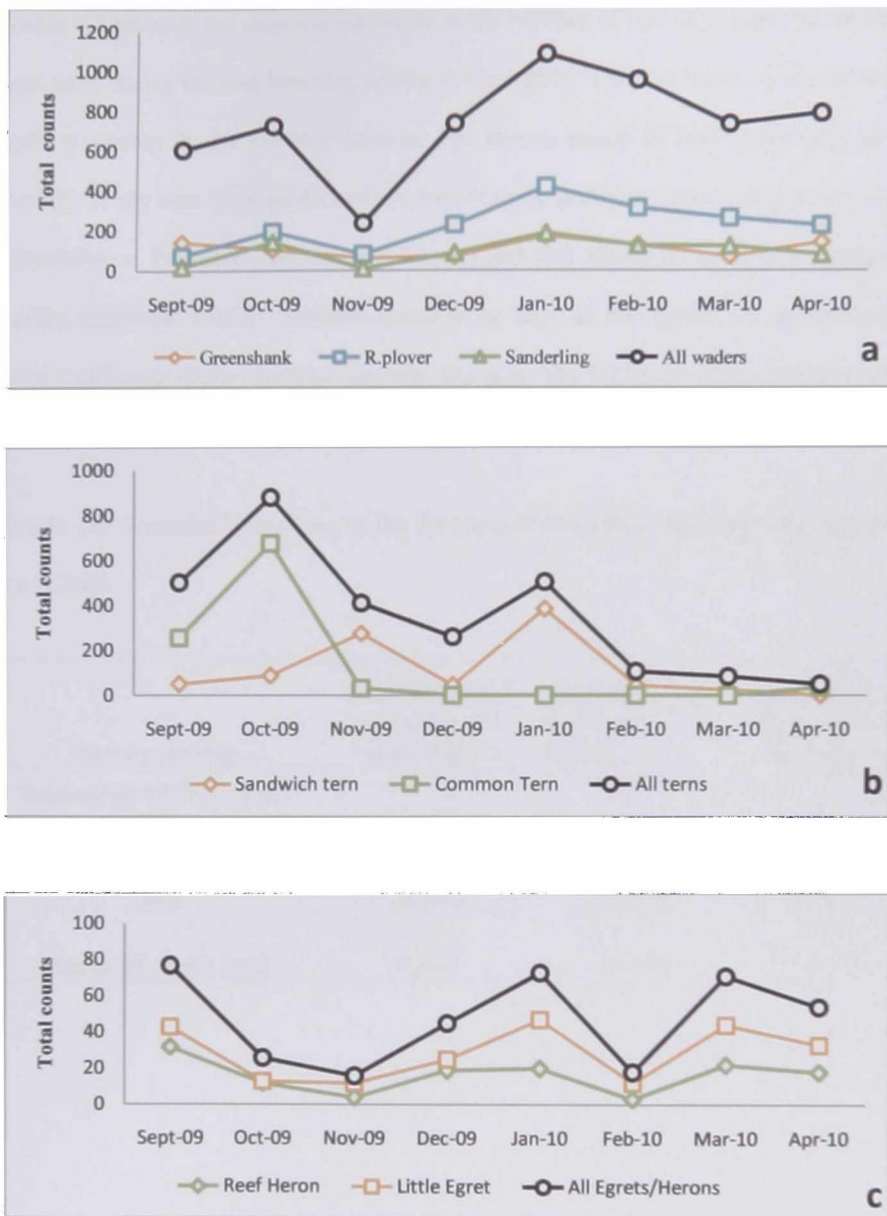


Figure 4.20: Seasonal Variations in the Abundance of the Commonest Waders (a), Terns (b), and Herons (c).

Table 4.8 presents the seasonal variations in the number of bird days spent by waders and terns during the non-breeding season at the MPRS. The non-breeding season was split according to the northern seasons and known trends of bird movements and activity at the site: (i) September to November (the northern autumn migration), (ii) December to February (the northern winter) and (iii) March to April (the northern spring migration period). Waders spend more days at the MPRS during northern winter and terns during northern autumn. Terns use the MPRS less than waders in all seasons but the period September-November 2009 (northern autumn).

Table 4.8: Seasonal Variations in the Number of Bird Days Spent by Waterbirds on MPRS

Survey periods	Total bird days spent by all waterbirds	Total bird days spent by waders	Total bird days spent by terns
September 09-November 2009	155900	71880	72610
December 09-February 2010	152206	107508	38594
March 10-April 2010	31664	26558	2720

4.3.4 Habitat selection by waterbirds at the Muni-Pomadze Ramsar Site

4.3.4.1 Long-term changes in the habitat available for waterbirds

There are five main habitat types: (i) forest habitat, (ii) sand dunes and beach, (iii) constructed salt ponds, (iv) riverine, and (v) shallow lagoon and mudflats, found at the Muni-Pomadze Ramsar Site. Of these, the shallow lagoon and its exposed mudflats, shrubs (mangroves), and salt-pans are the most important for waterbirds. They are used as feeding grounds, as well as roosting and nesting sites by waterbirds. Figure 4.21 shows the satellite imageries of the site at three points in time over the last 17 years, 1990, 2000 and 2007. The imageries show that drastic alterations have occurred in the MPRS since 1990. There was an increase in the extent of bare surfaces and build-up areas while the Muni lagoon, the forest vegetation and herbaceous shrubs show decreases in extent of coverage. The size of the Muni lagoon reduced by about 45% over a 10-year period (1990-2000) and more than 50% over the 17-year period. The extent of the built-up areas and bare surfaces quadrupled over the same period. Only 1% of the extent of herbaceous shrub coverage was left at the site as at 2007.

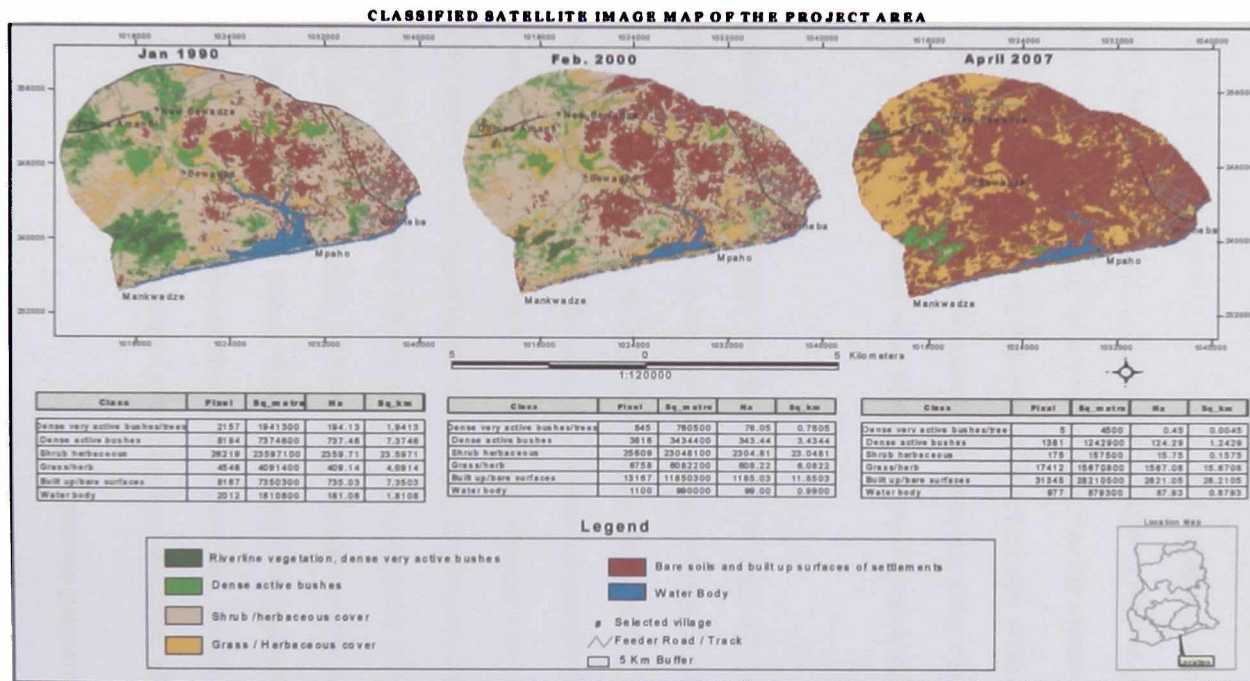


Figure 4.21: Long-term Changes in the Muni-Pomadze Ramsar Site

4.3.4.2 Spatio-temporal variations in the abundance and distribution of waterbirds at the Muni-Pomadze Ramsar Site

4.3.4.2.1 Waterbird diversity

Table 4.9 shows the importance of the different sections of the Muni Lagoon in terms of species diversity indices. The number of all waterbird species recorded during the low tide counts at the MPRS over the study period, ranged from 18 to 26 species in the different sites. The highest numbers of species were recorded in Site C, the mid-portion of the lagoon (26), A (mudflat I) (25) and SP, the salt ponds (23) and the lowest in E, the portion closest to the sea (18). The numbers of waterbird species varied significantly among some of the sites (Kruskal-Wallis test $H = 12.67$, $P < 0.05$). Nonetheless, the LSD Fisher test showed significant variations ($P < 0.05$) between Sites A and D (mudflat II), A and E, C and E, and E and SP. Site A represented the most important site in terms of abundance, as it accounted for 27% of waterbirds found at the MPRS at low tide. It was followed by Site C (23. 5%), and Site SP (18.7%), with Site B (5.8%) being the least important.

Shannon and Simpson species diversity indices were greater in Site B ($H' = 2.24$ and $1-D = 0.86$). It was also found that waterbird species were most evenly-distributed in Site B as compared to other sites (Table 4.18). The mudflat area, Site A ($H' = 2.24$, $1-D = 0.82$) had greater species diversity than the second mudflat, Site D ($H' = 2.03$ and $1-D = 0.79$). Site E had the least diversity of all sites ($H = 1.44$ and $1-D = 0.65$). Site SP was also an important site with Shannon diversity $H' = 2.21$ and Simpson $1-D = 0.85$.

Table 4.9: Waterbird Diversity Indices for Different Sites of the Muni Lagoon.

Diversity indices	Sampling Sites					
	A	B	C	D	E	SP
Number of species (S)	25	22	26	21	18	23
Abundance (Mean/m³)	2411	526	2131	1168	1129	1695
Magarlef richness (d)	3.08	3.35	3.26	2.83	2.42	2.96
Pielou's evenness (j')	0.7	0.75	0.65	0.67	0.50	0.70
Shannon- wiener (H')	2.24	2.30	2.10	2.03	1.44	2.21
Simpson (1-D)	0.82	0.86	0.78	0.79	0.65	0.85

The spatial agglomerative dendrogram based on waterbird abundance at MPRS revealed that waterbirds can be grouped into three significant station groups distinguished at a Bray-Curtis similarity level of 77%. The SIMPROF test based on the spatial variation in waterbird community composition showed no apparent reason for the split at 72.4 and 77.6% similarity ($\pi = 0.19$, $P < 99.1\%$ and $\pi = 0$ at $P < 100\%$). However, an evidence of structure was found at $P < 0.05$, at both 61.35 and 67.41% ($\pi = 2.6$ and 2.4 and $P < 0.6$ for both) (Figure 4.22). Thus, sites A, C and D can be combined as one group; sites SP and B as another group and site E as the third.

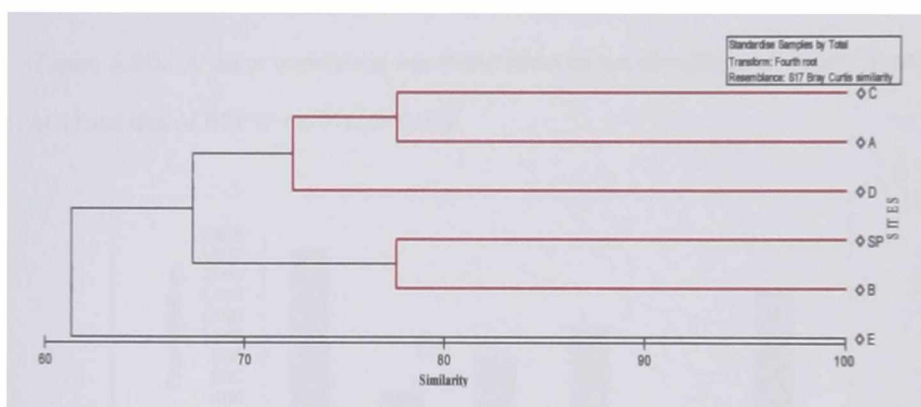


Figure 4.22: Group Average Agglomerative Dendrogram of Bray-Curtis Similarity of Waterbird Species Abundance Data from Six Different Sites in the Muni Lagoon over a Period of Eight Months.



4.3.4.2.2 Spatio-temporal variations in the abundance and distribution of the major waterbird groups at Muni Lagoon

- **Waders**

Waders were more abundant at Sites A (total sightings of 1,854), SP (1,346) and D (1,089) of the Muni Lagoon. The lowest count was recorded at site E (128) (Figure 4.23). Sites A and SP maintained relatively high numbers of waders throughout the study period. Sites C, D and B showed clear increases in numbers during the period December 2009 to January 2010, while Site E maintained very low numbers throughout (Figure 4.24). A linear correlation was found between the abundance of waders in sites ACD and that of BSP ($r = 0.976$; $P < 0.01$).

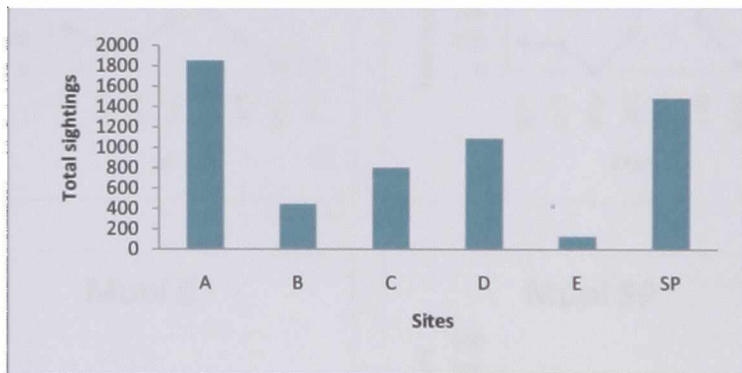


Figure 4.23: Distribution of waders in Muni lagoon over the study period (Sept 09/Apr10).

