Assessing the potential impacts of climate change on coastal wetlands in North-Eastern NSW using geoinformatics

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ASSESSING THE POTENTIAL IMPACTS OF CLIMATE CHANGE ON COASTAL WETLANDS IN NORTH-EASTERN NSW USING GEOINFORMATICS

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Bachelor of Science (Honours), Master of Science

A thesis submitted in fulfilment of the requirements for the degree of Doctor of Philosophy

SOUTHERN CROSS UNIVERSITY School of Environmental Science and Management
Lismore, New South Wales Australia

AUGUST 2011
DECLARATION

I certify that the work presented in this thesis is, to the best of my knowledge and belief, original, except as acknowledged in the text, and that the material has not been submitted, either in whole or in part, for a degree at this or any other university.

I acknowledge that I have read and understood the University's rules, requirements, procedures and policy relating to my higher degree research award and to my thesis. I certify that I have complied with the rules, requirements, procedures and policy of the University (as they may be from time to time).

Print Name: Clement Elumpe Akumu

Signature:..................................................

Date: ..................................................
ABSTRACT

The coastal wetland communities of north-eastern New South Wales (NSW) Australia exist in a subtropical climate with high biodiversity and are already under threat due to several anthropogenic factors such as urbanisation, residential development and agricultural development. In addition, they are also potentially threatened by the continuous variation in climate. Nevertheless, there is no known research about the extent of the potential impacts of climate change on these delicate yet dynamic ecosystems.

The aim of the study is to assess the potential impacts of climate change on the coastal wetlands in north-eastern NSW, Australia. In assessing the potential impact of climate change, it is important to examine a range of issues including the present extent and types of wetlands in the study area, the environmental requirements and tolerances of keystone wetland flora, inundation by elevated sea level and changes in biogeochemical processes that may result from elevated temperatures. The objectives are: (i) to provide an overview of wetland classifications in relation to their communities and their environmental variables with an emphasis on NSW; (ii) to map the current and past wetland communities in north-eastern NSW in order to identify any changes in quality and extent; (iii) to predict the potential spatial distribution of selected wetland species (*Avicennia marina, Banksia integrifolia, Melaleuca quinquenervia* and *Leptospermum liversidgei*) as a result of climate change (mean annual temperature increase); (iv) to predict the potential impact of sea level rise on the coastal wetland communities by the end of the century; (v) to estimate the amount of methane emission from the coastal
wetlands using satellite data and to estimate emission with a temperature increase and (vi) to provide management, mitigation and adaptation strategies that could be used to minimize the impacts of climate change on the coastal wetland ecosystems.

Landsat TM satellite imagery of September 1989, February 2009 and Landsat ETM+ of June 2001 were used to identify, map and monitor the wetland communities. Supervised classification was performed using the maximum likelihood standard algorithm. Normalized Difference Vegetation Index (NDVI) was produced and the health of the wetland vegetation was evaluated. Bioclimatic modeling such as BIOCLIM was used to predict the potential spatial distribution of the wetland species under current and future climatic scenarios. Sea Level Affecting Marshes Model (SLAMM) was used to predict the potential impacts of sea level rise on the coastal wetland communities. A process-based methane emission model that included a productivity factor, temperature dependent (T factor), wetland area, methane flux, precipitation and evaporation ratio was used to estimate the amount of methane emission from the wetlands. The temperature dependent factor was obtained through land surface temperature (LST) estimation algorithms. Measurements of methane fluxes from the wetlands were performed using static chamber techniques and gas chromatography. Geographic Information System (GIS) provided the framework for mapping, modeling and analysis.

The study found significant changes in the quality and extent of the coastal wetland communities in the months of September 1989, June 2001 and February 2009. Furthermore, it was found that a rise in mean annual temperature beyond 7°C would
likely lead to a complete loss of suitable habitats for the wetland plant species (*Avicennia marina*, *Banksia integrifolia*, *Melaleuca quinquenervia* and *Leptospermum liversidgei*) in north-eastern NSW. In addition, a meter rise in sea level could decrease coastal wetlands such as inland fresh marshes from about 225.67 km$^2$ in February 2009 to around 168.04 km$^2$ by the end of the century. High variability of methane emission was also found from the coastal wetlands. Forested wetlands produced the highest amount of methane *i.e.*, 0.0016±0.00009 teragrams (Tg) in the month of June, 2001. This would increase to about 0.0022±0.0001 Tg in the month of June with a 1°C rise in mean annual temperature by the year 2030 in north-eastern NSW.

This research provides valuable information that could be used by coastal wetland managers in planning and conservation. In order to enhance the conservation of these ecosystems, effective management strategies such as protection and buffering of existing and potentially suitable habitats is recommended. Furthermore, the implementation of efficient mitigation and adaptation strategies of climate change could alleviate the devastating impacts on these sensitive wetland ecosystems.
LIST OF PUBLICATIONS RESULTED FROM THIS THESIS

**Peer-Reviewed Journal Articles**


**Conference Proceedings and Abstracts**

1) Pathirana, S., **Akumu, C.E.** and Baban, S. (2009) *Examining the impact of climate change on the spatial distribution of coastal wetland communities in north-eastern NSW.* Proceedings of the 32^nd^ Applied Geography Conference, October, 28-31, Louisiana, USA


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My research involved frequent field visits in order to collect gas samples, identification of wetland types and plant species, acquisition of geographic coordinates, ground truthing in order to perform validation and accuracy assessments in the coastal region of north-eastern NSW. I want to thank Max Egan who assisted me at the start of my field work and Paul Kelly who accompanied me throughout my field visits. Many thanks to Dr Dirk Erler and Dr Joanne Oakes at Southern Cross University who assisted me with the preparation of the gas sampling protocol. I also acknowledge the immense technical support received from Greg Luker- GIS Lab Manager.

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CHAPTER ONE

1.0. GENERAL INTRODUCTION

With growing evidence of a human influence on global atmospheric greenhouse gases and theoretical predictions of increased average temperature and consequent sea level rise, managers of natural resources need tools with which they can predict and therefore ameliorate or respond to changes in the productive status of those resources. Among the most vulnerable natural ecosystems to climate change are coastal wetland ecosystems, because they will potentially be affected by changes in temperature and sea-level. The science of geoinformatics provides a suite of tools with which changes in environmental parameters can be mapped onto existing topography to predict the space within which a particular habitat type could potentially exist in the future and where it will no longer be able to persist. This project uses a GIS, Remote sensing and GPS approach to map and model the potential impact of climate change on the coastal wetlands in north-eastern NSW, Australia. The current and potential spatial extent of the wetland communities are modelled under a range of future climatic scenarios. Not only can the spatial extent of wetland habitats be mapped and modelled but this can be linked to changes in ecological processes that occur within these habitats such as productivity, or consumption or production of nutrients or gases. As an example, this project models the changes likely to occur in the production of methane from coastal wetlands under a future climate scenarios. Based on the modeling outcome from this project, suitable climate change mitigation and adaptation strategies can be provided in order to enhance wetland conservation.
1.1. CLIMATE CHANGE AND SEA LEVEL RISE

According to Warwick et al. (1993), climate change refers to trends and fluctuations in climatic factors such as carbon dioxide content of the atmosphere, temperature and rainfall. The current and projected climate change is caused through the ‘greenhouse effect’ whereby gases such as carbon dioxide, methane, nitrous oxide, and fluorocarbons absorb and re-emit solar energy (IPCC, 2007a; Mitchell, 1989). About one-third of radiating energy from the sun at very short wavelengths, predominantly in the visible or near-visible part of the spectrum that reaches the earth’s atmosphere is reflected directly back to space. The remaining two-thirds are absorbed by the earth’s surface and to a lesser extent by the atmosphere. In order to balance the absorbed incoming energy, the earth has to radiate the same amount of energy on average back to space. However, due to the fact that the earth is much colder than the sun, it radiates energy at much longer wavelengths, mainly in the infrared part of the spectrum. Nonetheless, most of the thermal radiation emitted by the land and ocean is absorbed by the atmosphere, including clouds and reradiated back to earth (IPCC, 2007a; Mitchell, 1989). This greenhouse effect process warms the surface of the planet and preventing it from falling below the freezing point of water. The dominant atmospheric gases such as nitrogen and oxygen contribute little to the greenhouse effect. Instead, gas molecules that are more complex such as water vapour, carbon dioxide, methane, nitrous oxide and ozone mostly contribute to greenhouse effect (Hardy, 2003; IPCC, 2007a). This implies that if human activity raises the concentrations of these greenhouse gases, this will result in more solar energy being absorbed and reradiated back to earth, thereby inducing global warming.
Anthropogenic emissions of carbon dioxide (CO₂) have been increasing since the late 1950s compared to its emission from natural cycles in the past (IPCC, 2007a). Charles Keeling in 1958, initiated a program of high-accuracy measurement of atmospheric CO₂ and this forms the framework for documenting the changing composition in the atmosphere (Keeling, 1961, 1998).

From the air composition analysis carried out in Greenland and Antarctica, it was found that CO₂ abundances were significantly lower during the last ice age than over the last 10,000 years of the Holocene (Berner et al., 1980; Delmas et al., 1980; Neftel et al., 1982). From 10,000 years ago up to 1750, CO₂ abundances in the atmosphere remained relatively stable within 280±20ppm (Indermühle, 1999). During the industrial era which began in the late 18th century, CO₂ increased exponentially to approximately 367 ppm in 1999 (Etheridge, 1996; IPCC, 2001; Neftel et al., 1985) and to 379 ppm in 2005 (IPCC, 2007a). Furthermore, methane and nitrous oxide have also increased in the atmosphere over the same time period (Steele, 1996). Methane had a relatively constant abundance of 700ppb prior to the 19th century. From the 19th century, methane abundance has been on a steady increase with an abundance of 1,745 ppb in 1998 (IPCC, 2001) and 1,774 ppb in 2005 (IPCC, 2007a). This increase can be readily explained by anthropogenic emissions (IPCC, 2007a). The increasing trend is also similar to nitrous oxide which showed an increase in abundance of 314 ppb to 319 ppb in 1998 and 2005 respectively (IPCC, 2001, 2007a) compared to the range of 180-260 ppb prior to the industrial era (Flückiger, 1999). Even though, some experimental measurements of greenhouse gases have been carried out in the Antarctica and Greenland (Berner et al., 1980; Delmas et al., 1980; Neftel et al., 1982), there are still gaps in our understanding of emission estimates. The
emission estimates of greenhouse gases such as methane are still not available at both local and national scale in most parts of the world especially in Australia.

The increasing temperature of the earth’s surface (global warming) caused by a rising greenhouse gases was first noticed by the analysis of deep ice cores from Greenland (Dansgaard, 1984). This was further supported by the synthesis of palaeoclimatic observations carried out by Broecker and Denton (1989) and by the end of the 1990s, abrupt global temperature changes had became apparent (IPCC, 2001). Instrumental observations over the past 157 years showed that global surface temperature has risen with significant regional variations. The earth’s warming phenomenon has occurred in two phases in the last century. On a global average, the surface temperature has increased by 0.35°C in the period of 1910s to 1940s. From 1970s to present, there is a stronger increase of 0.55°C with predictions of between 1°C and 6.4°C by the year 2099 (IPCC, 2001). This would have significant impacts on the earth’s ecosystems especially those affected by changes in rainfall, temperature and sea level rise such as coastal wetlands.

A ‘wetland’ is defined in this research as an area with a shallow water body, fresh, brackish or saline, that floods permanently or temporally and sustains water-adapted plants and animals. According to Arthington (2003), an increase in water temperature will alter fundamental ecological processes and the geographic distribution of wetland plant species. This might also lead to the loss of species provided there are no migratory corridors available for the species to colonize. However, even though the migration
phenomenon is important because it might prevent species extinction, there are few localized studies on modeling potential distribution of wetland species especially in Australia. There are gaps in our understanding as to what areas are suitable for migration by most wetland species as a result of temperature increase.

In addition to changing the environmental parameters affecting species distributions, increasing global surface temperature will also cause the mean sea level to rise, changing the physical range available to some coastal wetland types. The current and anticipated rise in sea level is mainly due to thermal expansion and the loss of land-based ice due to increased melting. According to the IPCC fourth assessment report, the estimates for the 20th century showed that the global sea level rose at an average rate of about 1.7 m m y⁻¹ (IPCC, 2007a). Furthermore, Satellite altimetry data shows that since 1993, sea level has been rising at a rate of approximately 3 mm yr⁻¹ significantly higher than the average during the past half century (IPCC, 2007a). One projected estimate of the rising sea level in the 21st century is about 4 m m y⁻¹ under the IPCC Special Report on Emission Scenarios (SRES) A1B scenario by the mid-2090s (IPCC, 2007a). According to the modeling carried out by McFadden et al. (2007) on coastal wetlands (excluding sea grasses), there will be a global loss of 33% and 44% given a 36cm and 72cm sea level rise respectively in the period of 2000 to 2080. Regionally, there would be a severe loss of coastal wetlands in the Atlantic and Gulf of Mexico coasts of North and Central America, the Caribbean, the Mediterranean, the Baltic and most small oceanic islands due to their low elevation (Nicholls, 2004). Nevertheless, there are still gaps in our understanding of the potential impacts of sea level on coastal wetlands especially at local
scale i.e. what wetland types are most vulnerable to sea level rise and what mitigation and adaptation measures are there to curb the possibly devastating impacts of sea level rise on the wetland ecosystems at local and regional levels?

1.2. THE GEOINFORMATICS APPROACH TO CLIMATE CHANGE IMPACT ASSESSMENT

A range of approaches have been deployed in assessing the potential impacts of climate change on coastal wetlands including the use of Geoinformatics tools. This involves the application of geographic information science (GIS) and remote sensing techniques in assessing the potential impacts of climate change. GIS is a computer system comprised of hardware, software, and data which allows the user to layer different types of information together provided the data have a common geographic location (Badurak, 2000).