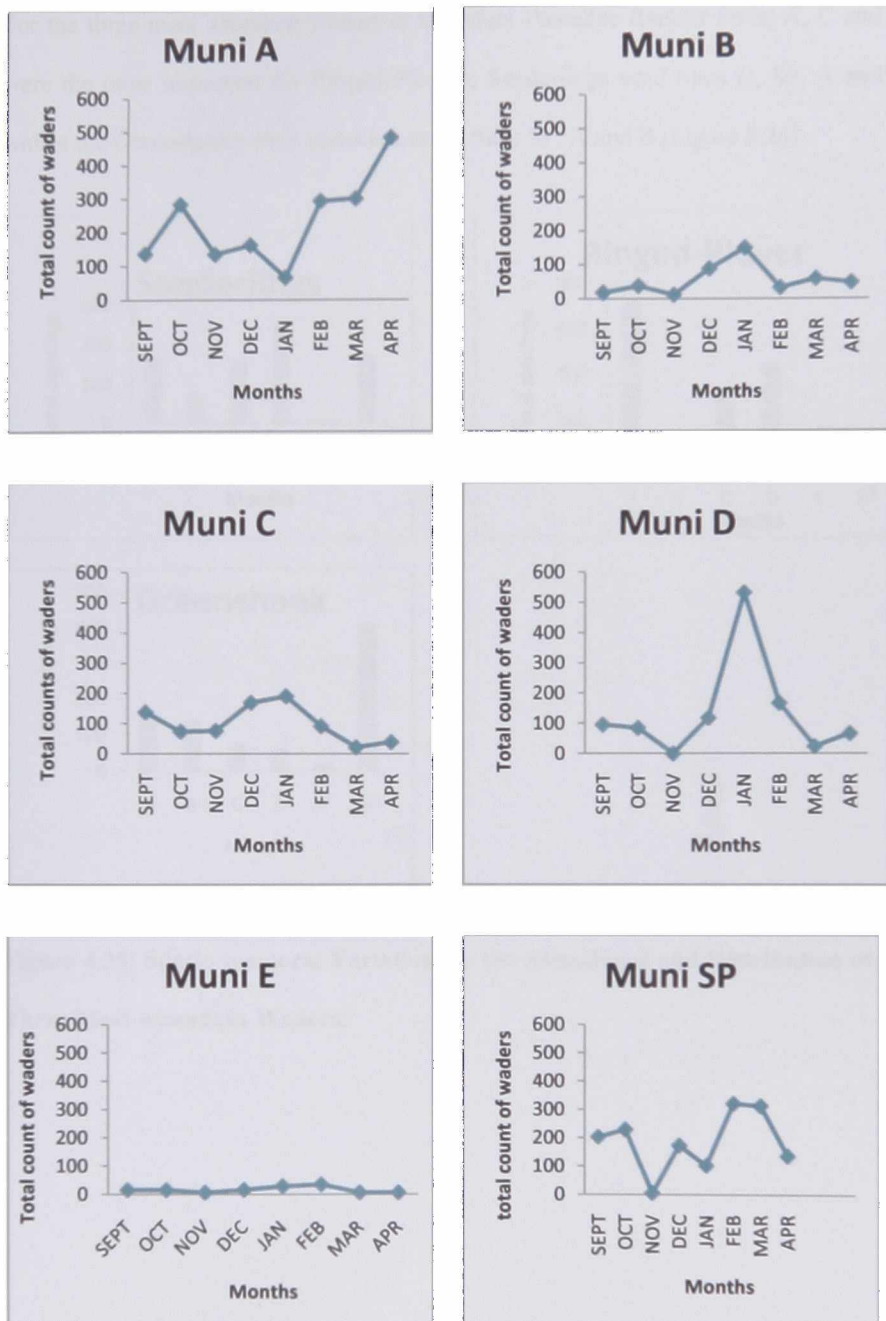


Figure 4.24: Spatio-temporal Variations in the Abundance and Distribution of Waders at Different Sites of the Muni Lagoon.

For the three most abundant waders at the Muni Pomadze Ramsar Sites, A, C and D, were the most important for Ringed Plovers, Sanderlings used Sites D, SP, A and C, while the Greenshanks were concentrated at Sites SP, A and B (Figure 4.25).

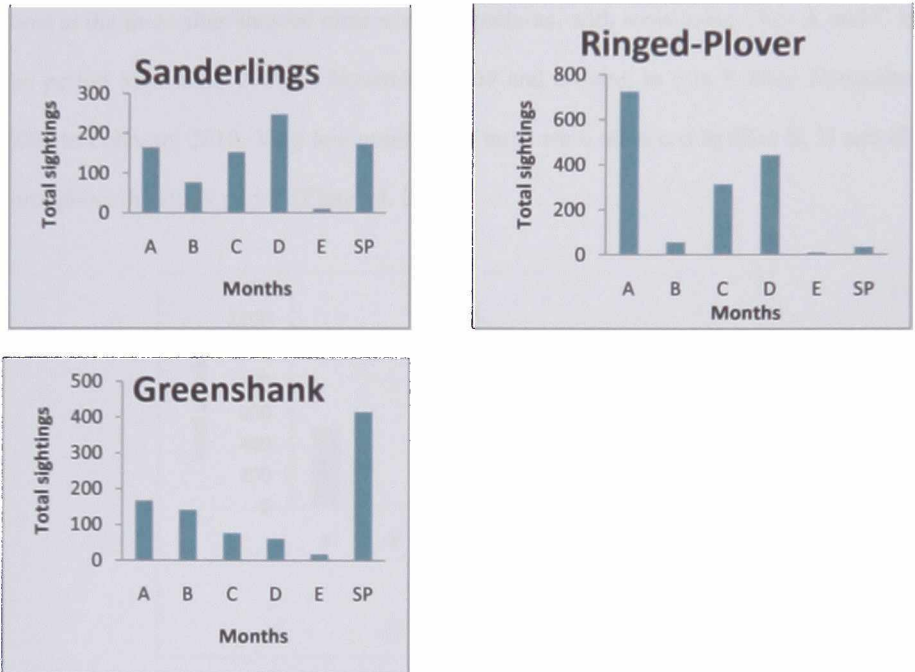


Figure 4.25: Spatio-temporal Variations in the Abundance and Distribution of the Three Most Abundant Waders.

- **Terns**

The spatial distribution of terns at MPRS as demonstrated in Figure 4.26, shows that Sites A, C and E were the most important roosting sites for terns. The distribution of terns at the three sites showed clear seasonal patterns, with terns using Sites A and C in the period September 2009 to November 2009 and moving in Site E from November 2009 to February 2010. Very low numbers of terns were observed in Sites B, D and SP throughout the study period (Figure 4. 27).

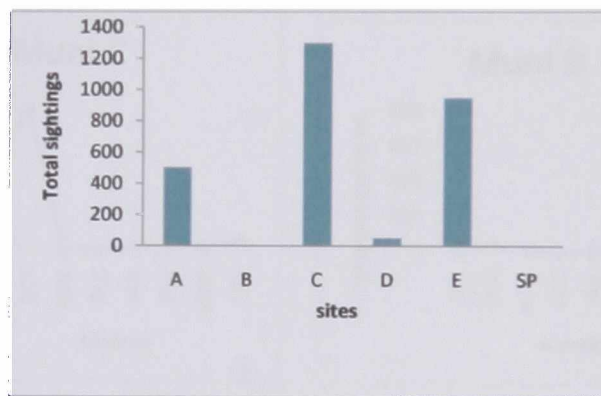


Figure 4.26: Distribution of Terns in the Muni Lagoon over the Study Period (Sept09/Apr10).

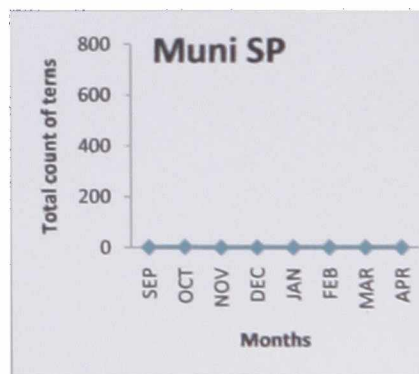
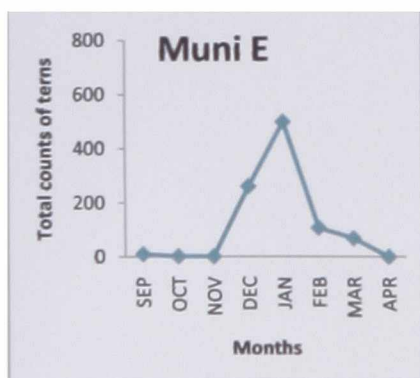
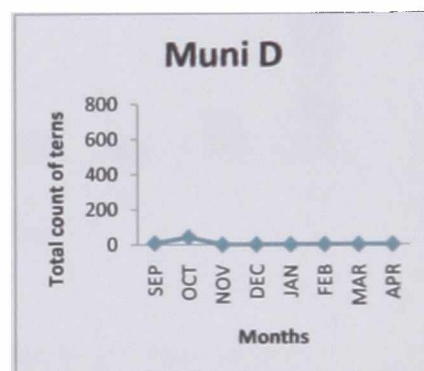
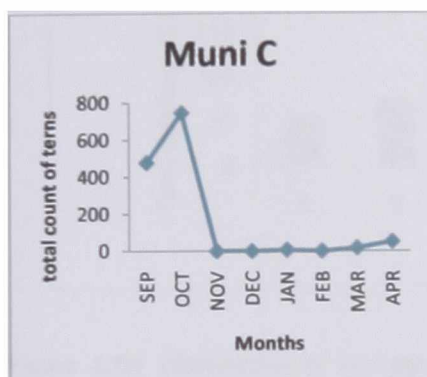
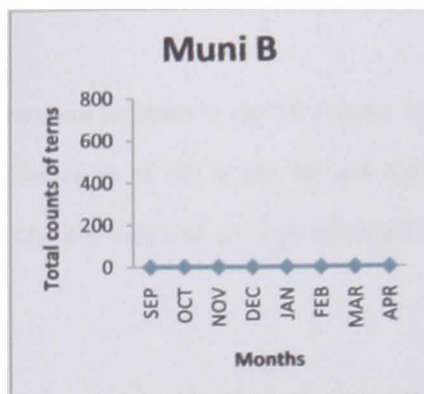
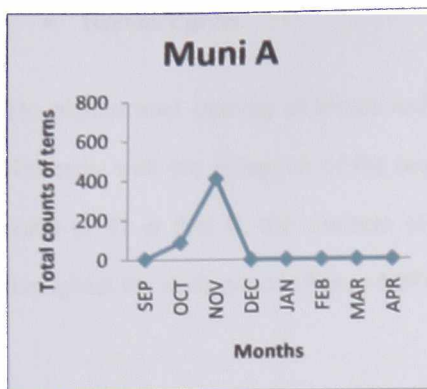


Figure 4.27: Spatio-temporal Variations in the Abundance and Distribution of Terns at the Muni Lagoon

- **Heron/Egrets**

The highest total sighting of herons and egrets was recorded at site SP (Figure 4.28). However, with the exception of the September count of 107 at Site SP and January count of 47 at Site B, the numbers of herons and egrets at all sites remained low throughout the study period (Figure 4.29).

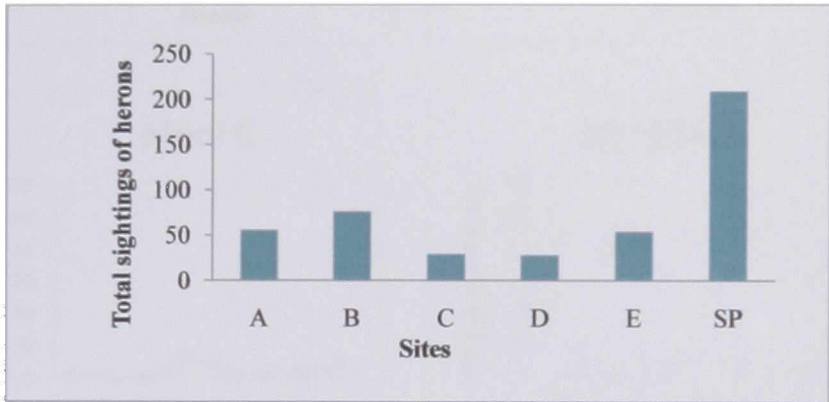


Figure 4.28: Distribution of Herons/Egrets at the Different Sites of the Muni Lagoon over the Study Period.

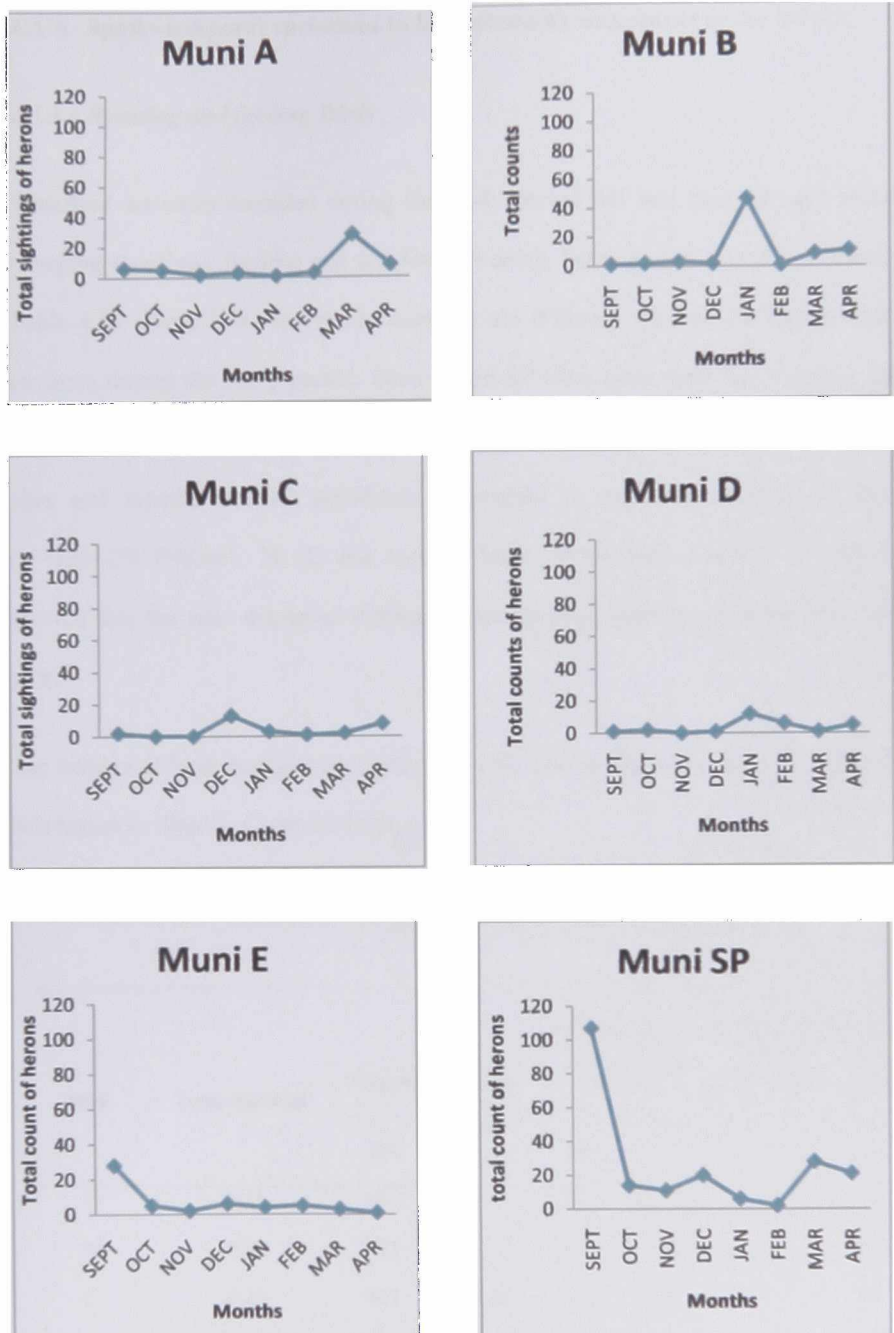


Figure 4.29: Spatio-temporal variations in the abundance and distribution of Herons and Egrets at the Muni lagoon

4.3.5 Spatio-temporal variations in habitat use by waterbirds at the MPRS

4.3.5.1 Roosting and feeding Birds

Waterbird activities recorded during the study period fell into three groups; roosting (sleeping standing), feeding and comfort (preening, bathing, and stretching) activities. Table 4.10 shows how waterbirds used the six different sites of the lagoon and its environs during the study period. Sites C and SP were used more for roosting, while Sites A and D were used most by feeding birds. A chi-square association test between sites and months showed significant interaction in terms of number of feeding ($\chi^2=1951.27$, $P<0.001$, 35 df) and roosting birds ($\chi^2=5624.66$, $P<0.001$, 35 df). This showed that the sites displayed different characteristics depending on the time of the year.