GIS originated from thematic cartography and in 1959, Waldo Tobler published a paper in Geographical Review which explained a simple model for applying the computer to cartography. This was called the map in-map out (MIMO) system which had three elements i.e. a map input, map manipulation and a map output stage (Tobler, 1959). This formed the origin of geocoding, data capture, data management and analysis, data display modules which later produced contemporary GIS packages. Within a short duration from then, the write-up of computer programs using programming languages like FORTRAN to draw maps became eminent. Early computer mapping packages such as SURFACE II,
IMGRID, CALFORM, CAM, and SYMAP were also developed (Clarke, 2003). GIS became well-established as a technology in the 1980s and 1990s. In 1982, IBM introduced its PC (personal computer) and within a few years, some large and desktop GIS packages such as Arc/Info and IDRISI made a transition to the microcomputer. In the 1990s, GIS stretched far beyond its origin in mapping science to encompass disciplines such as geology, archeology, epidemiology, and criminal justice. In addition, it became fully integrated with global positioning system (GPS). High-resolution imagery became common as a reference base for GIS data and finally the emergence of the internet and e-commerce has placed GIS onto the World Wide Web as Web-GIS (Clarke, 2003). GIS is a very important tool in impact assessment because it facilitates the analysis of multiple layers of data and allows statistical analysis of multiple factors while maintaining their spatial representation. It also facilitates the collection of future relevant data and complex analyses.

Remote sensing is the discipline of deriving information about the earth’s land and water surfaces using images acquired from an overhead perspective, by using electromagnetic radiation in one or more regions of the electromagnetic spectrum, reflected or emitted from the earth’s surface (Campbell, 2006). Remote sensing from space received its first impetus through remote sensing from rockets. From the period of 1946-1950, small cameras were carried aboard captured V-2 rockets that were fired from the White Sands Proving Ground in New Mexico. From then, numerous flights involving photography were made by rockets, ballistic missiles, satellites and manned spacecraft. However the photographs produced were of low quality (Lillesand & Kiefer, 2000). In
the 1960s, remote sensing became apparent to the civilian community from the Mercury, Gemini and Apollo space programs. By the end of the programs, the value of remote sensing from space became prominent. Multispectral orbital photography for earth resource studies was also obtained from Apollo earth orbit flights (Apollo 9) made prior to lunar landings. The photographs were produced using panchromatic film with green and red filters, black and white IR film, and colour IR film (Lillesand & Kiefer, 2000). In the past few decades, remote sensing from space has developed to a great extent with the launch of numerous satellites platforms and sensors including Landsat, SPOT (Système Pour l’Observation de la Terre), Terra and Radarsat. Landsat satellites were developed in the 1960s and have been in function since 1972. Landsat-1 was launched on July 23rd, 1972. A similar satellite Landsat-2 was placed into orbit in January 1975. Landsat -3, -4, and -5 followed in 1978, 1982, and 1984 respectively. Landsat -6 was lost on launch in October 1993 and Landsat 7 was placed into orbit on April 15th, 1999. The following sensors were flown by Landsat satellites: The Return Beam Vidicon (RBV) Camera, Multi-spectral Scanner (MSS), Thematic Mapper (TM), and Enhanced Thematic Mapper Plus (ETM +) (Mather, 1999). SPOT has been in operation since 1986. Each of the satellites carries two identical HRV (High-Resolution Visible) sensors. They use a linear array of charge-coupled devices or CCDs, so that all the pixels in an entire scan line are imaged at the same time. SPOT -1 was launched on February 22nd, 1986, SPOT-2 in January 1990 and SPOT-3 followed in September 1993. Due to technical error, SPOT-3 was lost. SPOT- 4 was successfully launched on March 24th, 1998. SPOT-4 carries a new sensor called VEGETATION with resolution at 1.15 km at nadir, and swath width of 2,250 km (Mather, 1999). Terra was launched as a result of a joint effort between NASA and Japan's Ministry of Economy, Trade and Industry (METI) and the Earth Remote Sensing Data
Analysis Center (ERSDAC). It carries five remote sensing instruments including the Advanced Spaceborne Thermal Emission and Reflectance Radiometer (ASTER). ASTER is designed to acquire land surface temperature, emissivity, reflectance and elevation (Mather, 1999). It consists of three subsystems namely: Visible and Near-Infrared (VNIR), Shortwave Infrared (SWIR) and Thermal Infrared (TIR). Radarsat was launched on November 4th, 1995 and is powered by solar arrays. This is because it is in a sun-synchronous orbit at an altitude of 798 km and an inclination of 98.6° to the equatorial plane. It carries the synthetic aperture radar (SAR) sensor and operates in the C band with HH polarization (Van der Meer et al., 2002).

Satellite data from these remote sensing platforms and sensors could be used to assess the potential impacts of climate change. According to Intergovernmental Panel of Climate Change (2007a) fourth assessment report, warming of about 3°C is projected in the equatorial and coastal areas in Africa while a rising temperature of more than 4°C is projected in the Western Sahara by the end of the century. In Europe, under the A1B scenario, the simulated annual mean warming from 1980 to 2080 and 1999 to 2099 varies from 2.3 °C to 5.3 °C in northern Europe and from 2.2 °C to 5.1 °C in southern Europe. A similar range of 1.8 °C to 5.0 °C has also been projected in Central America. In Asia, warming of about 3.3 is projected for South Asia and East Asia while central Asia, Tibet and Northern Asia have projections of 3.7 °C, 3.8 °C and 4.3 °C respectively by the end of the century. By the year 2070 in Australia, the mean annual temperature is projected to increase by 6 °C over most of Australia with slightly less warming in some coastal areas and Tasmania. The projected climate change and possible impacts can be geographically modeled and mapped easily using satellite data and geographic information system.
Nonetheless, most of these satellites’ data have limitations due to the factor of scale. Scale represents the spatial resolution of a sensor and this is the ability of a sensor to record and display fine detail as separated by its surroundings (Woodcock & Strahler, 1987). According to Woodcock and Strahler (1987), the choice of an appropriate scale in remote sensing is determined by the output information desired about the ground scene, the method of analysis to be used to extract the information from the satellite image and the spatial structure of the scene itself.

The Geoinformatics approach in assessing the potential impacts of climate change on coastal wetlands is advantageous relative to other approaches such as biophysical, quantitative and economic modeling because it is easier to depict past, present or future climate impact patterns. Furthermore, it uses simple indices to evaluate present-day regional potential for different activities based on climate and other environmental factors. It is also less time consuming and more effective in mapping changes induced in the patterns of wetlands by a given change in climate, thus showing the extent and rate of shifts. In addition, it is easier to identify regions that may be vulnerable to climate change and it is possible to compare impacts on different activities in the same geographic region. Finally, it can easily be used in conjunction with other general circulation models, biophysical simulation models, and integrated databases to conduct regional and global impact analysis (Burton et al., 1998; João, 1998).

Even though, it is easier to map wetlands patterns and changes using Geoinformatics tools, the availability of detailed wetland maps using GIS and remote
sensing is still not available in most parts of the world especially in Australia at local levels. There are also gaps in our understanding of the potential impacts of climate change on most coastal wetlands within local and regional levels in the continent using Geoinformatics approach.

1.3. RESEARCH OBJECTIVES AND RATIONALE OF THE STUDY

The fact that coastal wetlands are very sensitive to climatic factors and the potential impacts and mechanisms are not well understood in order to provide remediation justifies the need for research on climate change impacts on coastal wetlands. Also, due to the fact that there is no known research on the possible impacts of climate change on the coastal wetlands in the subtropical region of north-eastern NSW with high biodiversity (Morand, 1994, 1996; 2001) justifies the need for research on these wetland ecosystems. Furthermore, there are still gaps in our understanding on the potential impacts of climate change using geoinformatics approach at both local and national level. The research could provide useful information on the potential impacts, mitigation and adaptation strategies that could be used in management and planning by local authorities in order to curb the devastating impacts of climate change.
The aim of this study is to assess the potential impacts of climate change on the coastal wetlands in north-eastern NSW, Australia. The objectives are:

1. To provide an overview of wetland classifications in relation to their plant communities and their environmental variables with emphasis in NSW (chapter two).

2. To map the current and past wetland communities in north-eastern NSW, in order to identify any changes in quality and extent (chapter four).

3. To predict the potential spatial distribution of selected wetland species as a result of climate change i.e. mean annual temperature increase (chapter five).

4. To predict the potential impact of sea level rise on the coastal wetland communities by the end of the century (chapter six).

5. To estimate the amount of methane emission from the coastal wetlands using satellite data and their emission with a temperature increase (chapter seven).

6. To provide management, mitigation and adaptation strategies that could be used to minimize the impacts of climate change on the coastal wetland ecosystems (chapter eight).
1.4. ORGANIZATION OF THE THESIS

Chapter one presents the current understanding of climate change and sea level rise. It describes the science, history and projections of climate change and sea level rise and their possible impacts to wetlands. Furthermore, it provides a brief history of the development of Geoinformatics tools and their usefulness in climate change impact assessments. In addition, the research gap i.e. the need to assess the potential impacts of climate change on the coastal wetlands in north-eastern NSW is identified. The aim and specific objectives of the study are also presented. Chapter two reviews some of the wetland classifications with emphasis to NSW wetland classification. Chapter three describes the study area of the research such as the location, climate and geology. Chapter four provides discussion on monitoring the coastal wetland communities through mapping the current and past wetland communities in order to identify any change in the quality and extent. Chapter five discusses the results on the potential spatial distribution for the selected wetland species i.e. *Avicennia marina* (grey mangrove), *Banksia integrifolia* (Coastal Banksia), *Melaleuca quinquenervia* (broad-leaved paperback) and *Leptospermum liversidgei* (Swamp May) as a result of climate change i.e. temperature increase. Chapter six discusses the potential impacts of sea level rise on the coastal wetland communities. This chapter provides predictions of wetlands that are most vulnerable to sea level rise. Chapter seven discusses the results of methane emission estimates from the coastal wetlands using satellite data and their emission with temperature increase. Chapter eight discusses some of the possible management, mitigation and adaptation strategies that could be used to minimize the impacts of climate
change on the coastal wetland communities. The overall conclusions of the research are given in chapter eight.
CHAPTER TWO

2.0. A REVIEW OF WETLAND CLASSIFICATIONS

This chapter provides a scientific literature of wetland classifications (past and present) at both international and national levels. The importance of wetlands and the different classification designs are also discussed. Furthermore, it describes wetland types in relation to their plant communities and environmental variables including hydrology with an emphasis on NSW wetland classification.

Wetlands are increasingly recognized for their numerous values such as being a source of primary productivity, providing habitats for wildlife, enhancing water quality and providing opportunities for recreation (Mortsch, 1998). Several wetland classification systems have been developed to describe wetland types, but due to the high variability in wetland conditions, most are limited in applicability to particular geographic regions or particular wetland types (Tiner, 1999).

There are two fundamental designs for wetland classification system i.e. horizontal and hierarchical. In horizontal classification systems, habitats are divided into a series of classes or types. In contrast, hierarchical systems provide a matrix for separating wetlands into a multitude of types with different levels defined (Tiner, 1999). The higher levels have generalized characteristics such as landscape and water source while the lower levels are based on specific and detailed characteristics such as
vegetation life form, substrate characteristics and water level fluctuations (Tiner, 1999). Classification schemes are important because they have been used to identify habitats with common features for inventory, monitoring and establishment of representative reserve systems. Furthermore, they have been used to establish appropriate control sites for scientific surveys and for predicting common responses to environmental change. Wetland classifications have also been useful in mapping, planning, acquisition of biophysical information and regulatory purposes.

In the past, wetlands were traditionally characterised by their vegetation types which consisted grasses, herbs, shrubs and trees. In Europe, two main types of wetland plant communities were identified namely: saline swamps (with halophytic vegetation) and fresh-water swamps (Warming, 1909). Fresh-water swamps consist of reed-swamp, bush-swamp and forest-swamp. Reed-swamp is dominated by herbaceous plants, mainly tall perennial monocots such as *Phalaris arundinacea*, and *Phragmites australis*, while bush-swamp and forest-swamp are characterized by shrub and trees respectively. In 1910, Shreve *et al.* (1910) characterized a number of wetland types based on vegetation, life form, and soil type and/or landscape position from an early description of the vegetation of Maryland, North America. They considered forested wetlands to include: clay upland swamps, sandy loam upland swamps, flood plains, river swamps and stream swamps. Herbaceous dominated wetlands were either salt marshes or fresh marshes. In contrast, Chrysler (1910), also working in North America emphasized the importance of certain plant associations or zones within wetland types such as the gum-pine association of lowland forest, the *Nymphaea, Pontederia, Zizania, Typha*, alder and maple associations
of fresh marshes and different *spartina* zones for salt marshes. However, this was limited as it did not include environmental variables such as soil type and hydrology.

A comprehensive wetland classification system for North America was developed by Cowardin *et al.* (1979). This was used by the US government with the main objective for wetland inventory and management. This classification system is organized in a hierarchical format with the inclusion of ‘deep water habitats’ which had not been included in past traditional classification systems (Cowardin *et al.*, 1979). Wetlands and deep water habitats were sub-divided into Systems, Sub-systems and Classes (Figure 2.1).
Figure 2.1: Classification hierarchy of wetlands and deepwater habitats, showing Systems, Subsystems and Classes— the Palustrine system does not include deepwater habitats Source – (Cowardin et al., 1979).
A ‘system’ is a complex of wetlands and deep water habitats that share the influence of similar hydrologic, geomorphologic, chemical, or biological factors (Tiner, 1999). There are five systems in this classification: Marine, Estuarine, Riverine, Lacustrine and Palustrine.

2.1. RAMSAR CLASSIFICATION OF WETLAND TYPE

Although Cowardin et al.’s (1979) classification system was assumed to be comprehensive; it did not include wetlands that are created from human activities such as ponds and canals. A holistic approach to a multinational wetland classification system was therefore developed during the Convention of Wetlands (Ramsar Convention) in Iran. The classification system was adopted in 1990 and modified in 1996 and is called the Ramsar Classification of Wetland Type. This system recognizes, eleven types of marine and coastal wetlands, nineteen inland wetland types and nine human- made wetland types (R.C.B., 2006).

I- Coastal/Marine wetlands.

A – Permanent shallow marine waters in most cases less than six metres deep at low tide; includes sea bays and straits.
B – Marine subtidal aquatic beds; includes kelp beds, sea-grass beds, and tropical marine meadows.
C – Coral reefs.
D – Rocky marine shores; includes rocky offshore islands, sea cliffs.
E – Sand, shingle or pebble shores; includes sand bars, spits and sandy islets; includes dune systems and humid dune slacks.

F – Estuarine waters; permanent water of estuaries and estuarine systems of deltas.

G – Intertidal mud, sand or salt flats.

H – Intertidal marshes; includes salt marshes, salt meadows, saltings, raised salt marshes; includes tidal brackish and freshwater marshes.

I – Intertidal forested wetlands; includes mangrove swamps, nipah swamps and tidal freshwater swamp forests.

J – Coastal brackish/saline lagoons; brackish to saline lagoons with at least one relatively narrow connection to the sea.

K – Coastal freshwater lagoons; includes freshwater delta lagoons.

Zk(a) – Karst and other subterranean hydrological systems, marine/coastal

II - Inland wetlands

L – Permanent inland deltas.

M – Permanent rivers/streams/creeks; includes waterfalls.

N – Seasonal/intermittent/irregular rivers/streams/creeks.

O – Permanent freshwater lakes (over 8 ha); includes large oxbow lakes.

P – Seasonal/intermittent freshwater lakes (over 8 ha); includes floodplain lakes.

Q – Permanent saline/brackish/alkaline lakes.

R – Seasonal/intermittent saline/brackish/alkaline lakes and flats.

Sp – Permanent saline/brackish/alkaline marshes/pools.

Ss – Seasonal/intermittent saline/brackish/alkaline marshes/pools.
Tp – Permanent freshwater marshes/pools; ponds (below 8 ha), marshes and swamps on inorganic soils; with emergent vegetation water-logged for at least most of the growing season.

Ts – Seasonal/intermittent freshwater marshes/pools on inorganic soils; includes sloughs, potholes, seasonally flooded meadows, sedge marshes.

U – Non-forested peatlands; includes shrub or open bogs, swamps, fens.

Va – Alpine wetlands; includes alpine meadows, temporary waters from snowmelt.

Vt – Tundra wetlands; includes tundra pools, temporary waters from snowmelt.

W – Shrub-dominated wetlands; shrub swamps, shrub-dominated freshwater marshes, shrub carr, alder thicket on inorganic soils.

Xf – Freshwater, tree-dominated wetlands; includes freshwater swamp forests, seasonally flooded forests, wooded swamps on inorganic soils.

Xp – Forested peatlands; peatswamp forests.

Y – Freshwater springs; oases.

Zg – Geothermal wetlands

Zk(b) – Karst and other subterranean hydrological systems, inland

Note: "floodplain" is a broad term used to refer to one or more wetland types, which may include examples from the R, Ss, Ts, W, Xf, Xp, or other wetland types. Some examples of floodplain wetlands are seasonally inundated grassland (including natural wet meadows), shrublands, woodlands and forests. Floodplain wetlands are not listed as a specific wetland type herein.
III - Human-made wetlands

1 – Aquaculture (e.g., fish/shrimp) ponds

2 – Ponds; includes farm ponds, stock ponds, small tanks; (generally below 8 ha).

3 – Irrigated land; includes irrigation channels and rice fields:

4 – Seasonally flooded agricultural land (including intensively managed or grazed wet meadow or pasture).

5 – Salt exploitation sites; salt pans, salines, etc.

6 – Water storage areas; reservoirs/barrages/dams/impoundments (generally over 8 ha).

7 – Excavations; gravel/brick/clay pits; borrow pits, mining pools.

8 – Wastewater treatment areas; sewage farms, settling ponds, oxidation basins, etc.

9 – Canals and drainage channels, ditches.

Zk(c) – Karst and other subterranean hydrological systems, human-made

Source: (R.C.B., 2006).

This multinational classification system provides the basis for independent classification systems more applicable to the individual countries. In Australia, the national wetland classification system is based on the Ramsar convention classification system but modified slightly to suit the Australian situation (EA, 2001). The following wetland types are not included in the Ramsar classification system but have been added to the Australian system: non-tidal freshwater forested wetlands (A12) and rock pools (B17) (EA, 2001).
2.2. A DIRECTORY OF IMPORTANT WETLANDS IN AUSTRALIA (DIWA), WETLAND CLASSIFICATIONS, INCLUDING MODIFICATIONS

A - Marine and Coastal Zone wetlands

1. Marine waters; permanent shallow waters less than six metres deep at low tide; includes sea bays, and straits.
2. Subtidal aquatic beds; includes kelp beds, seagrasses, tropical marine meadows
2. (a) Algal beds
3. Coral reefs
4. Rocky marine shores; includes rocky offshore islands, sea cliffs, intertidal rock platforms
5. Sand, shingle or pebble beaches; includes sand bars, spits, sandy islets
6. Estuarine waters; permanent waters of estuaries and estuarine systems of deltas
7. Tidal mud, sand or salt flats; intertidal or supratidal
8. Tidal marshes; saltmarshes, salt meadows, saltings, brackish and freshwater marshes
9. Tidal forested wetlands; includes mangrove swamps, nipa/palm swamps, freshwater swamp forests
10. Brackish to saline lagoons and marshes with one or more relatively narrow connections with the sea
11. Freshwater lagoons and marshes in the coastal zone
11. (a) Freshwater reed / rush swamps in the coastal zone
12. Non-tidal freshwater forested wetlands, permanently or temporarily flooded (Swamp Forests)

12. (a) Wet heath

13. Karst or subterranean wetlands with a connection to the marine environment, includes anchialine systems.

**B - Inland wetlands**

1. Permanent rivers and streams; includes waterfalls, permanent waterholes in river reaches
2. Seasonal and irregular rivers and streams; includes minor anabranches, braided channel complexes
3. Inland deltas (permanent and temporary)
4. Riverine floodplains; includes temporarily flooded river flats, river basins, grassland and palm savanna
5. Permanent freshwater lakes (> 8 ha); includes large oxbow lakes
6. Seasonal/intermittent freshwater lakes (> 8 ha), floodplain lakes, billabongs, claypans
7. Permanent saline/brackish lakes
8. Seasonal/intermittent saline lakes
9. Permanent freshwater ponds (< 8 ha), marshes and swamps on inorganic soils; with emergent vegetation
10. Seasonal/intermittent freshwater ponds and marshes on inorganic soils; includes claypan complexes,
11. Permanent saline/brackish marshes

12. Seasonal saline marshes

13. Freshwater shrub swamps; shrub-dominated marsh on inorganic soils, includes lignum, ti-tree swamps

14. (a) Swamp forest, not in coastal zone

14. (b) Riparian vegetation and wet schlerophyll forest, not in coastal zone, above 10m contour

15. Peatlands; forest, shrub or open bogs

16. Alpine wetlands; includes alpine meadows and pools, temporary waters from snow melt

17. Freshwater springs, oases and rock pools; includes gnamma holes, mineralised mound and artesian springs

18. Geothermal wetlands

19. Inland, subterranean karst wetlands

C - Human-made wetlands

1. Water storage areas; reservoirs, barrages, hydro-electric dams, impoundments (generally > 8 ha)

2. Ponds, including farm ponds, stock ponds, small tanks (generally < 8 ha)

3. Aquaculture ponds; fish ponds, shrimp ponds

4. Salt exploitation; salt pans, salines

5. Excavations; gravel pits, borrow pits, mining pools
6. Wastewater treatment; sewage farms, settling ponds, oxidation basins

7. Irrigated land and irrigation channels, canals or ditches; includes rice fields

8. Seasonally flooded arable land, farm land

9. Canals, stormwater drains

10. Wetlands constructed for biodiversity benefit; includes for habitat creation, and water quality improvement or maintenance.

Source: (Burns, Cibilic, & Smith, 2006).

2.3. **WETLAND CLASSIFICATION IN NEW SOUTH WALES**

In addition to the national wetland classification scheme, the different states in Australia have developed their own wetland classifications. In the state of NSW, the main wetland classification used is that developed by Green (1997). The three main criteria considered during the development of the classification were: 

a) the results of the classification should lead to wetland types which are easily distinguished by people without a scientific background;

b) the classification must be broad, reflecting the broad level of guidelines to be included in the manual;

c) the classification must reflect wetland management issues so that wetlands with similar management problems are included in the same wetland type (Green, 1997). The resultant wetland types from the classification (Table 2.1) are based on geographic locations i.e. coastal, tableland and inland. They are further divided within these geographic locations based on hydrology, geomorphology and vegetation characteristics (Green, 1997). The wetland types are specific to
geographic location except the upland lakes and lagoon, and upland swamps that may occur in all three geographic locations even though are mostly found on tablelands (Green, 1997).

**Table 2.1: Wetland types in New South Wales**

<table>
<thead>
<tr>
<th>Coastal</th>
<th>Tableland</th>
<th>Inland</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mangroves and saltmarsh swamps</td>
<td>Upland lakes and lagoons</td>
<td>Permanent inland wetlands</td>
</tr>
<tr>
<td>Estuarine lakes and lagoons</td>
<td>Upland swamps</td>
<td>Inland floodplain lakes and lagoons</td>
</tr>
<tr>
<td>Dune swamps and lagoons</td>
<td></td>
<td>Inland floodplain meadows</td>
</tr>
<tr>
<td>Coastal floodplain swamps and lagoons</td>
<td></td>
<td>Reed swamps</td>
</tr>
<tr>
<td>Coastal floodplain forest</td>
<td></td>
<td>Lignum swamps</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Inland floodplain forests and woodlands</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Arid wetlands</td>
</tr>
</tbody>
</table>

Source: Green (1997)

**I - Coastal wetlands**

**Mangrove and saltmarsh swamps:** Mangrove and saltmarsh swamps are located in estuarine areas which are subjected to tidal flooding and support mangrove and saltmarsh vegetation (Green, 1997). Mangroves can occur on sand, rock, mud or on corals but are well established on deltaic silts at the mouths of large river systems (Saenger, 1995). Non-tidal basins found on estuarine sediments adjacent to mangrove and saltmarshes areas, as well as mudflats and small creeks present within or adjacent to these
communities are also included in this wetland type. The mangroves and saltmarsh communities are located along tidal shorelines and usually extend up coastal rivers as far as the tidal limit. They are exposed to seawater and the saltmarsh communities always occur on the landward side of the mangroves. The main source of water in this wetland type is estuarine and both the mangroves and saltmarshes are dependent on periodic tidal inundation. In New South Wales, mangroves dominate those areas inundated daily, whereas saltmarsh vegetation occurs in areas less frequently flooded. The dominant mangrove species found along the entire coastline of NSW is *Avicennia marina* (grey mangrove); *Aegiceras* is found in most central and northern NSW estuaries, extending further upstream and higher on the shore than *Avicennia*. Three other species occur in isolated patches in some northern NSW estuaries. Saltmarshes are dominated by plant species in the family Chenopodiaceae including genera such as *Atriplex, Bassia, Enchylaena, Halosarcia, Sarcocornia, Tecticornia* and *Hemichroa* (Adam, 2009; Saenger, Specht, Specht, & Chapman, 1977). Saltmarshes often have distinct zonation of vegetation relative to frequency of inundation from the sea (Green, 1997). According to Goodrick (1983), a typical zonation consists of *Salsicornia spp.* (samphire), *Sporobolus virginicus* (salt couch) and *Juncus maritimus* (salt rush) gradually from sea to land.

**Estuarine lakes and lagoons**: Estuarine lakes and lagoons are open saline or brackish water bodies which have a relatively narrow permanent or intermittent connection to the sea (Green, 1997). Lakes are different from lagoons mainly by size, usually lakes are largest. In NSW, saline lakes and lagoons occur along the coast with very high occurrence on the south coast. They are separated from ocean or estuary by a barrier of
sand dune, and usually have a connection that is usually intermittent with the ocean or estuary. They are opened to the sea during floods or at high tides or artificially e.g. dredging in response to pollution. There is always fluctuation in the water level because of irregular connection with the sea (Green, 1997). *Zostera capricorni* and *Zostera muelleri* are found in the north and south respectively of lagoons and lakes that are frequently open to sea. Less saline lagoons support *Ruppia* spp. (sea tassel) which is often found with *Lamprothamnium* spp. (stonewort). In addition, algae is usually present and the shallow edges support emergent plants like common reeds and sedges e.g. *Bulboschoenus* spp. *Schoenoplectus* spp. or *Baumea* spp. (Jacobs, 1983).

**Dune swamps and lagoons:** Dunal swamps and lagoons are freshwater wetlands that occur on coastal sand dunes or plains and include lakes, lagoons, shallow vegetated basins, heaths and forest (Green, 1997). They are located on prior dune systems which occur behind the present beach and foredune. They are found along the coast of NSW with highest occurrence on the north coast between Myall lakes and Queensland border. Their main source of water is from ground water, rainfall or runoff from local catchments. The water in dunal lagoons and swamps are acidic due to dissolved organic matter from peaty soils (Green, 1997). According to Timms (1988) and Winning (1992), dune swamps and lagoons may have different hydrological characteristics due to their variability in occurrence e.g.:

- Perched wetlands that are found on top of dunes on impermeable sand layer above regional groundwater table are reliant on rainfall and runoff as their source of water. This
is because they are formed by the accumulation of organic matter in the dune depression and this makes them impermeable to ground water.