The number of birds engaged in comfort activities was relatively smaller, even though it was higher in Sites A, C and E (Table 4.10).

Table 4.10: Use of Different Sites by Waterbirds in the Muni Lagoon

Sites	Total sightings	Proportion feeding		Proportion roosting		Proportion engaged in comfort activities	
		No.	%	No.	%	No.	%
A	2411	1641	68.1	713	29.57	57	2.33
B	526	271	51.5	252	47.91	3	0.59
C	2127	601	28.26	1506	70.80	20	0.94
D	1168	915	78.34	250	21.40	3	0.26
E	1129	125	11.07	976	86.45	28	2.48
SP	1693	466	27.53	1227	72.47	0	0

4.3.5.2 Spatio-temporal variations in the feeding activity of waterbirds

Tables 4.11- 4.16 present the seasonal patterns in the use of the six sites of the MPRS by feeding and roosting waterbirds. The feeding activity of waterbirds found at MPRS varied among sites and months. The highest feeding activity at any one time was recorded at Sites D (78%) and A (68%) and remained high throughout the study period, except in November 2009 at A and November 2009 and March 2010 at D. The least used site in terms of feeding activity was Site E (11%), while the remaining two (B, SP) were equally used. While most sites had relatively high feeding activity in September and October, Sites C and SP did not. Nevertheless, Sites E and B had high feeding activity from September to November. The Kruskal-Wallis test showed that, feeding activity differed significantly among sites ($H= 17.889$, $X^2 (5, 0.05) = 11.070$, $P<0.05$) and the LSD test showed further that the waterbird feeding activity in the following sites were significantly different from each other: A and C, A and E, A and SP, B and E, B and SP, D and E, and D and SP. Feeding activity of waterbirds at Site A was found to be significantly correlated with biomass of polychaetes in that habitat ($r = 0.75$ and $P< 0.05$).

Table 4.11: Seasonal Changes in the Use of Site A by Waterbirds

Months	Total	%.		% engaged in
	sightings	% Feeding	Roosting	comfort activities
Sept-09	146	70.55	27.40	2.05
Oct-09	376	69.41	30.59	0
Nov-09	548	18.98	81.02	0
Dec-09	165	92.73	7.27	0
Jan-10	67	77.61	19.40	2.99
Feb- 10	295	86.78	13.22	0
Mars-10	327	90.52	9.48	0
Apr-10	487	85.42	3.90	10.68

Table 4.12: Seasonal Changes in the Use of Site B by Waterbirds

Months	Total	% engaged in		
	sightings	% Feeding	% roosting	comfort activities
Sept-09	20	70	25	5
Oct-09	42	73.81	26.19	0
Nov-09	15	100	0	0
Dec-09	91	45.05	52.75	2.2
Jan-10	197	23.86	76.14	0
Feb- 10	34	70.59	29.41	0
Mars-10	68	70.59	29.41	0
Apr-10	59	86.44	13.56	0

Table 4.13: Seasonal Changes in the Use of Site C by Waterbirds

Months				% engaged in
	Total sightings	% Feeding	% Roosting	comfort activities
Sept-09	618	21.2	75.57	3.23
Oct-09	821	6.58	93.42	0
Nov-09	77	80.52	19.48	0
Dec-09	179	69.27	30.73	0
Jan-10	200	66	34	0
Feb- 10	93	75.27	24.73	0
Mars-10	41	26.83	73.17	0
Apr-10	98	17.35	82.65	0

Table 4.14: Seasonal Changes in the Use of Site D by Waterbirds

Months				% engaged in
	Total sightings	% Feeding	% roosting	comfort activities
Sept-09	106	86.79	13.21	0
Oct-09	129	58.14	41.86	0
Nov-09	0	0	0	0
Dec-09	119	67.23	32.77	0
Jan-10	544	83.09	16.36	0.55
Feb- 10	173	84.97	15.03	0
Mars-10	25	40	60	0
Apr-10	72	81.94	18.06	0

Table 4.15: Seasonal Changes in the Use of Site E by Waterbirds

Months	Total		%.		%.
	sightings	%.	Feeding	roosting	engaged in
					comfort activities
Sept-09	53	69.81	30.19	0	
Oct-09	21	66.67	33.33	0	
Nov-09	8	62.5	37.5	0	
Dec-09	283	4.95	95.05	0	
Jan-10	533	3.94	93.05	3	
Feb- 10	147	18.37	81.63	0	
Mars-10	77	7.79	76.6	15.58	
Apr-10	7	14.29	85.71	0	

Table 4.16: Seasonal Changes in the Use of Site SP by Waterbirds

Months	Total		%.		%.
	sightings	%.	Feeding	Roosting	engaged in
					comfort activities
Sept-09	311	5.15	94.86	0	
Oct-09	243	17.28	82.72	0	
Nov-09	16	68.75	31.25	0	
Dec-09	193	23.32	76.68	0	
Jan-10	108	0.93	99.07	0	
Feb- 10	323	26.32	73.68	0	
Mars-10	341	61	39	0	
Apr-10	158	36.71	63.29	0	

The agglomerative dendrogram (Figure 4.30) based on temporal feeding activity of waterbirds at the MPRS over the study period showed that the feeding activities can be categorised into five, although they all showed more than 50% Bray-Curtis similarity. They also showed low inter-seasonal variability in waterbird activity at the MPRS. The SIMPROF test ($P < 0.05$) showed evidence of structure among the sampling periods.

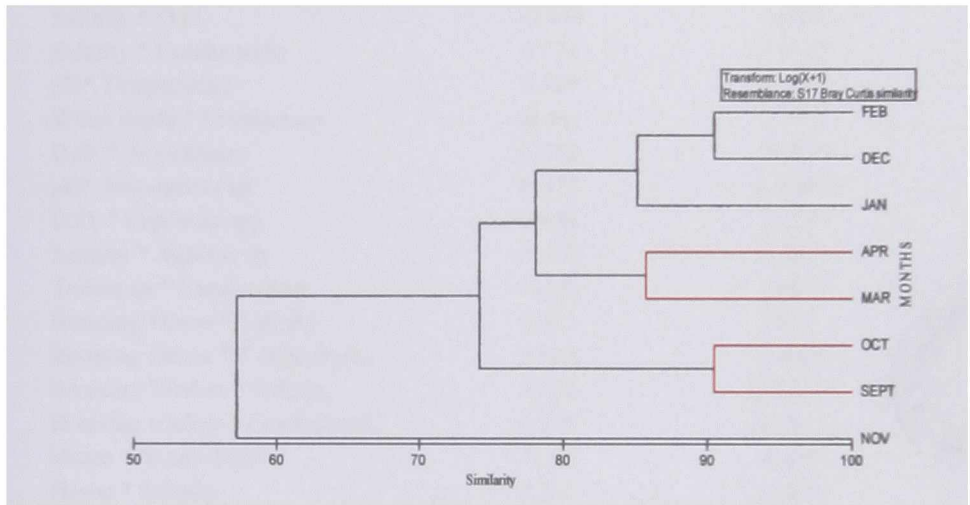


Figure 4.30: Group Average Agglomerative Dendrogram of Bray-Curtis Similarity of Waterbird Feeding Activity over a Period of Eight Months.

4.4 Biota-Environment Interaction at the MPRS

Changes in the environmental characteristics of the Muni lagoon were seen to some extent to have an impact on the macrobenthic assemblages, waterbird communities, distribution and activity. A combination of the environmental variables with biota data obtained from Sites B, C, E, SP sampled from December 2009 to April 2010 showed that there could be significant interactions among environmental variables, between

environment and benthic macro-invertebrate community, environment and waterbird community and between benthos and birds (Table 4.17).

Table 4.17: Biota-environment Interactions in Sites BCESP from December to April

INTERACTING VARIABLES	R	P
Salinity * D.O.	-0.448	0.048*
Salinity * Conductivity	0.524	0.018*
pH* Temperature	0.489	0.028*
Water depth * Crustaceans	-0.461	0.041*
D.O. * Polychaetes	0.512	0.021*
pH* <i>Notomastus</i> sp	0.575	0.008**
D.O. * <i>Capitella</i> spp	0.484	0.03*
Salinity * <i>Nephtys</i> sp	-0.459	0.042*
Tellina sp * Conductivity	-0.511	0.021*
Roosting Heron * Salinity	0.711	0**
Roosting Heron * Conductivity	0.465	0.039*
Roosting Waders * Salinity	0.648	0.002**
Roosting waders * Conductivity	0.598	0.005**
Heron * Water depth	-0.453	0.045*
Heron * Salinity	0.569	0.009**
Heron * Conductivity	0.504	0.023*
Roosting waders * Roosting terns	-0.501	0.024*
Roosting waders* Roosting Herons	0.517	0.02*
Roosting terns * waders	-0.536	0.007**
Feeding Herons* Crustaceans	0.536	0.015*
Tellina sp * Feeding Heron	-0.473	0.035*
Polychaetes* Bivalve	0.612	0.004**



Environmental variables were found to affect benthic community in the Muni lagoon. Dissolved Oxygen was found to positively affect polychaetes ($P < 0.05$), specifically the *Capitella* spp. ($P < 0.05$). Salinity, as one of the major environmental parameters that plays a significant role in the benthic community composition and structure in coastal lagoons, was found to be negatively correlated ($P < 0.042$) to the density/abundance of the *Nephtys* spp. in the Muni lagoon, whereas conductivity was negatively correlated to

the density of *Tellina* spp. ($P<0.021$). Another environmental variable, the pH of the water was found to be strongly and positively correlated ($P<0.01$) to the *Notomastus* spp. of Muni lagoon (Table 4.17).

Waterbird abundance and activity were found to depend to some extent on the environmental condition of the lagoon. Salinity and conductivity were seen to determine the roosting activity of both herons and waders (Table 4.17). The higher these environmental variables were, the higher the number of roosting waders and herons ($P<0.01$). Water depth was found to negatively affect the number of herons found at Muni. A positive correlation was displayed between feeding herons and crustaceans and between them and *Tellina* spp. a bivalve found at Muni lagoon, although herons have not been observed to feed on them. The abundance of herons at the Muni lagoon seemed to increase with increasing salinity ($P<0.01$) and therefore conductivity ($P<0.05$).

Nevertheless, interactions were found within the waterbird community and activities. The number of roosting terns on one hand, waders and roosting waders in the other, were negatively correlated as shown in Table 4.17. Furthermore, an opposite interaction was found between the number of roosting waders and that of roosting herons.

However, based on the data collected over the study period in all sections of the Muni lagoon (A,B, C, E, SP) except D, it was found that environmental variables and biota, could interact differently based on the area or the types of wetland in consideration as well as the dominant species that occur there (Table 4.18). The fact that Spearman's correlation failed to show any significance does not necessarily imply that there is no interaction between the tested variables.

Water depth was found to affect feeding waders differently depending on habitat. In Site B for instance, water depth and turbidity were negatively correlated to feeding waders, although the later was found to react positively to increasing wader depth in C (Table 4.26).

Roosting terns were affected by different environmental factors based on where they were located within the Muni lagoon; in that, when located in A they were negatively influenced by salinity while positively correlated to temperature when roosting in E. Roosting terns were recorded at Site A only once at low tide. The Site E represented a high tide roost for the terns.

Like, the roosting terns, roosting herons were found to be correlated with different environmental factors as the habitat in which they were found also differed. In Site E, they were negatively correlated with pH, but negatively correlated with water depth and turbidity in Site E. As temperature increased, the number of roosting waders was seen to increase (Table 4.18).

Feeding herons were negatively correlated with benthic organism (polychaetes ($P < 0.01$) and bivalves ($P < 0.05$). Whiles the density of polychaetes was negatively influenced by salinity, that of crustaceans was influenced by water depth and turbidity negatively and that of bivalves by pH (C) and temperature (E) (Table 4.18).

Table 4.18: Influence of Habitat Types on the Interaction between Environment and Fauna Abundance and Activity.

A		
Interacting variables	R	P
Salinity * Tern	-1	0**
B		
Interacting variables	R	P
Temperature* Roosting waders	0.81	0.015*
Water depth* pH	0.881	0.004**
Roosting Heron* Water depth	-0.724	0.042*
Water depth * Feeding Waders	-0.762	0.028*
Water depth * Crustaceans	-0.733	0.039*
Turbidity *Water depth	0.833	0.01**
Turbidity* Waders	-0.786	0.021*
Turbidity * Feeding waders	-0.929	0.001**
Turbidity * roosting Heron	-0.0896	0.003**
Salinity * Polychaetes	-0.743	0.035*
Turbidity *Crustaceans	-0.733	0.039*
Feeding Herons * Polychaetes	-0.845	0.008**
C		
Interacting variables	R	P
Water depth* Turbidity	0.708	0.05*
Water depth* Feeding Waders	0.878	0.004**
Water depth * Waders	0.805	0.016*
pH * Bivalves	-0.826	0.011*
E		
Interacting variables	R	P
Temperature* Roosting terns	0.778	0.023*
Temperature* Bivalves	0.726	0.041*
Roosting heron *pH	-0.847	0.008**
Feeding Herons * Bivalves	-0.752	0.031*
SP		
Interacting variables	R	P
Temperature * Water depth	0.9	0.037*
Temperature* Turbidity	-0.9	0.037*

The Canonical Correspondence Analysis (CCA) model also indicated that there could be correlation between environmental and biological variables. The horizontal and vertical axes represent the most important environmental gradient along which waterbirds and benthic organisms were distributed. The direction of each environmental vector represents the maximum rate of change for that particular environmental variable and its length indicates the relative importance to the ordination (Lamprey and Armah, 2008). The figure 4.31 below showed that there were correlation patterns between total abundance of birds and their activity with hydrological variables although the CCA failed to show any significant correlation between them as shown by table 4.17 and 4.18, when using Spearman's correlation coefficient.

Water depth was found to be the only environmental variable to have had a significant correlation with the abundance of macrobenthic invertebrates ($P < 0.05$) (Appendix). The CCA indicated that water depth was the variable that best differentiated habitat of the studied macrobenthic fauna (Figure 4.32).



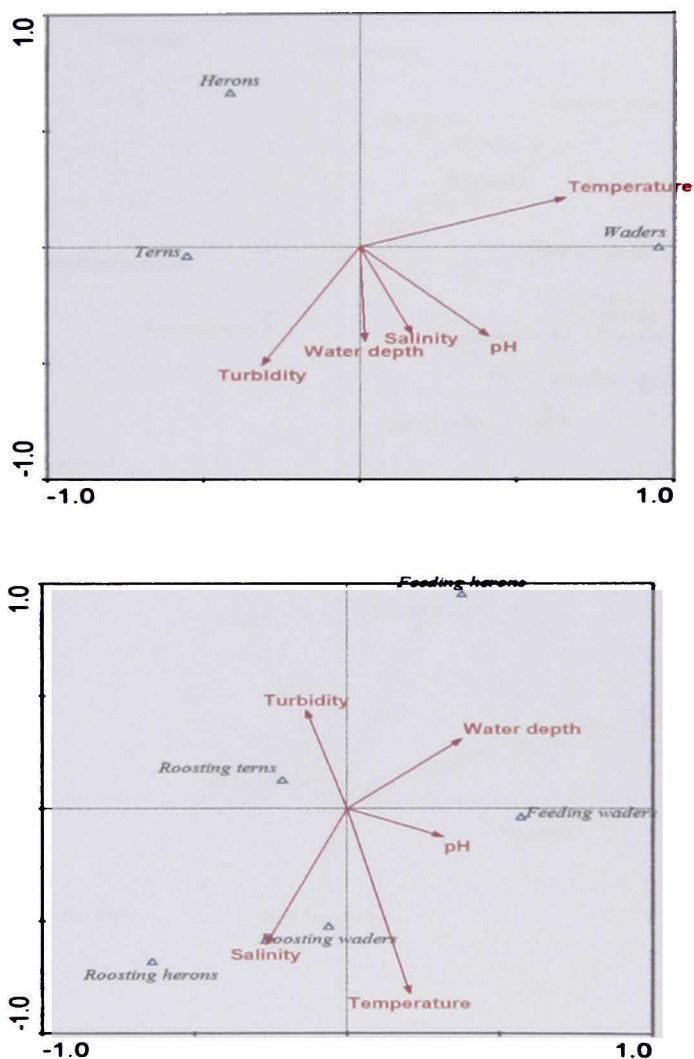
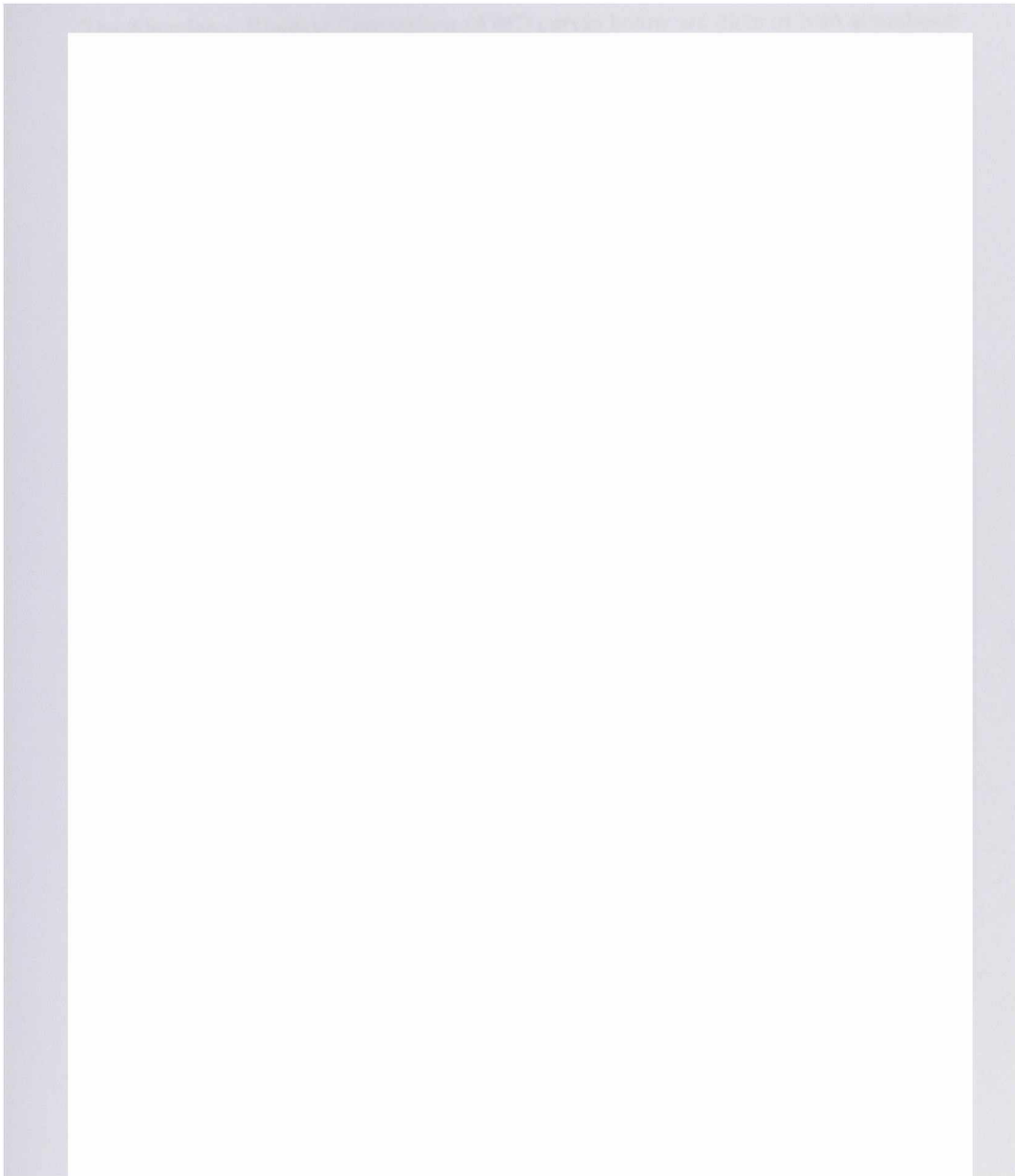


Figure 4.31: CCA Ordination Diagrams for the Abundance (left) and Activity (right) of Waterbird in Relation to Environmental Variables



4.5 Muni-Pomadze Habitat Condition

The Abundance Biomass Comparison (ABC) curves below are plots of both abundance and biomass K-dominance lines based on the density and biomass of polychaetes sampled at the Muni lagoon. The extent to which the biomass curve lies above the dominance curve was measured as W (Warwick and Clarke, 1991). A positive or negative W indicated the extent of disturbance as the undisturbed condition or disturbed ones respectively. The ABC curves (Figure 4.33) below showed that Sites E and B of the Muni lagoon were undisturbed ($w = 0.262$ and $w = 0.133$) whiles Site A was moderately disturbed ($w = -0.079$) and site C highly disturbed ($w = -0.211$).

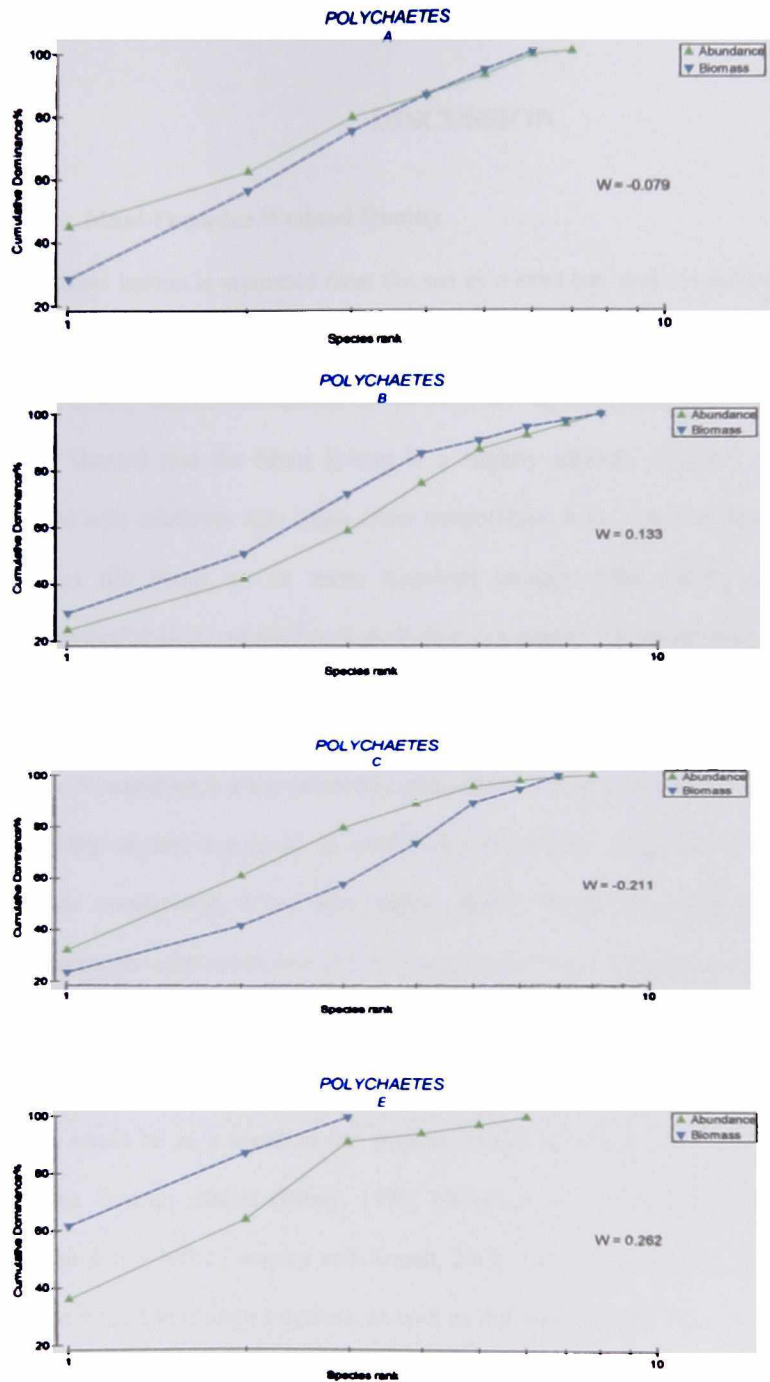


Figure 4.33: ABC Plots over the Eight Months Sampling Period (Sept 09-Apr 10) in the Different Sections of the Muni Lagoon.

CHAPTER FIVE

DISCUSSION

5.1 Muni-Pomadze Wetland Quality

The Muni lagoon is separated from the sea by a sand bar, and consequently in addition to its physiochemical characteristics, it qualifies as a coastal lagoon (Bird, 1994). The water quality assessment carried out on the Muni lagoon from September 2009 to April 2010 showed that the Muni system is a slightly alkaline, shallow, and hypersaline lagoon with relatively high mean water temperature. It also showed that the lagoon can support life based on its mean dissolved oxygen value, which falls within the recommended D.O. values, 5 to 8 mg/L that can support biological activities (Finlayson *et al.*, 2000).

Water temperature is a key environmental parameter that plays a fundamental role in the solubility of salts and gases, as well as the dissociation of dissolved salts, and hence electric conductivity (Carr and Neary, 2006). Water temperature also positively influences the respiration rate of living organisms in water and enhances decomposition rate, thereby decreasing dissolved oxygen (Chapman, 1992; Anthony *et al.*, 2009). The relatively high mean temperature of the waters of Muni Lagoon recorded during this study could be as a result of the shallow nature of the lagoon as compared to other studies. Several studies (Biney, 1990; Finlayson *et al.*, 2000; Gordon *et al.*, 2000; Armah *et al.*, 2005; Lamptey and Armah, 2008; Tay *et al.*, 2009) carried out in Ghana on the Keta and Songor Lagoons, as well as the Muni Lagoon have attributed the high temperature of Ghanaian coastal lagoons to their shallow nature. Furthermore, this study showed that in Sites B (freshwater inflow) and C (mid lagoon), temperature was highest

when water depth was lowest in March 2010, and vice-versa in November 2009, although no significant correlation was found between the two variables.

In November, low temperatures were recorded for all sites sampled, and this could be attributed to the time (8:15 am) of the measurements, as ambient temperature (which varies according to the time of the day) affects the temperature of the water. According to Turner (2003), changes in air temperature significantly influence that of the waters of coastal lagoons.

Results from this study also showed that water depths of the Muni lagoon were regulated by several factors. These include the amount and regularity of rainfall on one hand and the rate of evaporation on the other. Moreover, because Muni Lagoon remained open to the sea during the sampling period, sea tides and wave direction might have also had a role to play. Allen *et al.* (1981) indicated that water quantity and quality in a lagoon are affected by its water gain (rainfall, tide, runoff, etc.) and loss (evaporation). Tumbulto and Bannerman (1995) reported that seepage from the sea and discharges of water from the rivers represent minor water input to the Muni Lagoon. These must have also contributed to the variations in water depth.

The salinity and conductivity values of the Muni lagoon were characteristic of hypersaline coastal lagoons found in Ghana (Finlayson *et al.*, 2000; Gordon *et al.*, 2000; Lamptey and Armah, 2008; Tay *et al.*, 2009). Limnological surveys carried out in 1994 and 1999 (Gordon, 1994; Gordon and Ankrah, 1999; Gordon *et al.*, 2000) compared with the present values, showed that the mean salinity of the Muni lagoon had reduced by about 30%. The mean values varied from 56.88‰ in 1994 (Gordon *et al.*, 2000) to 40.28‰ in 2009/2010. This comparison also showed that the highest values recorded at the Muni lagoon varied from 165‰ in 1994 to 106‰ as shown in this study. The low

salinity values recorded during this study could be due to the fact that sea water must have diluted the waters of the lagoon, since the lagoon remained opened to the sea throughout the sampling period. Furthermore, the difference in the highest salinity values could be greater since the highest salinity recorded during this study was from the salt ponds while that from the previous studies was from the lagoon itself. Thus the lowering of salinity in the Muni lagoon is not without any consequences, as it favours macrobenthic organisms which can hardly survive in hypersaline conditions (Ntiama-Baidu *et al.*, 2000a).

The values of salinity ranged from 20.2‰ in A (mudflat I) to 106 ‰ in SP (Salt Pan) and that of conductivity from 38.6 in E to 74.8 ‰ in SP. Even though salinity of the coastal lagoon was expected to be higher in the dry season, that of the site A (Mudflat I) was the lowest compared to the other sites. This is probably due to its proximity to the sea and its connection to a nearby stream. The inflow of the sea water might have contributed in lowering the salinity of the Site A through dilution. Furthermore during the short rainy season, the low salinity values of Site A might have been as a result of freshwater inflow from a nearby stream in Site A which becomes prominent during this period. Though Site E (sea water inflow) was the closest to the sea, it had higher mean salinity, due to the presence of a freshwater stream in Site A.

It has been well established that conductivity increases with increasing salinity (Sutherland, 1996). As expected, salinity of the Muni Lagoon was also positively correlated with conductivity. The reduction in salinity in November in some of the sites could be as a result of rainfall. Site SP had both the highest salinity and conductivity values, due to commercial salt winning. Sea water was pumped into the salt pans and allowed to evaporate, leaving behind salt that will be harvested.