- Watertable-window wetlands are reliant on groundwater as their main water source because they are formed in the dune swales which lie below the regional groundwater thus form a ‘window’ to the water table.
- Upland contact wetlands depend on both runoff and groundwater as their water source because they occur between a sand dune and adjacent bedrock.
- Frontal dunes would depend on rainfall as their water source because they occur in small wind created hollows in frontal dunes.

Dune swamps and lagoons have distinctive vegetation with forest of *Melaleuca quinquenervia* (broad-leaved paperbarks) found on seasonally flooded areas. Other vegetation includes: sedges, rushes and wet heathland.

**Coastal Floodplain Swamps and Lagoons:** These are wetlands that occur on the floodplain of a coastal river e.g. shallow marshes, meadows vegetated by sedges and aquatic herbs, deeper ponds and billabongs with large areas of open water (Green, 1997). They are associated with major rivers, and may be found at several different positions on floodplains such as; the area where the floodplain slopes away from the river and meets an adjacent terrain or as a ponded tributary where the river levee has dammed the junction of a small tributary (Pressey, 1986; Winning, 1992). The main source of water to this wetland type is from seasonal or intermittent flooding from river. Coastal floodplain swamps and lagoons have dynamic vegetation structure and can change with seasonal variation in water depth (Winning & King, 1995). Plant species such as jointed twig-rush
(Lepidosperma spp), spikerushes (Eleocharis spp.), cumbungi (Typha spp.), water ribon (Triglochin procera) and frogsmouth (Philydrum lanuginosum) are found along the margins (Jacobs, 1983). Lagoons may support submerged plant species like ribbonweed (Vallisneria gigantean), bladderwort (Utricularia spp.), watermilfoil (Myriophyllum spp.), and pondweed (Potamogeton spp.). Coastal floodplains swamps and lagoons support flooding aquatic species such as water primrose (Ludwigia peploides), swamps lily (Otellia ovalifolia), waterlilies (Nymphoides spp. and Nymphaea spp.), duckweed (Lemna spp., Wolfia spp. and Spirodella spp) and Azolla spp. (Jacobs, 1983). However, the floodplain wetlands in north coast of NSW consist mostly of jointed twig-rush even tough they are typically associated with dunal swamps (Winning & King, 1995).

Coastal floodplain forests: Coastal floodplain forests are wetlands that usually occur on sandy sediments on the floodplain of coastal rivers and dominated by trees. These include the paperbark forests and woodlands around the coast (Green, 1997). Seasonal flooding from rivers is the main source of water for this wetland type and many large rivers in the north coast of NSW flood annually especially in the summer and autumn due to cyclonic depression (Green, 1997). The vegetation is dominated by broad-leaved paperbark (Melaleuca quinquenervia) even though narrow-leaved paperbark (Melaleuca linarifolia) and northern narrow-leaved (Melaleuca alternifolia) may occur along the margin or as scattered trees within this forest (Winning & King, 1995). Species typical to the herb layer include: rushes (Juncus spp.), common reed (Phragmites australis), water ribbon (Triglochin procera), Maundia triglochinoides and Persicaria strigosa. The dominant grasses include: matgrass (Hemarthria uncinata), spiney mudgrass (Pseudraphis
spinescens), swamp rice grass (*Leersia hexandra*) and watercouch (*Paspalum paspalodes*) (Jacobs, 1983; Winning & King, 1995).

**II - Tableland wetlands**

**Upland lakes and lagoons:** Upland Lakes and lagoons are large or small bodies of freshwater located in low hills or mountains. They are often open water with sometimes marginal vegetation (Green, 1997). They occur mostly on tablelands but also include any open water body on the coastal or inland plains (excluding arid wetlands) which water source maybe runoff from local catchments, groundwater or rainfall (Green, 1997). Upland lakes and lagoons are found in depressions that are linked to past fault activity, erosion and depressions weathered by past glacial activities (Winning, 1992). Large lakes do not support much aquatic vegetation even though sometimes there might be submerged plants like sea tassel (*Ruppia* spp.), ribbon weed (*Vallisneria gigantean*), *Lepilaena* spp. and water milfoils (*Myriophyllum* spp.). Upland lakes and lagoons usually support sparsely marginal vegetation but often restricted to common reed (*Phragmites australis*), cumbungi (*Typha* spp.), sedges and rushes (Jacobs and Brock, 1993).

**Upland swamps:** These are wetlands that consist of vegetation and are mostly found in shallow basins located in low hills or mountains. They include: shallow marshes, sedge swamps, ‘hanging’ swamps, wet heath and peaty swamps (Green, 1997). Their occurrence is mainly on or adjacent to the tablelands but also include any vegetated wetland on the coastal or inland plains excluding arid wetlands (Green, 1997). Within
the state of NSW, ‘hanging swamps’ of sedge and heath are found on valley sides on the sandstone plateau around Sydney and the granite plateau of New England. These are areas with ground water discharge due to the impermeability of the bedrock. Rainfall, groundwater, and runoff from local catchments constitute the main source of water for upland swamps (Green, 1997). Upland swamps consist mostly of vegetation and generally includes grasses, sedges, and shrubs. Sedge swamps are dominated by species in the family Cyperaceae such as razor sedge (*Lepidosperma limicola*), button bog-rush (*Gymnoschoenus spaerocephalus*), fringed cord-rush (*Restio fimbriatus*) and slender yellow-eye (*Xyris gracilis*). Shrubs species include: teatrees (*Leptospermum* spp.), bottlebrush (*Callistemon* spp.), hakea (*Hakea* spp.), banksia (*Banksia* spp.) and melaleuca (*Melaleuca* spp) (Keith & Myerscough, 1993; Winning & King, 1995).

**III - Inland wetlands**

**Permanent inland wetlands:** Permanent inland wetlands are mostly open water wetlands that are filled from a river under regulated flow conditions or which are permanently impounded by a structure and include lakes, billabongs and impounded channels (Green, 1997). They occur in different situations such as wetlands within the weir-pool created by a major structure on the river. They may also occur as lakes and basins that receive water that is diverted or pumped for storage or as wetlands that have poor connection with the river and are therefore filled under regulated flow conditions. Stream flow is the main source of water for permanent inland wetlands although some wetlands have tailwater as their main source. Permanent inland wetlands are mostly open water bodies with less
vegetation structure. Most often there are dead trees found in the water or around the wetlands which is an indication of low ecological productivity due to a change in the natural water regime. However, common reed (*Phragmites australis*) is usually found around the shoreline while cumbungi (*Typha spp.*) is often present in water of up to 1.5m deep around the margins of the wetland (Green, 1997).

Reed Swamps: Reed Swamps are wetlands with less open water and are located on the floodplain of a river which is subject to permanent or intermittent river flows and are dominated by reeds (Green, 1997). Examples in New South Wales include the Macquarie marshes at the end of the Macquarie River and the Great Cumbung Swamp at the end of the Lachlan River. The main source of water for reed swamps is the surplus or regulated flows from a river. The vegetation in reed swamp wetlands include dense stands of common reed (*Phragmites australis*) and cumbungi (*Typha spp.*). The cumbungi occupies the wetter part of the wetlands whereas the common reed occurs on areas exposed more often by low water levels. Common reeds may also occur in small wetlands with other species like spikerush (*Eleocharis spp*) and other rushes (Green, 1997).

Inland floodplain lakes and lagoons: These are wetlands located on the floodplain of a river and consist principally of open water that is subject to a cycle of flooding and drying (Green, 1997). The location of intermittent inland floodplain lakes and lagoons is dependent to their geomorphic origin (Pressey, 1986; Winning, 1992). For example they may be located on depressions filled by overbank flooding or a cut-off channel formed by
occlusion of an old river channel (Green, 1997). Seasonal or intermittent flooding from a river constitutes the main water source for inland floodplain lakes and lagoons. The vegetation of the floodplain lakes and lagoons are restricted along the margins due to their large area of open water. However in large intermittent lakes, lignum (*Meuhlenbeckia florulenta*) is common around the edges while oxbow lagoons usually support river red gums (*Eucalyptus camaldulensis*) along their banks. The sedges that are common to intermittent lakes and lagoons include: spikerushes (*Eleocharis* spp.), knotweeds (*Persicaria* spp.), nardoo (*Marsilea* spp.), water primrose (*Ludwigia peploides*), and aquatic grasses such as watercouch (*Paspalum paspalodes*) and spiny mudgrass (*Pseudaphis spinescens*) (Green, 1997).

**Lignum swamps:** Lignum swamps are inland wetlands that are dominated by lignum and mostly found on floodplains which are filled by surplus or flood flows (Green, 1997). Characteristically, they are situated at the end of a river system in extensive braided floodways or overflows associated with the river. However, they may also occur in depressions and billabongs adjacent to a river channel. The main source of water to this wetland type is the intermittent flooding from rivers. The prominent vegetation type is lignum (*Meuhlenbeckia florulenta*) and its abundance is high at areas where flooding is more frequent (Green, 1997). Other aquatic species most likely to be present in this wetland type include: spikerushes (*Eleocharis* spp.), water primrose (*Ludwigia peploides*), nardoo (*Marsilea* spp.), sedges (*Cyperus* spp.), knotweeds (*Persicaria* spp.) and buttercup (*Ranunculus* spp.) (Green, 1992; Jacobs and Brock, 1993).
**Inland Floodplain Forest and Woodlands:** These are inland floodplain wetlands that are dominated by trees and occur along major river systems (Green, 1997). They depend mostly on surplus flows or overbank flooding from a river as their main source of water. The understorey consists mostly of plant species that withstand inundation e.g. sedges, rushes, spikerushes and grasses such as watercouch (*Paspalum paspalodes*) and spiny mudgrass (*Pseudraphis spinescens*). However, some woodland types such as black box (*Eucalyptus largiflorens*) and coolibah (*Eucalyptus coolabah*) dominated woodlands consist of shrubby understorey with species like lignum (*Meuhlenbeckia florulenta*), nitre goosefoot (*Chenopodium nitriariceum*), saltbush (*Atriplex* spp.) and different herbs and grasses (Green, 1997).

**Inland Floodplain Meadows:** Inland Floodplain Meadows are shallow wetlands found on the floodplain of an inland river and are dominated by emergent species of grasses, sedges or rushes (Green, 1997). Shallow seasonal and intermittent flooding from a river constitutes the major source of water to this wetland type even though flood groundwater and rainfall may increase or help to maintain the water level. The grass species prominent to this wetland type include: spiny mud grass (*Pseudraphis spinescens*), barnyard grass (*Echinochloa colonum*), mat grass (*Hemarthria uncinata*) and water couch (*Paspalum paspalodes*). Rushes such as *Juncus* spp. and sedges *e.g.* *Carex* spp., *Cyperus* spp. occur in association with the grasses. Some common herb species include: nardoo (*Marsilea* spp.), knotweed (*Persicaria* spp.), buttercup (*Ranunculus* spp.), and watermilfoil (*Myriophyllum* spp.) (Green, 1997).
**Arid Wetlands:** They occur in arid areas which are not floodplains and gets in water principally from rainfall, groundwater or a local catchment e.g. salt lakes, salt pans, playa lakes and claypans (Green, 1997). According to Goodrick (1984), they may occur in different geomorphic situations such as: terminal playas (lakes) of major streams and receiving runoff via major and minor streams, or as claypans and playas of old drainage systems found on dunefields and sand plains, or as claypans receiving local runoff in dunefields and sandplains. This wetland type usually has very high rates of evaporation and the duration of inundation depends on inflows and seasonal evaporation rates (Goodrick, 1984). The water may be fresh, brackish or saline and there is no vegetation in large playas and saltlakes except algae and sea tassel which may grow on the more saline wetlands (Goodrick, 1984). Samphire (*Salicornia* spp.), saltbushes (*Atriplex* spp.), copperburs (*Sclerolaena* spp.) and bluebush (*Maireana appressa*) are predominant in the fringes of saltlakes and the beds of smaller saltpans (Green, 1992). Claypans are dominated by canegrass (*Eragrostic australis*) while in other wetlands, samphires and different species of grasses, herbs and copperburrs may be present (Goodrick, 1984; Green, 1992).

However, even though Green (1997) provided a concise wetland classification for the state of NSW, he did not include artificial wetlands in his classification. A holistic wetland classification based on origin, location, hydrology and vegetation is therefore suggested for this study (Table 2.2).
Table 2.2: Suggested wetland classification

<table>
<thead>
<tr>
<th>NATURAL</th>
<th>Coastal</th>
<th>Coastal swamps</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Mangroves and saltmarshes</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Dunal wetlands</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Forested wetlands</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Coastal upland water bodies</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Estuarine water bodies</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Coastal swamps</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Inland</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Permanent inland rivers/streams/creeks</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Permanent inland lakes and lagoons</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Inland floodplain wetlands</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Inland forested wetlands</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Inland swamps</td>
<td></td>
</tr>
<tr>
<td>ARTIFICIAL</td>
<td>Ponds (including farm ponds)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Irrigated land e.g. irrigation channels, rice fields</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Salt exploitation site e.g. salt pans</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Excavations e.g. mining pool, ground pits</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Waste water treatment areas e.g. sewage farms</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Canals and storm water drains</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Aquaculture ponds e.g. fish pond, shrimp ponds</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Seasonally flooded agricultural land</td>
<td></td>
</tr>
</tbody>
</table>

The suggested wetland classification in this study includes natural and artificial wetlands. The natural wetlands consist of both coastal and inland wetlands. The coastal wetlands included forested wetlands, coastal swamps, mangroves and saltmarshes, dunal wetlands, estuarine water bodies and coastal upland water bodies. The presence of artificial wetlands distinguishes this scheme from that of Green (1997). Furthermore, tableland wetlands (upland swamps and upland lakes and lagoons) are classified as coastal or inland wetlands in this classification because they are mostly found in coastal or inland areas in north-eastern NSW. The introduction of artificial wetlands in the classification is important because it could be useful in monitoring land cover/use change due to human activities.