The values of the dissolved oxygen and pH recorded during this study showed little variability. This is common in most of the Ghanaian coastal lagoons, such as Songor and Keta (Piersma and Ntiamoah- Baidu, 1995; Finlayson *et al.*, 2000; and Lamptey and Armah, 2008), Muni lagoon (Biney, 1995; Gordon *et al.*, 2000), and other coastal lagoons located along the Gulf of Guinea (Kouame *et al.*, 2009) in lagoons of Cote d'Ivoire.

Dissolved oxygen (D.O.) is one of the most important parameters in aquatic ecosystems that are used by aerobic organisms for their metabolism. The overall mean value of the dissolved oxygen at Muni lagoon was 7.06 (S.D. 1.4mg/L), within the recommended D.O. values of water that can support biological processes (World Resource Institute, 2003). These values could be attributed to the shallowness of the lagoon that, under the effect of the wind, favours oxygen penetration through wind action. During this study, algal mats were often observed on the sediments, and through photosynthesis they contribute to the increase in D.O. The level of dissolved oxygen in water is known to depend not only on the physical characteristics of the water such as temperature, the rate of aeration from atmosphere, temperature, salinity and air pressure but also the combination of these factors with the fauna present in the water.

Even though D.O. is expected to increase with reducing temperature (Joos *et al.*, 2003; Anthony *et al.*, 2009), in Site C of the Muni lagoon the highest D. O mean values were recorded against the highest spatial mean temperature values. This is reasonable since large algal mats were often observed on the sediment at Site C. With increasing temperatures, these algae through photosynthesis, increase the level of oxygen in the surface water during the day. The Site D (mudflat II) had the lowest mean values, probably due to the fact that the site was surrounded by the remnant mangroves that

provide shade, thus lowering temperature which is known to increase the rate of oxygen absorption by water. Decomposition of these plants absorbs oxygen, thereby reducing its concentration in the water. Nevertheless, little temporal mean variation in DO was observed at Muni, probably due to the presence of significant algal mats on the sediments.

The mean pH values of the Muni waters were fairly alkaline with low variability, typical of coastal lagoon waters (Finlayson *et al.*, 2000; Lamptey and Armah, 2008). It is partly due to the influence of sea water which has a pH range of 7.5- 8.4 (Riley and Chester, 1971), as the lagoon was opened. The hydrogen ion concentration (pH) was low, probably because wind induced mixing would lead to very homogeneous water mass and the biological processes could have resulted in buffering the change in pH (Finlayson *et al.*, 2000).

The mean turbidity values (27.49 NTU S.D. 20.06) of the Muni lagoon showed that it is a turbid system that has a great variability. The minimum turbidity values recorded at Site C were probably due to the low disturbance as compared to the site with the highest turbidity values (Site A, mudflat I). The consolidated nature of the soils of Site C probably has also played a role; the ground was mostly covered with dead shells. Hence even when people cross the site or carry out activities, there was no much of sediment re-suspension.

Site A had the highest clay percentage and the lowest mean water depth; hence through wind mixing, the water could easily become turbid. Besides, trampling by people crossing the lagoon occurred a lot especially at site A, which is the shallowest thereby leaving behind a “soup like” solution of clay.

There was an increase in turbidity in November 2009 in all the sampled sites and a decrease in March 2010. In November, turbidity was very high even though water levels were low, it could be as a result of wind mixing of the water. Rainfall occurred in March, thereby contributing to the decrease in turbidity recorded during that period.

These physiochemical parameters of the waters of the Muni Lagoon contribute enormously to the abundance and quality of biological resources found in it. Nevertheless, the high values of temperature, salinity and turbidity recorded during this study are likely to affect waterbird communities both directly and indirectly through their effect on the macrobenthic organisms and fish, that are the primary food of the waterbirds. These macrobenthic and waterbird species should have either a limited tolerance range or level, above or below which they may be affected, or eliminated from the Muni system.

Temperature, as the governing factor in aquatic life processes is very important to macrobenthic and waterbird communities. Against the background of the narrow temperature tolerances of aquatic organisms, the high temperatures of Muni Lagoon could jeopardise their lives. The optimal temperature for aquatic life range from 5-25°C, and an increase in water temperature up to 34-37°C could decrease the density of benthic fauna. According to Anbuezhian *et al.* (2009), a study conducted in the Coastal Belt of Thondi, Southeast Coast of India, high temperature of about 29.5°C recorded in the pre-monsoon season had an impact on the distribution of macrobenthic community and low temperature recorded in December 2009 and January 2010 months influence higher faunal density.

Similarly, salinity is a limiting factor in the distribution of benthic organisms. Like temperature, the level of salinity in aquatic systems is important to aquatic plants and animals as species can survive only within certain salinity ranges (Friedl *et al.*, 2004).

A key element contributing to stress is the frequent fluctuations in environmental parameters that may hinder the growth and reproductive processes of organisms in question, even though some species may be well-adapted to surviving in saline environments (Koranteng *et al.*, 2000; Carr and Neary, 2006). This can cause changes in the structure and distribution pattern of organisms. According to Barnes (1980), coastal lagoons are naturally stressed environments due to the frequent variability in their waters. Muni Lagoon is therefore not an exception of a stressed environment based on the wide fluctuations in its physiochemical (mainly salinity and temperature) environment, the dominance of its benthic fauna by a single opportunistic species sampled during this survey. Even though salinity of the Muni lagoon has reduced (from 2000 to 2009/2010), the environment has not improved. Its macrobenthic diversity is still being mainly contributed by just one taxon.

Based on the Abundance/Biomass Comparisons, Site A (Mudflat I) was found to be moderately disturbed while Site C (mid Lagoon) is highly disturbed. Site A receives household solid wastes and sewage which are washed into the lagoon during high tide and rains. This act by local people confirms the statement by Ryan and Ntiamoa-Baidu (2000) that wetlands are considered “wastelands” in Ghana and other countries of the world. The high turbidity values could also account for the disturbance in the area, since it inhibits light penetration and therefore have an effect on the chemistry of the water and photosynthesis. According to Addo Oduro (Pers. Comm.), the Biological Oxygen Demand (BOD) values obtained during his research on the impacts of human activities

on the Muni Lagoon during the same period, were above the permissible levels. Furthermore, Gordon *et al.* (2000) recorded the highest biological oxygen demand (BOD) values in the southern sector of the lagoon, which correspond to Sites A and C in this study. The use of fertilizers by farmers in areas around the lagoon could have also played a role in the high BOD values recorded at Site C.

The ability of aquatic organisms to cope or survive in such stressed environment further determines their survival, community structure and composition. The macrobenthic community in Muni lagoon can be characterised as typical of a stressed hypersaline ecosystem. Of the macroinvertebrates sampled during this study, the Capitellid worms were the most abundant. These species are opportunistic species that are capable of tolerating very low oxygen and high salinity concentration (Day, 1967a; Ahulu *et al.*, 2006).

Based on the macrobenthic community pattern obtained by multivariate analysis, two main zones can be distinguished. A group of Sites, A, B, C, D, and E located within the lagoon and the Salt Ponds (SP). The lagoon is influenced by the sea water and freshwater inflow from the three seasonal rivers that drain into the lagoon. Macrobenthic organisms are therefore recruited into the lagoon from these two sources, thus explaining the great species diversity at Sites E and B and abundance of the polychaetes and bivalves in the lagoon.

The high diversity of families recorded in the month of February 2010, could be attributed to the high tide level and rainfall that ensured recruitment from the sea and water inflow from the rivers (through immigration and an increase in reproduction) of certain faunistic species.

Sites E (seaward section) and B (river inlet) were the most diverse and had the highest species richness. This is probably due to the inflow of river waters from Site B, and that of sea waters from Site E, into the lagoon, through which new species are recruited into the lagoon. This study is in accordance with da Silva *et al.* (2005), a study carried out in a hypersaline coastal lagoon, Lagona de Araruama, in Brazil, the site closest to the sea scored the highest species richness, which also gradually reduced to the inner lagoon.

The Salt Ponds constructed for commercial salt wining are enclosed and only receive sea water pumped into them. The water is then allowed to evaporate in order to harvest salt, thus making the habitat very hypersaline (106 ppt). The hypersaline character of the Site SP is detrimental to the macrobenthic communities. It is therefore not surprising to record the lowest abundance and richness of macrobenthic species when salinity is so high (Liang *et al.*, 2002) in created wetlands such as Salt Ponds. Nevertheless, Rossiter and Crawford (1981), and Nelson *et al.* (2000) also found higher species richness, abundance/density and diversity in natural wetlands.

At the Muni Pomadze Ramsar Site, the macrobenthic community is dominated by the opportunistic *Capitella* spp. which is known to proliferate in such stressful environment. The macrobenthic fauna of the Muni lagoon experienced variations as a response to the fluctuating hydrology and water quality during the study period. The two peaks in macroinvertebrates at Muni lagoon recorded in January and March 2010 showed that the temporal variation in the density of benthic organisms could be linked to increasing rainfall in the area. Rain water through dilution of the concentrated salty water of the lagoon favours re-colonisation by benthic organisms. The Muni site recorded its highest amount of rain in January 2010. This is in agreement with several studies (Lamprey and Armah, 2008; Gordon, 2000) that recorded the highest abundance and diversity of

macrobenthic species during the rainy season in Keta and Muni Lagoon in Ghana, respectively.

Furthermore, polychaetes of Muni lagoon were positively correlated with the dissolved oxygen and negatively correlated with salinity; therefore the reduction in salinity and the relatively high mean dissolved oxygen recorded in the site in March 2010 have contributed to the high abundance of macroinvertebrates recorded during the same period.

Site C, located in the middle of the Muni lagoon had the highest density of polychaetes and bivalves. This agrees with findings by Lamptey and Armah (2008) in another hypersaline lagoon, the Keta Lagoon. Site SP, which is the most hypersaline of all, had the least density and diversity. It is therefore obvious that high salinity affects the diversity and abundance of living aquatic organisms, as recorded by García de Lomas *et al.* (2005) and Gordon (2000) in hypersaline ecosystems. Another stressor endured by benthic fauna in hypersaline environments is the wide variability in salinity that may inhibit the physiological responses of the organism.

From this study, the macrobenthic fauna displayed wide spatio-temporal patterns and are characterised by high inconsistency and dominance patterns within the sampling period as shown by Mistri *et al.* (2001).

Polychaetes were more abundant than bivalves at Muni Lagoon, probably due to the fact that bivalves cannot tolerate high salinity, as well as polychaetes (Delaue, 1994). Polychaetes of Muni Lagoon were dominated by the capitellid worms. Benthic species belonging to the family Capitellidea are known to survive in hypersaline ecosystem, more especially the opportunistic *Capitella capitata*. Bivalves were dominated by

Anadara senilis, which were distributed along the banks, especially at low water levels. (Gordon, 2000; Ahulu *et al.*, 2006)

The highest macrobenthic density was recorded in the middle portion of the Muni Lagoon. The macrobenthic community of the Site C increased under unfavourable condition, because it is dominated by *Capitella* spp., which could take advantage of such conditions.

5.2 Waterbirds at the Muni-Pomadze Ramsar Site

5.2.1 Changes in waterbird populations

The Muni Pomadze Ramsar Site is the fifth most important wetland in Ghana (Wuver and Attuquayefio, 2006; Ministry of Lands, Forestry and Mines, 2007). It was designated as a Ramsar Site, in 1992, on the basis of its internationally important Tern (Royal, Sandwich, Black and Common) populations and its total waterbird population, estimated at 23,000 (Ntiamoa-Baidu *et al.*, 2000a). The MPRS is also known to support 48 out of the 88 species of waterbird known to occur on the coast of Ghana (Ntiamoa-Baidu *et al.*, 2000a).

During this study, the maximum numbers of the four tern species (Royal, Sandwich, Black and Common) recorded at the MPRS were below the maximum counts recorded in the 1986-1998 survey (Table 5.1), and also below the 1% of the total East Atlantic Flyway population, thus failing to qualify as internationally important. Possible explanations for this would be that there has been a decrease in the size of the

populations of the three tern species using the Muni Pomadze Ramsar Site or an increase in the global population estimates of the species.

The fact that most terns usually feed at sea during the day (Grimes, 1979; Ntiamoa-Baidu *et al.*, 2000a), instead of on the coastal lagoons, could be the reason why the numbers were low since counts were carried out only during the day. Terns are most abundant at the lagoon roosting sites either at dawn before they go feeding or after dusk when they come back to the night roosting grounds (Ntiamoa-Baidu *et al.*, 1992; 2000a).

This study shows however that the MPRS is still important in terms of the number of waterbird species it supports, with 33 species being recorded during the eight-month study period compared to the 48 species recorded over a period of 12 years (Ntiamoa-Baidu *et al.*, 2000a). The maximum counts of 1,070 waders, 145 herons and 881 terns recorded during this study were far below the 3,530 waders, 345 herons and 16,925 terns recorded during the 1986-1998 survey (Ntiamoa-Baidu *et al.*, 2000a). Natural and anthropogenic causes like habitat degradation, which lead to the loss in feeding and roosting habitats, reduction in the quality of wetlands could account for this reduction.

Five of the waterbird species recorded at Muni, the Spur-winged Plover, Whimbrel, Wood Sandpiper, Common Sandpiper, and the Sanderling occurred commonly on the coast of Ghana, in nationally important numbers. Compared with the 1986-1998 counts, the maximum count of five species recorded in this study was higher, 26 species were recorded in lower numbers and 16 species recorded in 1986-1998 were not seen at all in this study. One species, the Squacco heron (*Ardeola ralloides*) recorded once in this study, was not reported in the 1986-1998 survey. Of the six least common species recorded at the MPRS, the Senegal Thick-Knee (a local resident) and Dunlin (a

Palearctic migrant) are uncommon on the coastal wetlands of Ghana, thus their few numbers. The Collared Pratincole was recorded a few times, since as a local breeder its numbers only increased as its breeding period was approaching. Some species have restricted habitat or display habitat preferences (Oystercatcher, Sanderling, Jacana (*Actophilornis Africana*) and may therefore be more abundant at some sites than others (Ntiama-Baidu *et al.*, 2000a).

There have also been some changes in the relative abundance of the different wader species. The Common Ringed Plover has now become the most abundant species, replacing the Curlew Sandpiper which was the most abundant at MPRS and at all other wetland sites on the coast of Ghana (Ntiama-Baidu, 1991). Although the population of Ringed Plovers at Muni was higher than that of the Curlew Sandpiper, it is clear that this change was not caused by an increase in the population of Ringed Plovers at MPRS. The populations of both species have decreased, and the maximum counts of the Ringed Plover and the Curlew Sandpiper were 35.5% and 87.5% respectively lower than the numbers recorded in the 1986-1998 study.

Though few species (e.g. Wood sandpiper, Spur-Winged Plover, Common Sandpiper) have increased in numbers, majority of waterbird species have decreased in abundance. The Common Sandpiper is one of the species that are known to use mangroves frequently. Even though mangroves are being destroyed at MPRS (Entsua-Mensah, 1997), the number of Common Sandpipers recorded was higher than in the earlier study. The abundance of small fiddler crabs (*Uca* spp.), which the Common Sandpiper could easily prey upon (van de Kam *et al.*, 2004), made the MPRS their suitable alternative habitat.