There is generally a variety of classification schemes in the world including the Ramsar wetland classification, a directory of important wetlands in Australia and NSW.
wetland classifications. The origin, location, hydrology and vegetation of most wetlands, play an important role in their classification. Wetland classifications are mostly limited because the output classes are dependent on the criteria considered in the categorization. Furthermore, the variability in wetland types and conditions in the world makes most wetland classifications less comprehensive. Nevertheless, the multinational wetland classification system developed by the Convention of Wetlands (Ramsar Convention) provides a useful basis for the development of national and local classification schemes.
CHAPTER THREE

3.0. THE STUDY AREA

Chapter three describes the area of study with discussions on the climate, tidal regimes, soils, geology, land use and vegetation. This is because, in order to assess the potential impacts of climate change on localized wetland types, it is imperative to understand the regions' climate and other environmental and anthropogenic factors that would have an effect on the wetlands functioning, size and information generated from satellite data.

3.1. LOCATION

The study area extends from Evans Head to Tweed Heads within 20m elevation from the coastline of north-eastern New South Wales (Figure 3.1). This is because the coast is defined in this study as all areas less than 20m elevation from the coastline. It is a low land area bounded by latitudes 28°09'S and 29°06'S, and longitude 153°00'E.
Figure 3.1: Study Area from Tweed Heads to Evans Head, north-eastern NSW Australia
Major urban centres within the study area include: Kingscliff, Murwillumbah, Pottsville, Ballina, Broadwater, Woodburn and Byron Bay. The coastal region consists of different wetland types such as mangroves and saltmarshes, swamps, estuarine water bodies and forested wetlands (Plates 1-4).

Plate 1: Intertidal mangroves and saltmarshes in north-eastern NSW with species such as *Avicennia marina*

Plate 2: Coastal swamps in north-eastern NSW dominated by stands of wet heath
Plate 3: A typical brackish/saline estuarine water body in north-eastern NSW

Plate 4: A view of typical forested wetlands consisting predominantly of tree species such as *Melaleuca quinquenervia*
3.2. CLIMATE AND TIDE

The region is subtropical with a pronounced summer and autumn ‘wet’ season and drier winters and springs. It is one of the wettest areas in the state of NSW with high erosive rainfall and mild winters. Rainfall is very high and reliable (Edwards, 1979). February-March is generally the wettest period receiving about 30% of the average annual rainfall. The period of August to September is the driest with 3% if the average annual rainfall (Morand, 1996). Cyclonic depressions off the coast of Queensland are responsible for some of the most erosive rainfall in NSW. The percentage of erosive rainfall in January is about 14%, February 23% and March 12% (Rosewell & Turner, 1992). In Murwillumbah (Bray park) the month of February has the highest mean rainfall of around 236mm while September has the lowest mean rainfall of about 41.8mm in the period of 1972-2010 (Figure 3.2).
Figure 3.2: Monthly mean rainfall (mm) for Murwillumbah (1972-2010)

The mean monthly rainfall and temperature ranges for Murwillumbah is a representation of the entire study area with February-March generally the wettest period.

Temperature ranges are very much consistent throughout the study area with mean maximum temperature of about 25°C during the December to March and mean maximum temperature of about 12°C during June to August. In Murwillumbah (Bray park) the month of January has the highest mean maximum temperature of about 29 °C while July has the lowest mean maximum temperature of around 20 °C in the period of 1972-2010 (Figure 3.3).
Frost is generally rare in the intermediate coastal strip but may occur inland several times in each year (Morand, 1996). The mean annual rainfall of some major urban centres within the study area such as Murwillumbah, Byron Bay, Ballina and Broadwater were 1577.8mm, 1715.2mm, 1752.1mm and 1482.0mm respectively from the period of 1921-1990 (Table 3.1).
Table 3.1: Selected climate data for the major centres within the study area from 1921-1990 (BOM, 2009)

<table>
<thead>
<tr>
<th>Centres</th>
<th>Mean Annual Rainfall (mm)</th>
<th>Mean No. of Rain days&gt;1mm/yr</th>
<th>Mean Annual Maximum Temperature (°C)</th>
<th>Mean Annual Minimum Temperature (°C)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Murwillumbah</td>
<td>1577.8</td>
<td>111.3</td>
<td>25.8</td>
<td>14.4</td>
</tr>
<tr>
<td>Byron Bay</td>
<td>1715.2</td>
<td>119.5</td>
<td>23.7</td>
<td>16.5</td>
</tr>
<tr>
<td>Ballina</td>
<td>1752.1</td>
<td>119.3</td>
<td>24.4</td>
<td>14.2</td>
</tr>
<tr>
<td>Broadwater</td>
<td>1482.0</td>
<td>103.4</td>
<td>24.7</td>
<td>13.5</td>
</tr>
</tbody>
</table>

There is generally adequate soil moisture available for plant growth throughout the year (Edwards, 1979). However, the lower temperatures during the winter period restrict the growth of both tropical and temperate species but temperature is likely to be within the optimal range for these species during summer (Edwards, 1979).

There are extreme natural events in this region which include: cyclones, floods, severe storms, and hailstorms. Cyclonic events have been of different intensity and irregular while flooding due to rainfall can be rare or more frequent depending on the location. Severe storms, not related to cyclones have been increasing since the 1950s while there has been more than one severe hailstorm every year somewhere in the region (Specht, 2008). These extreme events have contributed to beach erosion as the saturated sand moves less freely to compensate for the physical pressure of storm waves than unsaturated sand, resulting in dune face collapse (Specht, 2008). Furthermore, the frequency of these extreme events would also exacerbate wetland erosion.
The tides along this coastal region are mixed, mostly semi-diurnal with two high waters and two low waters each tidal day. The Mean High Water Springs (MHWS) is about 1.4m while the Mean Low Water Springs (MLWS) is about 0.3m above Lowest Astronomical Tide (LAT). The Mean High Water Neaps (MHWN) is around 1.2m while the Mean Low Water Neaps (MLWN) is about 0.5m above Lowest Astronomical Tide (M.S.Q., 2009)

3.3. SOILS

A variety of soil types is present in the area including alluvial krasnozems, chocolate soils, alluvial, red/yellow/brown podzolics, siliceous sands, red earths, skeletal/lithosols and acid sulphate soils (Figure 3.4).
Figure 3.4: Soil types found in the Study Area

Source: GIS Lab SCU
The dominant soil types in the wetland area are siliceous sands, dark loams and alluvials. Siliceous and calcareous sands are found mostly on beaches and dunes while krasnozems, chocolate, and lithosols are commonly found on scarps. Alluvial clays occur on deltaic and back barrier sediments. There is the potential for acid sulphate soils to occur in most parts of the coastal plain (Morand, 1994, 1996; 2001). This is because sea level rise will introduce sulfates in the sea water to the coastal plain which will react with land sediments containing iron oxides and organic matter to produce iron sulfides in the waterlogged sediments. However, with the exposure of these sulfides to air due to reduced rainfall and lowered water tables, they oxidize to produce sulfuric acid which could have severe impacts on estuarine pH and lakes.

In addition to the numerous soil types, the area has different soil landscapes including Empire Vale (ep), Mullumbimby (mu), kooyong (ky), Burns Point (bp), Tyagarah (ty), Tuckean (tu), Cudgen (cu), Angels Beach (ab) and Coraki (ck) (D.T. Morand, 2001).

3.4. GEOLOGY

Most of the area lies predominantly on Mesozoic sediments within the Clarence-Moreton Basin. It consists of metamorphic rocks and sediments that have been overlain by tertiary volcanics of the Mt Warning shield volcano. There are outcrops of volcanics in the region consisting of alkaline basalt, andesite and andesitic breccia (Smith, Miyake, & Houston, 1998). Quaternary (Pleistocene) alluvium occurs over some area along the sheet and is associated with delta, estuary and similar features on small catchments along the coastal plain. Large deposits of Quaternary sand occur along the coast. This consists
of marine and Aeolian quartz sands that have formed beaches, dunes and sandsheets. The older Pleistocene dune systems lie inland of the younger Holocene beaches, foredunes and hind-dunes. The sand masses have been subjected to extensive mining. Estuarine muds and clays occur within creeks and lagoons while peats are present in swamps formed in swales and deflation depressions (Morand, 1994, 1996; 2001).

The oldest rocks in the study area forming the floor of the basin include: the Palaeozoic Neranleigh-Fernvale Group, Triassic/Jurassic Bundamba Group, Jurassic Walloon Coal Measures, Jurassic-Cretaceous Kangaroo Creek Sandstone and the Upper Jurassic-Lower Cretaceous Grafton Formation.

The Palaeozoic Neranleigh-Fernvale Group: This was previously known collectively as the Fitzroy Beds (Chesnut & Swane, 1980) and they are made up of a series of thinly bedded fissile shales, siltstones and sandstones with sometimes more massive units such as greywackes, tuffs, agglomerates, sandstones and massive cobble conglomerates (Chesnut & Swane, 1980). Extensive deformation and folding has resulted in very steep dipping strata. Outcrops of the Neranleigh-Fernvale Group include Cape Byron and Broken Head.

Triassic/Jurassic Bundamba Group: Triassic/Jurassic Bundamba Group occurs very close to the Neranleigh-Fernvale Group and consists of thickly bedded coarse-grained quartzose sandstone (Wells & O’Brien, 1994). Around Broken Head, it is made up of a thickly bedded sequence of pebble sandstones (Chesnut & Swane, 1980)
Jurassic Walloon Coal Measures: These are thinly bedded lithic sandstones, shales and coal seams (Chesnut & Swane, 1980) with concretionary ironstone (McElroy, 1962). The Walloon Coal Measures erode easily and form a subdued landscape of low hills. They extend through Nimbin Gap into the Tweed catchment (McElroy, 1962).

Jurassic-Cretaceous Kangaroo Creek Sandstone: It consists of thinly to moderately thickly bedded, medium to coarse quartz sandstones (Chesnut & Swane, 1980). It lies conformable over the Walloon Coal Measures and is generally exposed as footslopes in the major streams such as Fawcetts, Goolmanger and Terania Creeks. It also occurs as very low residual hills west of and at Coraki (Morand, 1994).

Upper Jurassic-Lower Cretaceous Grafton Formation.: Upper Jurassic-Lower Cretaceous Grafton Formation consists of thinly bedded fine to medium grained shaly and silty greenish to brownish lithic sandstones with also siltstones, claystones and coal. It lies conformably over the Kangaroo Creek Sandstones with a relatively sharp contact (Chesnut & Swane, 1980). Grafton Formation occurrence forms the low hills north of Casino and the very low residual rises and hills south of Casino with a thick veneer of Richmond River alluvium.

These old geologic rocks which formed the floor of the Clearance-Moreton basin have contributed to the large deposits of siliceous sands, dark loams and alluvials that make up most of the wetland soil types in the area.
3.5. LAND USE AND VEGETATION

There is a varied land use in the area and this has given rise to land use conflicts especially in highly urbanized areas. The following land use activities are being carried out in the area: a) tourism concentrated along the coastal strip b) Sugar cane growing c) dairy and beef grazing d) Banana growing e) Quarries f) national parks g) Residential development and urbanization. The rate of land use change varied among shires in the region with a increase rate of 17.8% for agricultural development and 3.7% for residential development in the period of 1998-2004 within the Ballina Shire (BSC, 2004).

In addition to climate change, land use change would affect the wetland size and extent. The rapid expansion in residential and agricultural development in this region, greatly threaten the existence of the wetland ecosystems.

The dominant vegetation types in the north coast region include: rainforests, woodlands, closed-grassland and heaths. The rainforests consist of plant species such as *Argyrodendron* spp., *Araucaria cunninghamii*, *Flindersia australis*, *Ceratopetalum apetalum*, *Doryphora sassafras*, *Lophostemon confertus* and *Eucalyptus pilularis*. The woodlands are dominated by plant species such as *Eucalyptus gunnerifla*, *Eucalyptus maculata*, *Eucalyptus molucanna*, *Angophora subvelutina*, *Melaleuca quinquenervia* and *casuarina glauca*. The dominant plant species for closed-grassland include: *Eucalyptus punctata*, *Eucalyptus tereticornis* and *Angophora floribunda*. The heaths mostly consist of *Banksia integrifolia*, *Banksia aemula* and *Acacia longifolia* (Morand, 1994). *Banksia*
aemula is commonly found in dunal wetlands while Melaleuca quinquenervia is the dominant species for forested wetlands in the area.

3.6. HOLOCENE HISTORY OF COASTAL BARRIER DEPOSITS

Coastal barriers and barrier islands are landforms which are elongated and formed by the deposition of beach material offshore, or across the mouths of inlets or embayments (Bird, 2008). They stand as a barrier between the open sea and an earlier coastline and extend above the normal level of highest tides (Shepard, 1952) and may partly or wholly enclosed lagoons and swamps (Schwartz, 1973). The Holocene barrier systems along the coast of north-eastern NSW have no continuous barrier island chains and consists mostly of quartzose sand that can readily be reshaped by wave or wind action. They are typically called the outer barrier (Troedson, Hashimoto, Jaworska, Malloch, & Cain, 2004) and were formed during the Holocene marine transgression where the rate of sea level rise was rapid (ca. 0.01 m yr) in the early part of the transgression to approximately 8000 years BP, then slowing down to approximately half this rate prior to attaining the present level at 6500–6000 years BP (Roy & Boyd, 1996). Coastal barriers then resulted from the shoreward sweeping of beach sediments during and since the Holocene especially where Pleistocene deposits that had been stranded on the emerging sea floor during the preceding marine regression were collected and built up, then driven landward as the marine transgression proceeded. This shoreward drift of beach sediment may have been facilitated by minor emergences during the oscillating Holocene marine transgression which set up broomlike shoreward sweeping (Bird, 2008).
3.7. SUMMARY AND CONCLUSION

The north-eastern coastal region of NSW is a low land area with a subtropical climate. The wettest rainfall months are February to March which often leads to flooding in some areas depending on the location. The tide is semi-diurnal with a mean high water spring level of about 1.4m. There are numerous soil types in region with siliceous sands, dark loams and alluvials being the dominant wetland soil type, formed due to the old rock groups which formed the floor of the basin. These include the Palaeozoic Neranleigh-Fernvale Group, Triassic/Jurassic Bundamba Group, and Jurassic Walloon Coal. The region's coastal outer barriers consist predominantly of quartzose sand deposited during the Holocene marine transgression. Furthermore, the coastal wetlands have a high biodiversity with plant species such as *Banksia integrifolia*, *Banksia aemula* and *Melaleuca quinquenervia*. Nevertheless, these wetland ecosystems are being threatened by both climate change and anthropogenic factors such as urbanization and agricultural development.
CHAPTER FOUR

4.0. WETLAND MAPPING AND MONITORING IN NORTH-EASTERN NSW, AUSTRALIA

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Please see full article i.e. (Akumu et al., 2010a) in Appendix 1.

I would like to acknowledge the two anonymous reviewers for their constructive comments.

Monitoring coastal wetland communities in north-eastern NSW using ASTER and Landsat satellite data

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Declaration of Authorship

Components of this chapter relating to monitoring coastal wetlands communities were done in entirety by C.E. Akumu in partial fulfilment of his PhD. C.E. Akumu led the writing of the paper. S.Pathirana, S.Baban and D. Bucher reviewed the manuscript prior to submission to Wetland Ecology and Management. The relative contributions of the four authors to the manuscript are indicated below.

Conception of the study: CEA (50%), SP (50%)
Design of the study: CEA (50%), SP (50%)
Collection of data: CEA (90%), SP (10%)
Analysis of data: CEA (90%), SP (5%), SB (5%)
Interpretation of data: CEA (90%), SP (5%), DB (5%)
Conclusions: CEA (95%), SP (5%)
Writing up of paper: CEA (100%)
4.1. INTRODUCTION

In chapter two, the different wetland types and their characteristics were discussed. However, in order to assess the potential impacts of climate change on wetlands; there is a need to identify and map the geographic distribution of the wetland types. The coastal wetlands in north-eastern NSW will be identified, mapped and monitored in this chapter. This would be achieved using satellite data acquired in the periods of September 1989, June 2001 and February 2009.

Wetland conservation is necessary because they are among the world’s most productive environments upon which countless species of plants and animals depend for survival (Ramsar, 1971a). Due to the importance of wetlands; a global intergovernmental treaty on wetland conservation was ratified on February 2nd 1971 known as the Ramsar Convention. The mission of the convention is “the conservation and wise use of all wetlands through local, regional and national actions and international cooperation, as a contribution towards achieving sustainable development throughout the world” (Ramsar, 1971b).

Monitoring would be of important in wetland conservation because it allows us to determine whether these ecosystems have changed over time in terms of size, extent and quality. Furthermore, their reliance on rainfall, surface runoff, groundwater levels and evaporation rates make them, and the ecological services they provide, vulnerable to even small climatic changes therefore monitoring is necessary.
The use of satellite remote sensing approach to monitor wetlands is advantageous because of its ability to map large wetland area. Furthermore, satellite remotely sensed data is less costly, readily available and easier to process compared to orthophoto. Also, satellite images of the earth’s surface are taken continuously over time which makes it easier for monitoring. In addition, due to the fact that satellite data are in digital format, they are easily integrated into GIS for further analysis (Ozesmi & Bauer, 2002).

Wetland mapping and monitoring using satellite data has been successfully used globally. Cho et al. (2004) used Landsat TM and IRS 1-C to map and monitor coastal wetlands in India. They found significant changes in the extent of wetlands such as mangroves and mudflats over a period of time. Landsat MSS was used to monitor changes in wetlands in East Africa (Haack, 1996). The study found a significant expansion in the wetland area over a period of 15 years. In Australia, Roshier and Rumbachs (2004) mapped wetlands in western New South Wales and South Western Queensland using NOAA-AVHRR images. They distinguished wet and dry lakes and estimated the surface area of water bodies using spectral matching. Furthermore, in 2008, the Department of Environment and Climate Change used Landsat and SPOT data to map and monitor the extent, condition and distribution of the Macquarie marshes in central west New South Wales. It identified and distinguished inundated areas and fringing zones within the Macquarie marshes. Even though wetland mapping and monitoring have been carried out in some parts of NSW, there is no known research of wetland monitoring in the north-eastern coastal region of NSW i.e. what are
the wetland types present?, what is the extend of wetland change over time and how successful is satellite remote sensing in wetland classification?

The main objective of this chapter is to map the current and past wetland communities in north-eastern NSW in order to monitor any changes in quality (greenness) and extent.

4.2. MATERIALS AND METHOD

Landsat TM images of September 1989 and February 2009 including Landsat ETM+ of June 2001 were used to map and monitor the wetland communities (Table 4.1). Landsat images were used because of the availability of cloud free images and also because of its high spatial resolution suitable for local/regional application relative to Moderate-resolution Imaging Spectroradiometer (MODIS) or Advanced Very High Resolution Radiometer (AVHRR).

Table 4.1: Characteristics of sensors

<table>
<thead>
<tr>
<th>Property</th>
<th>Landsat TM</th>
<th>Landsat ETM+</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bands and spatial resolution</td>
<td>Bands 1-5 &amp; 30 x 30 m, Band 6 120 x 120 m</td>
<td>Bands 1-5 &amp; 7, 30 x 30 m, Band 6 60 x 60 m, Band 8 15 x 15 m</td>
</tr>
<tr>
<td>Swath width</td>
<td>185 Km</td>
<td>185 Km</td>
</tr>
<tr>
<td>Repeat coverage interval</td>
<td>16 days</td>
<td>16 days</td>
</tr>
<tr>
<td>Orbit type</td>
<td>Sun-synchronous</td>
<td>Sun-synchronous</td>
</tr>
<tr>
<td>Altitude</td>
<td>705 Km</td>
<td>705 Km</td>
</tr>
<tr>
<td>Imagery Date</td>
<td>16/09/1989 &amp; 27/02/2009</td>
<td>21/06/2001</td>
</tr>
</tbody>
</table>
4.2.1. Image Processing

The following image processing steps were performed on the satellite images a) Georeferencing b) Subsetting c) Masking d) Converting DN to Radiance e) Converting Radiance to Reflectance f) Classification and g) Accuracy Assessment (Figure 4.1). Image processing has the ability to extract wetland information by analysing spatial, temporal and spectral patterns of the satellite data. Furthermore, the output information can be validated, and integrated with geographic information system (Adam & Gillespie, 2006; Campbell, 2006; Lillesand & Kiefer, 2000). The satellite images were acquired and converted from Binary BIL format into ER Mapper data format in ER Mapper software 7.1. The Landsat scenes were georeferenced based on an orthophoto and re-projected into the UTM-WGS 84 projection. Twenty-four ground control points were used in the rectification process with an overall RMS error of less than 1 pixel. The images were later subsetted to the study area and a DEM was used to mask out all areas that are more than 20 m altitude from the coastline. This is because the coastal zone was defined as all areas less than 20 m elevation from the coastline.
Figure 4.1: A flow chart of the image processing stages
Radiometric correction was carried out in the visible and near infrared regions of the satellite images. Radiometric correction entails the correction of image pixel values for sun elevation angle variation and the calibration of images to account for degradation of the sensors over time. The changes in sensors calibration factors will obscure real changes on the ground (Mather, 1999). This process involves the conversion of digital numbers to at-satellite radiances and at-satellite radiance to at-surface reflectance. This radiometric correction process includes atmospheric and illumination corrections.

4.2.1.1. Conversion of DN to Radiance

Landsat ETM+ and TM images

\[ L_{\text{rad}} = \text{Bias} + (\text{Gain} \times \text{DN}) \]  
\[ \text{Gain} = \frac{\text{Lmax}}{254} - \frac{\text{Lmin}}{255} \]

\[ \text{Bias} = \text{Lmin} \]

Therefore

\[ L_{\text{rad}} = \text{Lmin} + \left(\frac{\text{Lmax}}{254} - \frac{\text{Lmin}}{255}\right) \times \text{DN} \]

The spectral values of gain and bias for Landsat ETM+ (Table 4.2) were obtained from the image header file.

**Table 4.2:** Spectral values of bias and gain from the Landsat ETM+ image header file

<table>
<thead>
<tr>
<th>Bands</th>
<th>Gain</th>
<th>Bias</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>0.7756863</td>
<td>-6.1999969</td>
</tr>
<tr>
<td>2</td>
<td>0.7956862</td>
<td>-6.3999939</td>
</tr>
<tr>
<td>3</td>
<td>0.6192157</td>
<td>-5.0000000</td>
</tr>
<tr>
<td>4</td>
<td>0.6372549</td>
<td>-5.1000061</td>
</tr>
</tbody>
</table>
The Lmin and Lmax values (Table 4.3) used for Landsat TM scenes were adopted from Chander et al. (2007).

**Table 4.3:** Post-Calibration Dynamic Range Values for Landsat TM-5 images used

<table>
<thead>
<tr>
<th>Satellite</th>
<th>Landsat TM</th>
<th>Landsat TM</th>
</tr>
</thead>
<tbody>
<tr>
<td>Date of Acquisition</td>
<td>16/09/1989</td>
<td>27/02/2009</td>
</tr>
<tr>
<td>Band</td>
<td>Lmin</td>
<td>Lmax</td>
</tr>
<tr>
<td>-----</td>
<td>-----</td>
<td>-----</td>
</tr>
<tr>
<td>1</td>
<td>-1.52</td>
<td>152.10</td>
</tr>
<tr>
<td>2</td>
<td>-2.84</td>
<td>296.81</td>
</tr>
<tr>
<td>3</td>
<td>-1.17</td>
<td>204.30</td>
</tr>
<tr>
<td>4</td>
<td>-1.51</td>
<td>206.20</td>
</tr>
</tbody>
</table>

4.2.1.2. Conversion of Radiance to Reflectance

*Landsat ETM+/TM Images*

\[
R_{\text{TOA}} = \left( \frac{\pi \times L_{\text{rad}} \times d^2}{\text{ESUN}_i \times \cos (z)} \right) \quad \text{(4.3)}
\]

Where:

- \(R_{\text{TOA}}\): the planetary reflectance
- \(L_{\text{rad}}\): is the spectral radiance at the sensor’s aperture;
- \(\pi\): \(\approx 3.14159\)
- \(\text{ESUN}_i\): the mean solar exoatmospheric irradiance of each band
- \(d\): the earth-sun distance, in astronomical units, which is calculated using the following EXCEL equation (Archard & D’Souza, 1994; Eva & Lambin, 1998)

\[
d = (1 - 0.01672 \times \text{COS} (\text{RADIANS} (0.9856 \times (\text{Julian\_Day} - 4)))).
\]
z: solar zenith angle (zenith angle = 90 – solar elevation angle), solar elevation angle is within the header file of the satellite images.

Normalize Difference Vegetation Index (NDVI) of the reflectance images was generated using band 3 (Visible Red) and 4 (Near-infrared) of the Landsat images. This was carried out using the equation 4.4.

\[
\text{NDVI} = \frac{\text{Near-infrared - Visible red}}{\text{Near-infrared} + \text{Visible red}}
\]

4.2.1.3. Classification

Both supervised and unsupervised classifications (Figure 4.1) were carried on the Landsat images in ER Mapper version 7.1. Unsupervised classification was performed using ISOCLASS algorithm with maximum number of 10 classes. This examines the unknown pixels in the image and aggregates them into a number of classes based on the natural groupings or cluster present in the image. The classes were visually interpreted and compared with \textit{apriori} classes from groundtruthing. Supervised classification was performed on the reflectance images using a set of user-defined classes. This requires digitizing training sites into user-defined polygons based on knowledge of the wetlands classes obtained from regular field visits. The same polygons were used in all reflectance images. Statistics of the training sites were generated and evaluated. Supervised classification was performed using the Maximum likelihood standard algorithm. This is because it considers both the means and the variances of the training data in order to approximate the probability that a given pixel belongs to a particular class. Also, it produces better results compared to minimum distance to means or
parallelpiped classifiers (Ozesmi & Bauer, 2002). Non-wetland areas such as urban and agricultural areas were masked out from the classification.

In order to evaluate the accuracy of the classification, accuracy assessment was performed on the 2009 Landsat classified map. The classified map was compared with a geo-referenced topographic map and colour orthophoto of the region. 312 samples were selected from the classified map through random sampling method from the wetland classes and compared with the referenced map. Furthermore, groundtruthing was carried out in order to validate some of the wetland classes. A $k \times k$ confusion matrix (error matrix) table (Table 4.4) was developed and the producer accuracy, user accuracy, overall accuracy and kappa analysis were calculated. The maps were later exported to Geographic Information System (i.e. ArcGIS 9.3 software) for analysis of changes in wetland size and extent.