The maximum number of Whimbrels recorded at the site has not changed, as the population of the species is known to be stable in West Africa (Rose and Scott, 1997). Whimbrels use tidal areas in Africa and are also specialised feeders on the most numerous species of fiddler crab (van de Kam *et al.*, 2004), which they were observed feeding on at the MPRS.

Table 5.1: Waterbird Population Status at the MPRS

Common name	Scientific name	Maximum recorded at any onetime at Muni		Status
		1986- 1998	2009- 2010	
Waders				
Senegal thick-Knee	<i>Burhinus senegalensis</i>	1	2	Increase
Senegal Wattled Plover	<i>Vanellus senegalensis</i>	5*	-	-
Collared Pratincole	<i>Glareola pratincola</i>	65	52	Decrease
Spur-Winged Plover	<i>Vanellus spinosus</i>	2	36*	Increase
Ruddy Turnstone	<i>Arenaria interpres</i>	30*	14	Decrease
Kittlitz's Plover	<i>Charadrius pecuarius</i>	60	3	Decrease
Common Ringed Plover	<i>Charadrius hiaticula</i>	640	413	Decrease
White-Fronted Plover	<i>Charadrius marginatus</i>	35*	13	Decrease
Grey Plover	<i>Pluvialis squatarola</i>	210*	39	Decrease
Whimbrel	<i>Numenius phaeopus</i>	35*	34*	Stable
Wood Sandpiper	<i>Tringa glareola</i>	15	56*	Increase
Common Sandpiper	<i>Actitis hypoleucos</i>	25	57*	Increase
Common Redshank	<i>Tringa tetanus</i>	15	5	Decrease
Dunlin	<i>Calidris alpina</i>	2	1	Decrease
Bar- Tailed Godwit	<i>Limosa lapponica</i>	30*	7	Decrease
Little Stint	<i>Calidris minuta</i>	425	113	Decrease
Sanderling	<i>Calidris alba</i>	485*	308*	Decrease
Curlew Sandpiper	<i>Calidris ferruginea</i>	1700	212	Decrease
	<i>Himantopus</i>	475	103	
Black-Winged Stilt	<i>himantopus</i>			Decrease
Spotted Red Shank	<i>Tringa erythropus</i>	180	5	Decrease
Common Green Shank	<i>Tringa nebularia</i>	870*	187	Decrease
Marsh Sandpiper	<i>Tringa stagnatilis</i>	150	10	Decrease
Eurasian Oystercatcher	<i>Haematopus ostralegus</i>	2	-	-
Avocet	<i>Recurvirostra avosetta</i>	50	-	-
Knot	<i>Calidris canutus</i>	35	-	-
Black-tailed Godwit	<i>Limosa limosa</i>	50	-	-
Curlew	<i>Numenius arquata</i>	10	-	-
Broad billed Sandpiper	<i>Limicola falcinellus</i>	1	-	-
Jacana	<i>Actophilornis Africana</i>	1	-	-

Table 5.1: Waterbird Population Status at the MPRS (continued)

Common name	Scientific name	Maximum recorded at any onetime at Muni		Status
		1986-1998	2009-2010	
Terns & Gulls				
Common Tern	<i>Sterna hirundo</i>	8210**	676	Decrease
Sandwich Tern	<i>Sterna sandvicensis</i>	2230**	386	Decrease
Royal Tern	<i>Sterna maxima</i>	3200**	270	Decrease
Little Tern	<i>Sterna albifrons</i>	940	62	Decrease
Black Tern	<i>Chlidonias niger</i>	6570**	50	Decrease
Roseate Ten	<i>Sterna dougalii</i>	80*	-	-
Caspian Tern	<i>Sterna caspia</i>	1	-	-
Scooty Terns	<i>Sterna fuscata</i>	1	-	-
Black -headed Gull	<i>Larus ridibundus</i>	3	-	-
Lesser Black backed Gull	<i>Larus fuscus</i>	20	-	-
Herons, egrets & others				
Yellow Billed Egret	<i>Egretta intermedia</i>	6*	-	-
Black Heron	<i>Egretta ardesiaca</i>	12*	-	-
Grey Heron	<i>Ardea cinerea</i>	16	12	Decrease
Squacco Heron	<i>Ardeola ralloides</i>	-	1	-
Western Reef Egret	<i>Egretta gularis</i>	78	32	Decrease
Little Egret	<i>Egretta garzetta</i>	269	51	Decrease
Great Egret	<i>Egretta alba</i>	16	2	Decrease
Green-Backed Heron	<i>Butorides striata</i>	2	1	Decrease
Long-Tailed Cormorant	<i>Phalacrocarax africanus</i>	16	68	Increase
White face Tree Duck	<i>Dendrocygna viduata</i>	45	-	-

*Nationally significant numbers. **Internationally significant numbers
1986/1998 data from Ntiamoa-Baidu *et al.*(2000)

The changes in waterbird numbers could probably be attributed to (i) habitat alteration that had occurred at MPRS (e.g. Increasing siltation/sedimentation of the lagoon, which could probably reduce the feeding efficiency of the Curlew Sandpiper, as the sediment might become harder), (ii) the probable existence of more favourable or better wetlands along their migratory pathway, which they migrate to, neglecting the degrading coastal wetlands, (iii) the Ringed Plover having a greater ability to adapt to changing/

threatened environments than the Curlew Sandpiper. According to Gbogbo *et al.* (2009), Ringed Plovers were also the dominant wader species in the wetlands of Greater Accra Region. It is therefore important to pay critical attention to these changes in population composition of waterbirds as they could be indicative of serious wetland loss and alteration.

Coastal wetlands, such as the MPRS, and other wetlands around the world are subject to pressures from growing human populations, industrial and infrastructural development; leading to increasing rates of habitat change and pollution (Madsen, 1998; Willoughby *et al.*, 2001; White *et al.*, 2007). This study found that from January 1990 to April 2007, there had been serious alterations to the habitats used by waterbirds at the MPRS. More than 50% of the Muni lagoon and its exposed mudflats that serve as roosting sites for terns and other waterbirds, and represent an important feeding area for waders, had been lost in 17 years, as shown by the satellite imageries. Over the same period, the built-up (infrastructure) and bare surfaces have quadrupled and 99% of the herbaceous shrubs, which provide roosting sites for waterbirds such as herons, have been lost. The changes in the terrestrial parts of the Ramsar site could be the direct result of population growth, and its associated demands for farm lands as well as unsustainable activities such as fishing, sand winning, charcoal production and hunting. The reduction in the wetland areas could result from the loss of forest and degradation of the catchment areas. The human population of the MPRS increased from 32,000 in 1984 to about 50,000 in 2000 (Ghana Statistical Service, 1984; 2005). Human population growth exerts pressure on coastal wetlands but also leads to increasing infrastructural development and change in vegetation cover (Ntiemoa-Baidu and Gordon, 1991; Ryan and Ntiemoa-Baidu, 1998; Attuquayefio and Gbogbo, 2001).

Moreover, as population increases, more wetland habitat is encroached upon for residential and agricultural purposes. For example, since the main activities of the inhabitants of MPRS are fishing and farming, the increasing population would result in increasing demand for natural resources. Fuelwood harvesting, cattle grazing, sand and gravel mining which can deteriorate the texture of the soil are common activities carried out at the MPRS. As a result of tree cutting, most of the forest blocks (Yenku A and B) in the Ramsar site have been destroyed.

The study area has also been affected by the reduction in rainfall as a result of the increase in temperature prevailing in coastal areas of Ghana (Environmental Protection Agency (EPA), 2000). These factors, coupled with the vulnerability of the soil to erosion, as well as bushfires would account for the habitat degradation at the Muni wetland (Amatekpor, 1994; Gordon *et al.*, 2000; Wuver and Attuquayefio, 2006). When left bare, the soils of the MPRS, especially those with moderate to high erodibility, located on the upper ground on slopes (2-20%) and summits of the hills, ultimately end up being carried into the lagoon (Gordon *et al.*, 2000). Consequently, this would lead to siltation and/or sedimentation, thereby reducing the size of the lagoon, and probably leading to its pollution depending on the type of soil/rock from which the sediments are derived. This supports the findings that the moderately to highly erodible soils on slopes ranging from 2-25%, when left bare could yield up to 16 tonnes per year of sediment, which could eventually end up in the lagoon (Gordon *et al.*, 2000).

The loss of waterbird habitats and associated losses in suitable feeding and roosting sites would result in reduction in the use of the site by waterbirds. For instance, mangroves provide good high tide roosts for Whimbrels, herons, and Turnstones, apart from their role in ecosystem maintenance (Van de kam *et al.*, 2004). It is known that

waders belonging to Guilds 2, 3, and 4 would be most at risk (Gbogbo, 2007b) from habitat loss/fragmentation and pollution, in Ghana, due to their large numbers as decreasing trends have been reported in other areas such as the Banc d'Arguin in (Zwarts *et al.*, 1998), and other wetlands in South Africa and Senegal.

Of the populations of waders with known trends, 48% are known to be declining, compared with the 16% that are increasing (International Wader Study Group, 2003). Several researchers have attributed the world-wide decline in wader populations to human related unsustainable activities (International Wader Study Group, 2003; Stroud *et al.*, 2006) such as burgeoning coastal human populations, habitat changes resulting from land-claim, dredging, urbanisation and encroachment upon protected areas, water abstraction, loss of wetlands (Dolman and Sutherland, 1995; Goss-Custard *et al.*, 1995; Zöckler *et al.* 2003), shell-fisheries (Atkinson *et al.*, 2003), human disturbance (Liley, 2000, Burton *et al.*, 2002), and climate change (increasing temperature, decreasing rainfall resulting in the reduction of the numbers of temporal wetlands) (Dodman and Diagana, 2007). These factors in one way or the other affect the birds at their breeding, staging or wintering grounds during their migration.

According to Norris *et al.* (2004), bird numbers often change in response to habitat losses and additions through human actions, although it is also acknowledged that not all changes in waterbird numbers are the result of habitat changes. Thus, the reduction in the numbers of waterbirds recorded at the MPRS would be understandable, considering the habitat alterations that have occurred since its designation as a Ramsar Site.

It is known also that waterbirds either migrate to other wetlands along their flyway or relocate to neighbouring wetlands when conditions are unfavourable (Dodman and

Diagana, 2007; Newton, 2008). Although there is no evidence of a shift in waterbird abundance from one wetland to another, there is evidence of local movements among the coastal wetlands in Ghana (Ntiama-Baidu, 1991; Gbogbo, 2007b). For example Sanderlings ringed in Esiama in Western Ghana, have been recorded at other wetlands including MPRS.

Habitat alterations are known to affect birds in several ways. Birds are however affected not only by wetlands in which they breed or live, but also wetlands they use during their migration cycle; thus the slightest alteration could have a great impact on the diversity, abundance, habitat use and reproduction by the birds and future generations. Above all, waterbird species, or individuals have their preferred habitats. When conditions are altered, depending on the ability of the bird to adapt to the new environment or switch to another; they either stay or move away from the place (Zwarts *et al.*, 1992).

Although the MPRS no longer supports internationally important numbers of waterbird species, it still provides roosting sites for the rare Roseate Tern (*Sterna dougalii*). Some species (e.g. Redshank, Bar-tailed Godwit) were seen to change to the breeding plumage prior to migration to their breeding grounds, hence the importance of the site as a moulting ground. Terns use the site mainly for roosting, although a few Black and Little Terns have been recorded feeding in the lagoon and salt pans respectively during this study. The MPRS therefore still serves as important roosting, feeding, breeding and moulting grounds for waterbirds (Ntiama-Baidu *et al.*, 2000a). This could be attributed to the prevailing favourable conditions at the Muni lagoon such as (i) food availability and accessibility, (ii) limited competition and predation, and (iii) safe roosting sites among others.



5.2.2 Seasonal patterns in the occurrence of waterbirds at the MPRS

Waterbird species display different patterns in their distribution during their non-breeding period (Ntiamoa-Baidu, 1991), and this varies according to time (season, time of the day, etc.), sites (intertidal wetlands, lakes, etc) and species (Ntiamoa- Baidu *et al.*, 2000a; van de Kam *et al.*, 2004; Bolduc and Afton, 2008).

At the Muni-Pomadze Ramsar Site, variations in avian distribution and abundance were observed during the study period. The observed patterns can be categorised into three time periods with peak numbers in October and January, and decreasing numbers in November. The fluctuations in the number/ abundance of waterbirds at the MPRS are explained by the seasonal migration patterns of waterbirds. Palaearctic migrants arrive on the coasts of Ghana at the end of August/early September and maintain relatively high numbers till April (Ntiamoa-Baidu, 1991).

From this study, it can be concluded that there were two peaks of migratory waterbirds abundance at the MPRS as was observed in some years between 1986 to 1998 (Ntiamoa-Baidu *et al.*, 2000a). The peak month of wader abundance recorded varied from December (4 out of the 12 years 1986-1998 survey), January (2), February (2), September, October, November and March (recorded once for each, during migration) (Ntiamoa- Baidu *et al.*, 2000a). In other coastal wetlands of Ghana, Gbogbo (2007b) recorded the highest count in September 2005 at the managed wetlands (Densu Delta and Sakumo), and in March 2006 at the unmanaged ones (Laiwi and Mukwe Lagoons). The highest peak in wader counts recorded during this survey was therefore similar to those recorded in January 1988 and 1997.

The first influx of birds to the Muni site, which was dominated by terns, reached its peak in October, and by November, only a very small portion of the population was left at the site. It is well known that the peak period for terns occurring on the Ghana coast, is October (Grimes, 1979; Ntiama-Baidu, 1992; van der Winden *et al.*, 2002), thus explaining the October peak recorded at Muni during the survey. Furthermore, tern abundance on the Ghana coast during that period has been attributed to the increased availability and location of food sources in the coastal waters of Ghana (Grimes, 1979).

According to Grimes (1979), the seasonal upwelling offshore Ghana usually lasts for only a few months after it begins in August. During that period, shoals of *Sardinella* and anchovies, a good food source for terns, move from deeper waters to the surface. Terns therefore feed on these sardines and use the closest roosting sites available, such as exposed mudflats in coastal lagoons, and sand dunes, etc.

In late November, few of the sardines remain at the surface, forcing terns to move further offshore in search of new sources of food. There is evidence however that terns use the coasts of Namibia from February to March. From the time they leave Ghana to the period they appear in Namibia, there is a gap in knowledge concerning their locations (van der Winden *et al.*, 1996).

The second influx of migratory waterbirds occurred in January 2010. During this period, the most abundant birds at the site were waders which were dominated by Ringed Plovers. Waders stayed longer at the site, as, more than half of the January peak remained at the site in March. Even in the month of April that coincides with the northern spring migration when waterbirds move back to their breeding grounds, the numbers of waterbirds recorded were relatively high. This may be explained by the usual congregation of waterbirds at staging sites to refuel before final migration

(Summers *et al.*, 1995). The conditions at the staging sites may have severe consequences for the migration of waterbirds, and their breeding success. Based on the macrobenthic survey carried out during the study, it is clear that the high numbers of waders coincided with the peak in the density of polychaetes, their primary food source.