4.3. RESULTS AND DISCUSSION

The study found the following wetland types in the study area: mangroves and salt marshes, coastal swamps, dunal wetlands, forested wetlands, coastal upland water bodies, and estuarine water bodies (Figure 4.2, 4.3 & 4.4). Mangroves and saltmarshes were estuarine areas which are subjected to tidal flooding and support mangrove and salt marsh vegetation (Green, 1997). The dominant mangrove species found in this area was *Avicennia marina* (grey mangrove). Forested wetlands were dominated by trees and occurred on fertile soils, mostly at low altitude. The vegetation was dominated by broad-leaved paperbark (*Melaleuca quinquenervia*). Dunal wetlands were fresh water wetlands on coastal sand dunes or plains that
support woodland, heathland, sedges, and rushes. Coastal swamps were fresh water wetlands around the coast that consisted of shallow marshes, wet heaths, and meadows vegetated by sedges and aquatic herbs. Species common in this classification included: *Leptospermum* spp. *Melaleuca squarrosa*, *Callistemon citrinus*, *Epacris* spp., *Hakea teretifolia*, *Bauera* spp. *Dillwynia floribunda*, and *Symphionema paludosum*. Estuarine water bodies were large open saline or brackish water bodies with a relatively narrow permanent or intermittent connection to the sea. Coastal upland water bodies were large or small fresh or brackish water bodies along the coast and included lakes, rivers and ponds. Mangroves and saltmarshes were found along the estuarine waters. This is probably because they prefer areas subjected to tidal flooding.
Figure 4.2: Wetland classes after supervised classification of the study area - Landsat TM, September 1989
Figure 4.3: Wetland classes after supervised classification of the study area- Landsat ETM +, June 2001
Figure 4.4: Wetland classes after supervised classification of the study area - Landsat TM, February 2009
The mean Normalize Difference Vegetation Index (NDVI) value for mangroves and saltmarshes were 0.430, 0.665 and 0.660 in the months of September 1989, June 2001 and February 2009 respectively. Dunal wetlands had a mean NDVI value of 0.470 in September 1989, 0.665 in June 2001 and 0.670 in February 2009. The mean NDVI value of 0.360 was for forested wetlands in the September 1989 scene. This increased to 0.685 in the June 2001 scene and reduced to 0.650 in the February 2009 scene. Coastal swamps had a mean NDVI value of about 0.335 in September 1989. This increased to 0.565 in June 2001 and to 0.610 in February 2009 (Figure 4.5, 4.6, 4.7 & 4.8). In the month of February 2009, the highest mean NDVI value of 0.670 was for dunal wetlands and the lowest NDVI value of 0.610 was for coastal swamps (Figure 4.5). This indicates that the dunal wetlands were healthier and greener relative to the other wetland communities such as forested wetlands and swamps. Forested and dunal wetlands had the highest mean NDVI values of 0.685 and 0.470 in the period of June 2001 and September 1989 respectively while coastal swamps had the lowest mean NDVI values of 0.565 and 0.335 in the months of June 2001 and September 1989 respectively (Figure 4.6 and 4.7).
**Figure 4.5:** NDVI values for wetlands in north-eastern NSW in February 2009
Error bars represents standard error of the mean (S.E.M).

**Figure 4.6:** NDVI values for wetlands in north-eastern NSW in June 2001
Error bars represents standard error of the mean (S.E.M).

**Figure 4.7:** NDVI values for wetlands in north-eastern NSW in September 1989
Error bars represents standard error of the mean (S.E.M).

**Figure 4.8:** Changing NDVI values for wetlands in 1989, 2001 & 2009
The range of mean NDVI values for wetlands in February 2009, June 2001 and September 1989 were 0.610-0.670, 0.565-0.685 and 0.335-0.470 respectively. The low range of mean NDVI values (0.335-0.470) for wetlands in September 1989 images relative to June 2001 and February 2009 values i.e. 0.565-0.685 and 0.610-0.670 respectively is probably due to low rainfall in the month of September 1989. The average rainfall values obtained from the Bureau of Meteorology were about 13.7mm, 59.7mm and 255mm in the months of September 1989, June 2001 and February 2009 respectively.

The study also found changes in the extent of wetlands in the months of September 1989, June 2001 and February 2009 (Table 4.5). The total area of mangroves and salt marshes in September 1989 was around 36.80 Km$^2$ (7.42%). This area decreased to 36.56 Km$^2$ (7.60%) in June 2001 and increased to around 36.58 Km$^2$ (7.61%) in the month of February, 2009. The total area covered by coastal upland water bodies at the time of satellite image acquisition was about 32.80 Km$^2$ (6.62%) in the month of September 1989. This decreased to 32.74 Km$^2$ (6.80%) in the June 2001 scene and increased to around 32.76 Km$^2$ (6.82%) in the February 2009 scene. This fluctuation in coastal upland water bodies may be due to precipitation and temperature. This is because an increase in precipitation would increase the area occupied by lakes and rivers while temperature increase would lead to high evaporation rate and therefore a decrease in surface area (Beresford et al., 1981).
Table 4.4: Error Matrix Table for Landsat TM, February 2009 wetland classification

<table>
<thead>
<tr>
<th>Remote sensing data</th>
<th>F</th>
<th>M</th>
<th>D</th>
<th>S</th>
<th>L</th>
<th>E</th>
<th>Row Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>F</td>
<td>43</td>
<td>0</td>
<td>4</td>
<td>3</td>
<td>0</td>
<td>0</td>
<td>50</td>
</tr>
<tr>
<td>M</td>
<td>0</td>
<td>49</td>
<td>0</td>
<td>3</td>
<td>0</td>
<td>0</td>
<td>52</td>
</tr>
<tr>
<td>D</td>
<td>4</td>
<td>0</td>
<td>44</td>
<td>3</td>
<td>0</td>
<td>0</td>
<td>51</td>
</tr>
<tr>
<td>S</td>
<td>5</td>
<td>1</td>
<td>0</td>
<td>47</td>
<td>0</td>
<td>0</td>
<td>53</td>
</tr>
<tr>
<td>L</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>48</td>
<td>2</td>
<td>50</td>
</tr>
<tr>
<td>E</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>3</td>
<td>53</td>
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<tr>
<td>Column Total</td>
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<td>50</td>
<td>48</td>
<td>56</td>
<td>51</td>
<td>55</td>
<td>312</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Reference data</th>
<th>Producer’s Accuracy</th>
<th>User’s Accuracy</th>
<th>Overall Accuracy</th>
<th>Overall Kappa Statistics</th>
</tr>
</thead>
<tbody>
<tr>
<td>D = Dunal wetlands</td>
<td>91.67%</td>
<td>86.27%</td>
<td></td>
<td></td>
</tr>
<tr>
<td>F = Forested wetlands</td>
<td>82.69%</td>
<td>86.00%</td>
<td></td>
<td></td>
</tr>
<tr>
<td>S = Coastal swamps</td>
<td>83.93%</td>
<td>88.68%</td>
<td></td>
<td></td>
</tr>
<tr>
<td>L = Coastal upland water bodies</td>
<td>94.12%</td>
<td>96.00%</td>
<td></td>
<td></td>
</tr>
<tr>
<td>M = Mangroves and saltmarshes</td>
<td>98.00%</td>
<td>94.23%</td>
<td></td>
<td></td>
</tr>
<tr>
<td>E = Estuarine water bodies</td>
<td>96.36%</td>
<td>94.64%</td>
<td>91.03%</td>
<td>89.23%</td>
</tr>
</tbody>
</table>

Forested wetlands had a total area of about 162.01 Km² (32.70%) in the month of September 1989. This area decreased significantly to about 152.09 Km² (31.60%) and 149.26 Km² (31.08%) in the months of June 2001 and February 2009 respectively. The continuous decrease in forested wetlands may be attributed to anthropogenic factors such as an increase in urbanization. This is because the region is the fastest growing non metropolitan region in the state of NSW (Morand, 1996). The total area for estuarine water bodies at the time of satellite
image acquisition was about 35.50 Km$^2$ (7.17%), 35.97 Km$^2$ (7.47%) and 35.91 Km$^2$ (7.47 %) in the months of September 1989, June 2001 and February 2009 respectively.

The similar area covered by estuarine water bodies is probably due to the similar tide level at the time of satellite image acquisition. The tide level was 1.02m during the February 2009 scene acquisition, 1.07m and 1.23m during the June 2001 and September 1989 scenes acquisition. Coastal swamps had a total area of about 152.50 Km$^2$ (30.78%) in the month of September 1989. This decreased to 150.56 Km$^2$ (31.28%) in the month of June 2001 and increased to about 150.89 Km$^2$ (31.42 %) in the month of February 2009. The total area of dunal wetlands was about 75.75 Km$^2$ (15.29%), 73.37 Km$^2$ (15.24%) and 74.78 Km$^2$ (15.57 %) in the months of September 1989, June 2001 and February 2009 respectively.


<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Mangroves and salt marshes</td>
<td>36.80</td>
<td>7.42</td>
<td>36.56</td>
<td>7.60</td>
<td>36.58</td>
<td>7.61</td>
<td>-0.65</td>
<td>-0.60</td>
<td>+0.06</td>
</tr>
<tr>
<td>Coastal upland water bodies</td>
<td>32.80</td>
<td>6.62</td>
<td>32.74</td>
<td>6.80</td>
<td>32.76</td>
<td>6.82</td>
<td>-0.18</td>
<td>-0.12</td>
<td>+0.06</td>
</tr>
<tr>
<td>Forested wetlands</td>
<td>162.01</td>
<td>32.70</td>
<td>152.09</td>
<td>31.60</td>
<td>149.26</td>
<td>31.08</td>
<td>-6.12</td>
<td>-7.87</td>
<td>-1.86</td>
</tr>
<tr>
<td>Estuarine water bodies</td>
<td>35.50</td>
<td>7.17</td>
<td>35.97</td>
<td>7.47</td>
<td>35.91</td>
<td>7.47</td>
<td>+1.32</td>
<td>+1.15</td>
<td>-0.17</td>
</tr>
<tr>
<td>Coastal swamps</td>
<td>152.50</td>
<td>30.78</td>
<td>150.56</td>
<td>31.28</td>
<td>150.89</td>
<td>31.42</td>
<td>-1.27</td>
<td>-1.01</td>
<td>+0.22</td>
</tr>
<tr>
<td>Dunal wetlands</td>
<td>75.75</td>
<td>15.29</td>
<td>73.37</td>
<td>15.24</td>
<td>74.78</td>
<td>15.57</td>
<td>-3.14</td>
<td>-1.28</td>
<td>+1.92</td>
</tr>
</tbody>
</table>
The change in mangroves and saltmarshes between 1989 and 2001 was about -0.65% relative to about +0.06% in the period of 2001 and 2009. This trend was similar with coastal swamps which had a change of around -1.27% between 1989 and 2001, +0.22% between 2001 and 2009. Forested wetland change between 1989 and 2001 almost tripled the change between 2001 and 2009. The change in coastal upland water bodies between 1989 and 2001; 2001 and 2009 were -0.18% and +0.06% respectively in contrast to +1.32% and -0.17% for estuarine water bodies. Among the wetland types between the year 2001 and 2009, dunal wetlands had the most change of about +1.92% while mangroves and saltmarshes; and coastal upland water bodies had the least change of around +0.06%.

The highest producer’s accuracy i.e. 98.00% was for mangroves and saltmarshes. The producer’s accuracy was performed by dividing the total correct sample units of each wetland category by the total number of the wetland class sample units as indicated by the reference data (i.e. column total). Coastal upland water bodies had the highest user’s accuracy of 96.00%. The user’s accuracy was computed by dividing the total number of correct pixels in each wetland category by the total number of pixels classified in that category (i.e. row total). The minimum producer’s and user’s accuracy was found in the classification of forested wetlands. This is probably because forested wetlands had close reflectance values with other wetland types such as coastal swamps as some forested stands contained wet heath along the coast. The wetlands were successfully classified with an overall accuracy of 91.03% and this was computed by using the sum of the major diagonal (i.e. the correctly classified sample units) divided by the total number of sample units in the entire error matrix (Congalton & Green, 1999). The kappa statistical value for the classification was 89.23%. This is a measure
of how well the remotely sensed classification agrees with the reference data. The kappa analysis from the classification showed a high correlation between the wetland classification and the reference data used.

Change detection techniques such as image differencing, subtraction or regression were not used in the analysis because of limitation in the acquisition dates of the satellite imageries. The inconsistency in the seasonality of the time periods chosen for analysis i.e. September, June and February limited the use of change detection techniques in the analysis due to differences in vegetation communities and physical conditions among seasons.

4.4. CONCLUSIONS

The mapping and monitoring of the coastal wetlands using satellite data has been carried out successfully. The wetlands are easily classified and monitored using satellite remote sensing. Landsat TM of February 2009, September 1989 and Landsat ETM+ of June 2001 were processed and used to map and monitor the coastal wetlands. The are several wetland classifications in this subtropical region of Australia such as mangroves and saltmarshes, forested wetlands, coastal swamps, dunal wetlands, coastal upland water bodies and estuarine water bodies. The range of mean NDVI values for the coastal wetlands in February 2009, June 2001 and September 1989 were 0.610-0.670, 0.565-0.685 and 0.335-0.470 respectively. The areas covered by the wetlands were different in the months of February 2009, June 2001 and September 1989. This is most probably due to environmental conditions at the time of satellite image acquisition.
CHAPTER FIVE

5.0. MODELING THE POTENTIAL IMPACT OF CLIMATE CHANGE ON THE SPATIAL DISTRIBUTION OF WETLAND SPECIES IN NORTH-EASTERN NSW, AUSTRALIA

5.1. INTRODUCTION

In chapter four, the coastal wetlands in north-eastern NSW were mapped and monitored using satellite data of the periods September 1989, June 2001 and February 2009. Interestingly, the wetlands were of different extent and quality at the time of satellite image acquisition. Nonetheless, to better understand the possible impacts of future climate change on these wetlands, it is imperative that we assertively predict the current and future potential spatial distribution of their common wetland species. This chapter will predict the potential spatial distribution of selected wetland species as a result of climate change i.e. mean annual temperature increase.