According to Ens *et al.* (1994), juvenile birds are late migrants due to the conspecific interactions that prevent them from getting enough food when feeding in the presence of adult birds; hence they may need more feeding time after the departure of the adult birds. This study did not establish whether or not the birds remaining on the site in April were juveniles. There is evidence that shorebirds use several coastal wetlands in Ghana (Macdonald, 1977; Ntiamoa-Baidu, 1991; Gbogbo 2007b), by moving from one wetland to another. Furthermore, Macdonald (1977) suggested that in Ghana, migrant waders use different wetlands at different times before final departure. There is also evidence that depending on their life-stages, some species of waterbirds may use the sites only as a stopover site for refuelling or as a non-breeding ground. Thus, throughout the northern winter, the MPRS remains important as a staging site by Palaearctic and inter African migrants.

5.2.3 Tidal differences in waterbird abundance

Multiple counts carried out on each sampling day from December 2009 to April 2010 showed that there are tidal influences on the occurrence and abundance of some waterbird species at the MPRS. Common Terns and Kittlitz's Plover occurred only at low tide. Sanderlings, Curlew Sandpipers, Royal Terns, Spotted Redshanks, and Grey Herons (*Ardea cinerea*), were most abundant at high tide, while Ringed Plovers and Little Stints were most abundant at low tide. Intertidal wetlands are affected by the level of tides, with increasing levels of tide resulting in water level increases, reduction of the

salinity in case of hypersaline ecosystems and increase in salinity with freshwater ecosystems. Tidal level also brings changes in food availability and affects preferences by feeding waders.

In terms of open lagoons, the tide brings new nutrients from the sea and also improves accessibility to food for species like Sanderling and increases in food intake as a result of the softening of the sediment. Low tide also helps to expose food items for small waders.

It is well known that the numbers of waterbirds using a site may vary seasonally and across the tidal cycle. Burton *et al.* (2004) explained this variation in numbers as a result of the varying availability of food through the tidal cycle, as well as the distance of sites from high tide roosts and their isolation from alternative habitats. These variations depend however on the sites and tend to be species-specific (species' energetic requirements, food preferences and feeding strategy, etc.). Burton *et al.* (2004) observed that numbers of Oystercatchers, Dunlins and Curlew Sandpipers decreased towards low tide.

Significant differences were observed in the counts of waterbird species belonging to Guild 2. This difference is expected since birds belonging to Guild 2 mostly feed on exposed mudflats. It therefore means that at low tide, larger areas are exposed, favouring the accessibility and availability of the macrobenthic prey they feed upon.

From the observations, it may be concluded that it is better to count terns during high tide as their numbers better represent the size of their population. For Waders, some species such as the Ringed Plover and Little Stint are better counted at low tide when

their numbers tend to be high, while others such as the Whimbrel and Grey Plover are best counted at high tide.

5.3 Habitat Selection by Waterbirds

Based on the use of the site by waterbirds, the Muni Lagoon was categorised into three main sections. Site E (sea water inflow); Site A, C, and D (mid-lagoon and its surrounding mudflats), and Site B and SP (freshwater inflow and Salt Ponds). Site E was located at the opening of the lagoon to the sea, and site SP and B were located at the northern section of the lagoon. The Sites A, C, and D were located at the middle part of the lagoon. The site varied in terms of salinity concentration and the distance from the shoreline.

During the dry season, Site E had lower salinity due to its closeness to the sea as sea water diluted the hypersaline waters at Site E; however during the rainy season, Sites B and SP had the lowest salinity. The middle portion of the lagoon had average salinity values that ranged between those recorded for the other two sections. This salinity gradient would result in variations in macroinvertebrate fauna distribution (Leland and Fend, 1998; Liang *et al.*, 2002), which in turn, would determine the distribution and abundance of waterbirds that depend on them. Several studies have also shown that prey distribution is the most important factor affecting the spatial distribution of waterbirds at intertidal areas (Colwell and Landrum, 1993; Moreira, 1995). In this study, the distribution pattern of the birds followed the same pattern as the macrobenthic species richness.

Colwell and Sundeen (2000) however reported that the main factors accounting for shorebird distribution on coastal beaches of Northern California were the proximity to

the main feeding areas and the variation in habitat features which were thought to influence food availability and foraging activity. This would imply that waterbird distribution could be determined by the morphological adaptations and activities of the individual species.

During migration, waterbirds select their habitat based on three main factors; (i) availability of food, (ii) safe roosting sites, and (iii) the extent of disturbance (Piersma, 1994). Food availability, the major factor, is determined by the morphology of the bird or species in question, its feeding requirements, and the wetland type and hydrology in which it finds itself. Thus wetlands with bigger habitat areas would be expected to attract greater number of species, due to the higher number of microhabitats they may provide (Paszkowski and Tonn, 2000; van de Kam *et al.*, 2004). This was however not the case at the Muni Lagoon, as sites C (mid-lagoon section), A (mudflat I) and SP (constructed Salt Ponds) scored the highest number of species and abundance, while the largest site, E, had the least species diversity. It is possible therefore that smaller wetlands could hold equally important numbers of waterbird species (Hudson, 1983; Garay *et al.*, 1991; Gbogbo, 2007b). It is therefore important to realise that the size of an area on its own does not account for the higher diversity recorded but rather its combination with other factors such as vegetation cover and structural heterogeneity of habitats within the wetland (Fairbairn and Dinsmore, 2001).

Habitat selection by waterbirds can be influenced also by the activity and requirement of the birds, since during migration, migratory birds are more likely to be found where they can maximise their food intake. Migratory waders, for instance, are observed to prefer intertidal mudflats during their non-breeding period (Rosa *et al.*, 2003).

Piersma and Ntiamoa-Baidu (1995) and Ntiamoa-Baidu *et al.* (1998) characterised waterbirds into seven guilds based on their functional responses to food (feeding style, mechanisms or identification of the food) as follows:

- herbivorous ducks,
- visual surface foragers waders (Plovers, Common Sandpiper, Turnstone, Redshank, etc.),
- tactile surface foragers waders (Godwits, Sanderling, Little Stints, Curlew Sandpiper, etc.),
- pelagic foragers waders (Greenshank, Black-winged Stilt, Avocet),
- stalking herons (Little Egret and Reef Heron),
- fishing pelicans and fishing terns (Black Tern, Royal Terns, etc.).

Hence at the MPRS, based on the distribution of the three most abundant species, it was observed that birds belonging to different guilds may display different habitat preferences. This might have resulted from the various morphological differences as well as the type of food they feed on.

Ringed Plovers, belonging to the Guild 2, are visual surface foraging species. They were recorded most at Site A during the sampling period, where they were found feeding continuously on worms from the soft mud. Sanderlings which belong to Guild 3 used the site D most. They were seen feeding mostly at the edges of the water and also from the mud in mixed flocks with Ringed Plover, Little Stint, and Curlew Sandpiper. They usually follow the water movement caused by the wind or tide. The second favourite habitat, Site SP was most used for roosting. Sanderlings were found usually roosting in flocks on the sand or the exposed mudflats found within some of the unused ponds. Greenshanks were recorded mostly at site SP. This species belonging to Guild 4,

exploited the areas with relatively high water levels. They displayed different feeding strategies varying from pecking, sweeping to ploughing in an organised sequential manner. They were seen roosting with the Little Egrets and Curlew Sandpiper.

During this study, the distribution of terns was influenced by the tidal cycle and the availability of small islands within the lagoons. These islands were located in Sites E and C. Water depth at Site A was relatively stable and low, therefore exposing the mud that served as roosting sites for the terns, which used the Muni Lagoon mainly as roosting sites.

During the day, terns were seen to use more than one roosting site depending on the tide level. At high tide, they are located at Site E, and at ebbing or low tide, or even when the tide is rising they could be found roosting in Site C.

5.4 Waterbird Activities at the Muni-Pomadze Ramsar Site

The main activities of waterbirds recorded during this study were roosting, feeding, and comfort activity. These activities varied according to site and time based on the spatio-temporal changes in the importance of the different sites at the MPRS. Variations in the number of waterbirds roosting and feeding were observed for the different sites and months sampled. A chi-square association test between sites and months showed significant interaction in terms of number of feeding birds ($\chi^2=1951.27$, $P < 0.001$, 35df) and that of roosting birds ($\chi^2=5624.66$, $P < 0.001$, 35 df). This showed that the sites displayed different characteristics depending on the time of the year.

5.4.1 Comfort activities of waterbirds at MPRS

The proportions of waterbirds engaged in comfort activities showed that they spent more time either feeding or roosting than preening or bathing. A similar study of waterbird activities at Songor and Keta (Ntiamo-Baidu *et al.*, 1998) showed that comfort activity was the behavioural activity recorded least often. For the 33 species recorded at Muni lagoon, the proportion of the birds observed during daytime that were engaged in comfort activity ranged from 0 to 2.48%, as compared to 2-31% of the 25 birds species observed by Ntiamo-Baidu *et al.* (1998) at other coastal lagoons in Ghana. The proportions recorded during this study are also lower than the 0.3-25% for the over 62 bird species recorded by Cotgreave and Clayton (1994).

5.4.2 Roosting activity of waterbirds

From this study, it can be concluded that Sites SP, E, and C were used most by roosting birds. The two Sites, E and C, are natural habitats of the Muni lagoon within which small islands are located. The population of waterbirds using the Sites E and C as roosting sites was dominated by terns, as a result of their preference for small islands surrounded by open waters (Ntiamo-Baidu, 1992; van de Kam *et al.*, 2004).

Site SP (constructed Salt Ponds), was used mainly by roosting waders and herons. They used both the banks and the ponds themselves depending on the water levels for this purpose, especially during high tide and depending on the time of the day. When temperatures are high or at dusk, the numbers of roosting birds increased at the SP. The distribution at the roosting site of wader is as follows: Spur-winged Plovers on drier places, Sanderlings and Curlew Sandpiper, Little Stint together on the shallowest parts

of the salt ponds, and Greenshanks and Herons in the deeper parts of the Salt pans. Herons were also observed on the banks of the ponds.

Artificial habitats located closed to natural foraging habitats (Brusati *et al.*, 2001) are alternative habitats for waterbirds that move from their preferred habitats such as beaches and mudflats during high or low tides (Erwin *et al.*, 1994, Burger *et al.*, 1997) to the constructed ponds. Salt ponds not only provide good roosting sites for waterbirds, but also serve as foraging habitat for large numbers of migratory waterbirds (Takekawa *et al.*, 2001; Warnock *et al.*, 2002). Young (1998) recorded more herons feeding and roosting on coastal fish ponds.

The use of high tide roosts often varies among species and largely depends on the distance to the nearest favourable feeding areas and the potential risk of disturbance (van de Kam *et al.*, 1999). Human activities impose a potential threat, especially, with high tide roosts (Blanc *et al.*, 2006). Activities such as fishing that occur at MPRS affected the behaviour of roosting and feeding waterbirds. According to Blanc *et al.* (2006) birds perceive human beings as potential predators. At Songor lagoon, roosting terns, for instance were observed to fly out whenever people crossing the lagoon approach the roost (Ntiamoa-Baidu *et al.*, 1998).

5.5 Feeding Activity of Waterbirds at MPRS

The feeding activity of waterbirds at the MPRS is essentially determined by that of waders and herons, since terns used the site mainly for roosting. In order to maximise their prey intake during migration, waterbirds, especially waders, were observed to feed not only in areas of high prey density but more in areas of easy accessibility of suitable prey (Kushlan 2000; Gawlik, 2002). They therefore select profitable prey or profitable feeding areas. At the MPRS, foraging waterbirds were most abundant at Sites A and D, the two sites with mudflats of the Muni Lagoon. This observation confirms the importance of intertidal mudflats as feeding sites for migratory waders (van de Kam *et al.*, 2004; Zou *et al.*, 2006). The selection of these feeding sites is not haphazard, but a strategy by migrant waders that need to meet physiological demands during their non-breeding and/or refuelling periods that will ensure timely return to their breeding grounds.

Even though Site A did not record the highest density of macrobenthic prey upon which waders depend, it had the lowest water depth and therefore the highest food accessibility. It has been well-established that hydrological fluctuations within and among wetlands often dictate where and when waterbird species can access their food (Kingsford *et al.*, 2004).

In coastal wetlands, waders are more abundant when their foraging areas are exposed, and their prey items become exposed. Low water depth is therefore critical for foraging birds, and hence the importance of low tide for feeding waders (Burton *et al.*, 2004). Young and Chan (1997) pointed out that large wading birds such as herons and egrets choose feeding sites in drained fish ponds where water is shallow and fish are more concentrated. However, this cannot be generalised, since different waterbird species

have different water depth requirements (Bolduc and Afton, 2004; 2008). Even among waders, species belonging to different guilds may have different water depth range requirements. Ntiamoa-Baidu *et al.* (1998) found that the feeding habitat of waders in the Keta Lagoon ranged from dry mudflats to wet mud, and shallow water of not more than 20 cm. Ringed Plovers, for instance, fed on dry mud; Curlew Sandpipers on wet and water depth below 5 cm and Stilts and Greenshanks in deeper shallow waters (Piersma and Ntiamoa-Baidu, 1995), whereas diving birds, such as fishing terns and cormorants are probably not as limited by water depth. Several studies have found that waterbird density is highest at low water depth. It has been explained that there should be an optimum water depth at which waterbirds are expected to maximise their resource utilisation, otherwise it will mean that waterbird numbers are highest when water depth is virtually zero (e.g. Kingsford *et al.*, 2004; Holm and Clausen, 2006; Bolduc and Afton, 2004).

The key role played by water depth in the selection of feeding habitat by waders is recognised (Ntiamoa-Baidu *et al.*, 1998; Liang *et al.*, 2002). It could thus be assumed that waterbird communities differ among areas showing different hydrobiological regimes at a given time, since different species have different depth requirements (Bolduc and Afton, 2004; 2008). Water depth was found to affect feeding waders differently depending on the habitat in which they were and morphological constraints (Safran *et al.*, 1997).

The second probable reason for the abundance of feeding birds in Site A, is that Site C is highly disturbed/polluted as compared to A as shown by the Abundance/Biomass Comparison curves. In highly disturbed habitat as in the case of Site C, the biomass of macrobenthic invertebrates is lower than that of a site with lower disturbance. Since

migratory birds always try to maximise their food intake or energy, they most often opt for prey items with high biomass and that are easily accessible.

Furthermore, sediment type and hardness are also factors that could limit food accessibility by feeding waterbirds. Hence the lower number of feeding waterbirds recorded at site C could also be attributed to the abundant shells on the ground that could probably limit the feeding activity of the birds. Most waders prefer soft mud or sand, where they can easily probe.

No correlation was found between density of shorebird and/of their prey as suggested by Goss-Custard *et al.* (1977), Bryant (1979), Meire (1993), and Kalejta and Hockey (1994). There was however, a correlation between the abundance of feeding birds and the biomass of polychaetes sampled at Site A (mudflat I). This is probably due to the dominance of birds that feed by visual foraging method (Guild 2), such as ringed plovers. This is in agreement with Kalejta and Hockey (1994) who stated that visual foraging birds are better predicted by prey biomass rather than density.

The periodic movements or migration of a population or part of a population of an animal species away from a region and their subsequent return to that same region is caused by the local depletion of harvestable prey and high cost of maintenance during unfavourable conditions (Ens *et al.*, 1994). The refuelling periods involve the use of stopover or staging sites for feeding and resting which helps in the accumulation of energy, and is followed by alternate flight periods during which energy is consumed. Migratory waterbirds generally are bound to fixed time-schedules (Piersma, 1994); hence their heavy dependence on the resources found at their stop-over sites.

The peak wader period that occurred in January 2010 corresponded to a peak in the density of macrobenthic invertebrates at the Muni lagoon. Even though no definite conclusion can be made concerning a causality effect between the two, it is necessary to highlight the high density of invertebrates at the site at a time when bird numbers were also high.

The findings of this study show that several factors influence waterbird use of Muni-Pomadze Ramsar Site. At the landscape scale, individual sites varied in waterbird use as a response to seasonal changes in the ecological characteristics of the sites (Mungula *et al.*, 2005).

Variations were recorded in the hydrological regime, the macrobenthic invertebrate community, as well as changes in waterbird species diversity, abundance and distribution. The fluctuations in water depth and quality, coupled with the spatiotemporal variations in the density of macroinvertebrates at the Muni lagoon determined the use of the different sites as feeding habitats over the study period.

5.6 Waterbirds as Bioindicators of Wetland Quality

Waterbird species and populations are sensitive to changes in their ecosystem; hence their importance as bioindicators in wetland monitoring programmes (Furness and Greenwood, 1993). Continuous long-term monitoring of the populations of bird species can provide useful data in describing the implications of anthropogenic activities for habitat condition and biodiversity (Owusu *et al.*, 2002). Waterbird species and populations are used as bioindicators of environmental health with regards to pollution,

habitat degradation, vegetation disturbance, water quality and resource productivity as well as food abundance (BirdLife International, 2004; Ahulu *et al.*, 2006).

The habitat alterations that had occurred at the MPRS in the past 15 years would consequently affect its fauna. This study recorded decreases in the maximum counts of 26 waterbird populations recorded at the MPRS. Studies by Norris *et al.* (2004) and Newton (2004) have shown that Redshank populations, for example, declined as a result of nesting sites destruction due to land drainage in summer. Wetland quality, from the perspective of waterbirds can be described *inter-alia*, as the water quality, food availability, as well as roosting and nesting sites, which determine the welfare of a bird or bird community assemblages. Wetland quality is the most holistic factor that influences waterbird communities.

The results of this study showed that Site C (Mid-Lagoon portion) had a higher density of macroinvertebrates than Site A (Mudflat I); however, Site A recorded the highest proportion of foraging birds. This would suggest that foraging waterbirds would prefer a poorer site in terms of prey density with good food accessibility than one with high density but poorer accessibility (Rapport, 1991; Railsback *et al.*, 2003). Since the quality of a habitat or an environment is its ability to provide conditions appropriate for individual and population persistence of birds (Hall *et al.*, 1997), it is obvious that it goes beyond the scope of the birds immediate surroundings and the resources necessary for their survival and reproduction, but also the conditions that constrain their use (Morrison *et al.*, 2006).

The macrobenthic fauna of the Muni Lagoon is dominated by capitellid worms, and couples with the high mean salinity and temperature and their fluctuations indicate that conditions at the Muni Lagoon are typical of a stressed environment. It would therefore

be reasonable to suggest that in this case waterbirds are not good indicators of wetland quality, health or status, since the site still supports appreciable numbers of waterbirds. Further behavioural studies would be important therefore to complement the use of waterbirds as bioindicators of wetland quality. Temple and Wiens (1989) found that waterbirds are not early indicators of habitat degradation and proposed that birds may use a site till they exhaust all its natural resources before moving completely away.

Considering whether or not waterbirds are early indicators of wetland degradation, Johson (2007) stressed the need for complementary studies and classified bird-based indicators of habitat quality into broad groups, demographic, distributional and individual condition measures, to which behavioural observations could be added and the life history of the waterbird population also taken into consideration. It follows, therefore, that it is necessary to consider the response of birds to all factors in their environment in order to make conclusive deductions on the role of the birds in the wetland ecosystem and their value as bioindicators of wetland quality.

CHAPTER SIX

CONCLUSIONS AND RECOMMENDATIONS

6.1 Conclusions

The populations of waterbirds recorded at the MPRS suggest that the importance of the site for waterbirds currently is lower than what it was at the time of its designation as a Ramsar Site. The populations of the four tern species (Royal, Sandwich, Black and Common) which occurred in internationally important numbers at the time of designation were all far below the 1% threshold required to qualify for international importance. However, the MPRS still supports some nationally important populations of Spur-Winged Plover, Whimbrel, Wood Sandpiper, Common Sandpiper and Sanderlings. MPRS also provides valuable roosting sites for terns as well as feeding grounds for the non-breeding waders.

Habitat use varied among waterbird groups, with terns using the lagoon mainly for roosting, more specifically Site E (seaward section) and C (mid-Lagoon portion), and waders feeding more at Site A and D, the two mudflats surrounding the lagoon, roosting at Site SP, the constructed salt ponds. Herons were the least abundant at the MPRS and used Site B (freshwater inflow) and SP (Salt Ponds) most. Nevertheless spatiotemporal variations in the ecological character of the sites resulted in temporal variations in the activities of the birds and the macrobenthic density.

The influence of water quality on waterbird activities varied according to the sites. Of the water parameters measured, only salinity, conductivity and temperature significantly influenced roosting activity of waterbirds. The feeding activity of waterbirds in the

Muni Lagoon was influenced by hydrological fluctuations that affected the accessibility and availability of food. In Site A, the number of feeding birds was correlated to the macrobenthic biomass instead of density.

The Muni- Pomadze wetland ecosystem is degrading due to natural (vulnerability of soils to erosion, and large expanse of shrubs sensitive to bush fire) and anthropogenic activities (urbanisation, agriculture, population growth, etc.). As such 99% of the herbaceous shrub is lost, built-up areas have quadrupled and the Muni Lagoon has reduced to about 50% of its original size.

At the Muni Pomadze Ramsar Site, the macrobenthic community is dominated by the opportunistic *Capitella* spp. which is known to proliferate in such stressful environments. The macrobenthic fauna of the Muni lagoon experienced variations in response to the fluctuating hydrology and water quality during the study period. The two peaks in macroinvertebrate abundance at Muni lagoon recorded in January and March 2010 showed that temporal variation in the density of benthic organisms could be linked to increasing rainfall in the area. It was also observed that the highest counts of waders were recorded in January when the highest density of benthic organisms was recorded. The macrobenthic community structure of the Muni Lagoon has not changed much since it is still dominated by only one taxon.

Even though salinity of the Muni Lagoon has reduced, its fluctuations make the lagoon it a very stressful environment. Both waterbird and benthic species are likely to be affected by temperature and salinity stress in the Muni ecosystem. Turbidity of the Muni waters was also recorded to be high probably due to the shallowness of the lagoon. Low variability was recorded in pH and D.O.

Despite the habitat losses and the general deterioration of the health of the MPRS, the site continues to support significant numbers of waterbirds, although the populations have gone down. The findings of this study thus support that waterbirds are not early indicators of wetland quality. Depending on what has been lost or gained, a change in the environmental health of a wetland may not necessarily result in immediate changes in the waterbird community. Thus by the time the birds stop using the altered site, the site may be degraded beyond redemption.

6.2 Recommendations

As a result of this study, the following recommendations can be made:

- Habitat alterations that have occurred at the Muni-Pomadze Ramsar Site have led to the deterioration of its ecological character. The numbers of most waterbird species using the site have decreased and based on the current study, none of the waterbird species occurred in internationally important numbers. It is recommended therefore that the bird monitoring at the MPRS continue over a longer period to enable the re-assessment of the international importance of the site.
- The changes in the populations of the most abundant species of waders, the Curlew Sandpipers, should be investigated further. Curlew Sandpiper population is reducing in coastal wetlands of Ghana even though there is an increase in the global population.
- The use of waterbirds as bioindicators should be thoroughly understood and investigated before implementation. It should also be complemented with

measures of other biological and ecological parameters in order to provide adequate information on the habitat.

- The Muni-Pomadze Ramsar Site can be zoned further into southern, eastern and northern sections for management purposes, and the degree of protection to be provided.

- The management of the site should also be reinforced and accompanied by a habitat restoration program. The Muni lagoon needs to be restored, especially with the prevailing climatic instability, to prevent it from drying up completely in the very near future.

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APPENDICES

Appendix I: GPS coordinates for the sampling points

SITE	N	W
A	00°38'31"	05°19'33"
B	00°38'41"	05°20'05"
C	00°38'39"	05°19'42"
D	00°38'39"	05°19'51"
E	00°39'03"	05°19'24"
SP	00°38'15"	00°20'14"

Appendix II: Water Quality Analysis of the Muni Lagoon

Water quality							
Month	Temperature	Water depth	turbidity	Salinity	D.O	Conductivity	pH
A							
SEPT	28.2	12.5	63.7	38	5.8	58.08	8.11
OCT	32.5	12.5	21	36			8.3
NOV	27.9	12	73.5	20.2		68.98	8.5
DEC	33.6	13.67	89.3	45.3	8.3	44.2	8.5
B							
SEPT	29.8	22	33.7	35	5.9	62.24	8.11
OCT	32	32	21	40			8.6
NOV	28.2	35.5	48.2	38.4		67.04	8.7
DEC	33.3	26	19.4	45.67	5.5	68.4	8.5
JAN	33.9	21	15	40	7.9	74.15	8.12
FEB	36.9	30	29.5	36	7.6	59.36	8.43
MAR	39.5	18.7	7.95	39	6.9	62.44	8.21
APR	30	14	7.58	29	10.6	45.56	8.1
C							
SEPT	25.3	32	14.8	37	6	58	8.08
OCT	32.9	21.7	13	40			8.4
NOV	28.2	32	50.4	38.6		55.4	8.4
DEC	36	30	7.21	41	6.2	48.2	8.45
JAN	37.2	35	20	39	8.2	58	8.23
FEB	37.6	32	16.6	38	7.8	41.96	8.8
MAR	38.6	20.7	5.04	40	9.9	54.32	8.1
APR	30.9	25	19.54	40	8.9	73.64	7.8
D							
SEPT	27.3	31	29.6	38	6	61.56	8.14
OCT	34	31	35	39			8.3
E							
SEPT	25.2	40	29.6	39	5.8	63.28	8.06
OCT	31.2	45	9.6	37			8.4
NOV	27.9	17	46	25		46.28	8.33

DEC	35	22	46.2	40	5.6	38.6	8.7
							7.8
JAN	33.8	36	29	38	6.2	56.55	8
							7.6
FEB	34.7	39	8.4	40	6.4	42.88	5
MAR	33.5	39	5.71	38	8.6	55.64	8
APR	31	39.7	13.85	38	7.3	44	8.1

Water quality							
SP							
Mont h	temperature	Water depth	turbidit y	Salinit y	D.O .	Conductivit y	pH
DEC	32	11	21.7	55	5.5	74.8	8.2
							8.0
JAN	33.6	35.5	12	51.5	6	67	1
							8.1
FEB	32.2	30	36.2	38	6.2	57.28	4
							7.9
MAR	32.9	34	18.4	50	7.8	72.28	8
							7.4
APR	27.5	21	44.53	106	6.7	72.8	3

Appendix III: Spatio-temporal Variation in Major Feeding Groups at MPRS

Spatio temporal variation in major feeding groups at MPRS								
WADER								
	SEPT	OCT	NOV	DEC	JAN	FEB	MAR	APR
A	137	284	135	162	66	292	299	479
B	18	38	12	89	150	33	59	48
C	138	74	77	170	191	92	22	39
D	96	85	0	118	532	167	24	67
E	15	15	6	16	29	35	6	6
SP	203	229	5	173	102	321	313	137
TOTAL	607	725	235	728	1070	940	723	776

HERON								
	SEPT	OCT	NOV	DEC	JAN	FEB	MAR	APR
A	6	5	2	3	1	3	28	8
B	1	2	3	2	47	1	9	11
C	2	0	0	13	3	1	2	8
D	1	2	0	1	12	6	1	5
E	28	5	2	6	4	5	3	1
SP	107	14	11	20	6	2	28	21
TOTAL	145	28	18	45	73	18	71	54

TERN								
	SEP	OCT	NOV	DEC	JAN	FEB	MAR	APR
A	3	87	411	0	0	0	0	0
B	1	2	0	0	0	0	0	0
C	478	747	0	0	6	0	17	51
D	9	42	0	0	0	0	0	0
E	10	1	0	261	500	107	68	0
SP	1	2	0	0	0	0	0	0
TOTAL	502	881	411	261	506	107	85	51

APPENDIX IV: Biota-Environment Interactions**Effect of environmental parameters on Benthic species**

Variable	Marginal Effects		Conditional Effects		
	Var.N	Lambda1	LambdaA	P	F
Water depth	2	0.16	0.16	0.025	3.27
turbidity	3	0.16	0.04	0.62	0.76
pH	5	0.11	0.09	0.14	1.87
Temperature	1	0.03	0.02	0.77	0.5
Salinity	4	0.03	0.02	0.855	0.38

Effect of environmental parameters on Benthic major taxa

Variable	Marginal Effects		Conditional Effects		
	Var.N	Lambda1	LambdaA	P	F
Water depth	2	0.03	0.03	0.07	2.36
Salinity	4	0.01	0.01	0.775	0.27
turbidity	3	0	0	0.99	0.01
pH	5	0	0	0.485	0.68
temperature	1	0	0	0.85	0.16

Effect of environmental parameters on waterbird abundance

Variable	Marginal Effects		Conditional Effects		
	Var.N	Lambda1	LambdaA	P	F
temperature	1	0.04	0.04	0.2	1.24
pH	5	0.02	0.02	0.595	0.38
turbidity	3	0.01	0.01	0.655	0.35
Salinity	4	0	0.01	0.745	0.28
Water depth	2	0	0.02	0.5	0.63

Effect of environmental parameters on waterbird activity

Variable	Marginal Effects		Conditional Effects		
	Var.N	Lambda1	LambdaA	P	F
temperature	1	0.06	0.06	0.24	1.24
Salinity	4	0.04	0.03	0.53	0.62
Water depth	2	0.03	0.06	0.21	1.4
turbidity	3	0.02	0.04	0.435	0.78
pH	5	0.02	0.03	0.58	0.49

Appendix V: Habitat Use by Waterbirds

Survey periods	Total sightings	No. Days spent	Total days spent by birds				
Northern Autumn							
Survey periods	Total sightings	No. Days spent	Total days spent by birds	waders	terns		
5 Sept	1254	70	87780	607	42490	502	35140
9 Oct	1634	36	58824	725	26100	881	31716
14 Nov	664	14	9296	235	3290	411	5754
Total days spent during Northern Autumn			155900		71880	72610	
Northern Winter							
Survey periods	Total sightings	No. Days spent	Total days spent by birds				
16 Dec	1034	66	68244	728	48048	261	17226
13 jan	1649	38	62662	1070	40660	506	19228
20 feb	1065	20	21300	940	18800	107	2140
Total days spent during Northern Winter			152206		107508	38594	
Northern Spring							
Survey periods	Total sightings	No. Days spent	Total days spent by birds				
15 Mar	879	26	22854	723	18798	85	2210
10 Apr	881	10	8810	776	7760	51	510
Total days spent during Northern Spring			31664		26558	2720	