Climate change would have an effect on the natural distribution of wetland species. There are evidence from recent studies that showed that climate change has a profound impact on species range expansion and contraction (Davis et al., 2001; Hughes, 2000; Huntley, 1999; McCarty, 2001; Walther et al., 2002; Woodward, 1987). The IPCC (2001) report has predicted that by the end of the century, global average temperature could rise to as high as 6.4°C. Furthermore, CSIRO projection for NSW, Australia indicates an increase in annual-average warming of 0.2 to 1.6°C by the year 2030 in coastal and southern regions relative to 1990. By 2070, warming is expected to increase
to 0.7 to 4.8°C in the coastal and southern regions provided CO₂ concentrations are not stabilised by 550 ppm by the year 2150 (Hennessy et al., 2004). This therefore suggests that more substantial effects on wetland species would occur by the end of the century. It is therefore important that such changes are predicted so that appropriate adaptations can be suggested.

There have been some studies in predicting the potential spatial distribution of species due to climate change globally. These include the prediction of stiff sedge distribution in the United Kingdom and Ireland (Pearson et al., 2002) and the potential distribution of North American trees as a result of climate change (Mckenney et al., 2007). These studies generally found a decrease in the suitable climate space of the species with climate change. In Australia, various studies have investigated the potential spatial distribution of some selected plant and animal species including Williams (2007) and Kriticos et al. (2003). Williams (2007) predicted the likely ecological impacts of climate change on the wet tropics heritage area in Queensland. He found a decrease in the geographic pattern of species richness of regionally endemic rainforest vertebrates with an increase in temperature. Furthermore, Kriticos et al. (2003) modeled the potential spatial distribution of an invasive alien plant Acacia nilotica ssp. Indica in Australia. They found that global climate change is likely to increase the potential spatial distribution of Acacia nilotica in Australia thereby increasing the risk of invasion. In addition, Brereton et al. (1995) studied the potential habitats for 42 vertebrate species. They found that 41 species had their range reduced and 15 species were projected to lose their climatic space completely for a 3 °C warming. They also found that mountain
pygmy possum *Burramys parvus* lost its climatic habitat with a 1 °C rise in temperature. Furthermore, Chapman and Milne (1998) studied the ranges of a number of plants and animals using bioclimatic modeling with vegetation and soil types considered. They found that some species showed large reductions in range while others were either unaffected or gained a little in distribution. Also, Pouliquen-Young and Newman (1999) generated climatic envelopes from the present distribution of species in a forested area in the south-west of Western Australia. They assessed the effects of three incremental temperature and rainfall scenarios on three species of frogs, 15 species of endangered or threatened mammals, 92 varieties of the plant genus *Dryandra*, and 27 varieties of *Acacia*. Their results indicated that most species would suffer dramatic decreases in range with temperature increase. They found that all of the frog and mammal species studied would be restricted to small areas or would disappear with 0.5 °C global-average warming above present annual averages, as would 28% of the *Dryandra* species. Furthermore, they found that at 2°C global average warming, 66% of the *Dryandra* species, as well as all of the *Acacia* species would disappear. Beaumont and Hughes (2002) assessed the potential changes in the distribution of 77 butterfly species restricted to Australia. Even under very conservative climate change projections they found 88% of species distributions decreased by 2050, and under more extreme climate change projections 83% of species decreased in range by at least 50%. Furthermore, Newell *et al.* (2001) modeled the bioclimatic profiles of 12 plant species in Victoria. Their projections at 5-year intervals until 2100 showed a decline in suitable land areas in each case.
Despite the fact that a change in climatic variables such as temperature would affect the survival and distribution of most species, there is still no known research on modeling the potential spatial distribution of most wetlands species especially in north-eastern region of NSW Australia. This study has selected the following wetland species i.e. *Avicennia marina*, *Banksia integrifolia*, *Melaleuca quinquenervia* and *Leptospermum liversidgei* from mangroves and saltmarshes, dunal wetlands, forested wetlands and coastal swamps respectively to predict their potential spatial distribution with climate change (mean annual temperature increase). These species were selected as representatives of the wetland classes because they were either dominant species or are indicator species of the wetland types.

5.2. MATERIALS AND METHOD

5.2.1. BIOCLIM

BIOCLIM is a bioclimatic analysis and predictive system initially developed by Nix (1986) that interpolates up to 35 parameters (Table 5.1) for any given location for which the latitude, longitude and elevation are known. It is part of the ANUCLIM 5.1 package (Houlder et al., 2000). BIOCLIM model assumes that a species can tolerate locations where values of all climatic variables fit within the extreme values determined by the set of known locations (Carpenter et al., 1993). The distribution of species is determined by interpolating the climate within each grid cell of a Digital Elevation Model (DEM) and comparing it to the climatic profile of the species.
Table 5.1: Bioclimatic parameters used in BIOCLIM version 5.1.

<table>
<thead>
<tr>
<th>No.</th>
<th>Climatic variable</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Annual mean temperature</td>
</tr>
<tr>
<td>2</td>
<td>Mean diurnal range</td>
</tr>
<tr>
<td>3</td>
<td>Isothermality</td>
</tr>
<tr>
<td>4</td>
<td>Temperature seasonality</td>
</tr>
<tr>
<td>5</td>
<td>Maximum temperature of warmest period</td>
</tr>
<tr>
<td>6</td>
<td>Minimum temperature of coldest period</td>
</tr>
<tr>
<td>7</td>
<td>Temperature annual range</td>
</tr>
<tr>
<td>8</td>
<td>Mean temperature of wettest quarter</td>
</tr>
<tr>
<td>9</td>
<td>Mean temperature of driest quarter</td>
</tr>
<tr>
<td>10</td>
<td>Mean temperature of warmest quarter</td>
</tr>
<tr>
<td>11</td>
<td>Mean temperature of coldest quarter</td>
</tr>
<tr>
<td>12</td>
<td>Annual precipitation</td>
</tr>
<tr>
<td>13</td>
<td>Precipitation of wettest period</td>
</tr>
<tr>
<td>14</td>
<td>Precipitation of driest period</td>
</tr>
<tr>
<td>15</td>
<td>Precipitation seasonality</td>
</tr>
<tr>
<td>16</td>
<td>Precipitation of wettest quarter</td>
</tr>
<tr>
<td>17</td>
<td>Precipitation of driest quarter</td>
</tr>
<tr>
<td>18</td>
<td>Precipitation of warmest quarter</td>
</tr>
<tr>
<td>19</td>
<td>Precipitation of coldest quarter</td>
</tr>
<tr>
<td>20</td>
<td>Annual mean radiation</td>
</tr>
<tr>
<td>21</td>
<td>Highest period radiation</td>
</tr>
<tr>
<td>22</td>
<td>Lowest period radiation</td>
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<tr>
<td>23</td>
<td>Radiation seasonality</td>
</tr>
<tr>
<td>24</td>
<td>Radiation of wettest quarter</td>
</tr>
<tr>
<td>25</td>
<td>Radiation of driest quarter</td>
</tr>
<tr>
<td>26</td>
<td>Radiation of warmest quarter</td>
</tr>
<tr>
<td>27</td>
<td>Radiation of coldest quarter</td>
</tr>
<tr>
<td>28</td>
<td>Annual mean moisture index</td>
</tr>
<tr>
<td>29</td>
<td>Highest period moisture index</td>
</tr>
<tr>
<td>30</td>
<td>Lowest period moisture index</td>
</tr>
<tr>
<td>31</td>
<td>Moisture index seasonality</td>
</tr>
<tr>
<td>32</td>
<td>Mean moisture index of high quarter</td>
</tr>
<tr>
<td>33</td>
<td>Mean moisture index of low quarter</td>
</tr>
<tr>
<td>34</td>
<td>Mean moisture index of warm quarter</td>
</tr>
<tr>
<td>35</td>
<td>Mean moisture index of cold quarter</td>
</tr>
</tbody>
</table>
BIOCLIM has been mostly used in Australia and has also been successfully applied in other parts of the world such as Africa and North America (Lindenmayer et al., 1996). The output of BIOCLIM is typically a single map and an accompanying bioclimatic profile represented by a set of climatic indices (Jackson & Claridge, 1999; Lindenmayer, et al., 1996).

The methodology (Figure 5.1) involved field survey which was carried out regularly to randomly selected wetland types in order to identify the following wetland species: *Avicennia marina*, *Banksia integrifolia*, *Melaleuca quinquenervia* and *Leptospermum liversidgei*. Their geographic positions and elevations were recorded using a global positioning system (GPS). These data constituted the site file of the species and were imported into BIOCLIM to generate the climate profile of the species (.pro file) using the climate surface of Australia. A 25m pixel size Digital Elevation Model (DEM) of study area was also imported into BIOCLIM to generated the climate parameters of the study area (.bcp file) using the climate surface of Australia (Houlder et al., 2000). Both the climate profile of the species file (.pro file) and the climate parameters of the study area file (.bcp file) were used in BIOMAP to generate the predicted locations of the wetland species with temperature increase. Full set of bioclimatic parameters (all 35 parameters) obtained from the ANUCLIM package version 5.1 were used in the prediction. ‘Core’ areas in this study fell in the range of 2.5 to 97.5%. These are cells in the DEM where the value of each parameter fell within the 2.5 percentile and the 97.5 percentile in the species profile. The predicted locations were exported to GIS for mapping. Furthermore, high soil erosion areas were excluded from the predicted
locations assuming that these species would be not established on high soil erosion areas. This was performed by generating a soil erosion map and using it mask out high erosion areas from the predicted locations. The soil erosion map was produced using the Revised Universal Soil Loss Equation (RUSLE) which included the following parameters: rainfall-runoff erosivity factor, soil erodibility index, the topographic LS factor, the crop and land management factor and the support practice factor.
Figure 5.1: A schematic representation of the methodology displayed for the prediction of suitable habitats for wetland species due to climate change
The Revised Universal Loss Equation (RUSLE) was derived from Universal Soil Loss Equation (USLE) and is written as

\[ A = \frac{R \times K \times LS \times C \times P}{5.1} \]

Where

- \( A \) is the computed soil loss per unit area, usually in tons per hectare per year (t/ha/yr);
- \( R \) is the rainfall–runoff erosivity factor (usually in MJmm/ha h yr);
- \( K \) is the soil erodibility factor (t h/MJ mm), a measure of the susceptibility of soil to erosion;
- \( LS \) is the slope and slope steepness factor;
- \( C \) is the cover and management factor; and
- \( P \) is the conservation support-practices factor.

RUSLE factors and their derivation are properly documented (Renard et al., 1997; Wischmeier & Smith, 1978). This model has been a leading tool for soil erosion prediction and conservation planning in America and other parts of the world (Mitasaova & Mitas, 1999).

**Rainfall-runoff Erosivity factor (R):** It is a measure of the ability of rainfall to cause erosion. It is a product of kinetic energy and maximum 30-minute intensity (EI\textsubscript{30}) for each storm. In the state of NSW, there is a rainfall erosivity contour (isoerodent) map (Landcom, 2004; Rosewell, 1993) which was generated from point measurements across the state of NSW for more than 29 meteorological stations with more than 20 years of records (CaLM, 1995). This was interpolated in ArcGIS 9.3 to create a rainfall grid layer.
with a cell size of 25m using the topogrid programme which is based on the ANUDEM programme developed by Hutchinson (1989).

**Soil Erodibility Index (K factor):** This is the measure of the susceptibility of individual soil particles to detachment and transport by rainfall or runoff. The SI unit is t h/ MJ mm. The contributing factors of soil erodibility are soil texture, structure, organic matter and permeability. Soils with high proportion of clay would usually have low K values because of their high resistance to detachment. Soils with coarse texture such as coarse sand would also have low K values because they generate little runoff even though they are easily detachable. Medium textured soils, such as the silt loam soils have moderate K values whereas high silt content soils have high K values. This is because, high silt content soils are easily detached, tend to crust and produce high rates of runoff (Yang & Chapman, 2006). The K values for the various soil landscapes in the study area (Morand, 1994, 1996; 2001) were added to a theme of a soil landscape map for the region to generate a grid layer of soil erodibility in ArcGIS.

**The Topographic LS factor:** The slope length and steepness (LS) factor represent the effect of slope length and steepness on erosion. The effect of steepness on erosion is greater than slope length. However both are usually considered together as LS factor because they affect the magnitude of erosion. Slope length is defined as the distance from the source of runoff to the point where deposition begins or runoff becomes focused into a defined channel (Simms *et al.*, 2003).

The formula for calculating the computed LS factor is:
\[ T = \left( \frac{A}{22.13} \right)^{0.6} \left( \sin \frac{B}{0.0896} \right)^{1.3} \]  
\[ \text{----------------------------- (5.2)} \]

Where

A is the upslope contributing factor,

B is the slope angle (Simms, et al., 2003)

However, because upslope area better reflects the impact of concentrated flow on increased erosion rather than slope length (Mitasova & Mitas, 1999), slope length was replaced by upslope area in the LS factor. The process for computation in ArcGIS 9.3 was as follows:

- Using Spatial Analyst Extension, slope was derived from 25 m DEM
- Using the Hydrological Extension, sinks in the DEM were identified and filled
- Flow Direction was generated using the filled DEM as input grid
- Flow Accumulation was derived using Flow Direction as input grid
- Using Raster Calculator, LS was computed using the expression

\[
\text{POW} \left( \text{[flow accumulation]} \times 25/22.13, 0.6 \right) \times \text{POW} \left( \left( \sin \left( \text{[slope of DEM]} \times 0.01745/0.0896 \right) \right), 1.3 \right)
\]

**The Crop and Land Management (C factor):** This represents the effect of land use on soil erosion (Renard et al., 1997). It is the ratio of soil loss from land cropped under specific conditions to the corresponding loss from continuous tilled bare fallow (Wischmeier & Smith, 1978). In this study, a Land use/Land cover map was generated using Landsat TM scene and their various C factors allocated for each Land use/Land cover class (Table 5.2). A grid layer of C factor was later generated as an input in the model using ArcGIS
9.3. The land cover types ranged from 0.003 - 1 and were adapted values from (Gonzalez et al., 2008; Printemps et al., 2007; Simms et al., 2003).

**Table 5.2: Land cover types and their C factors**

<table>
<thead>
<tr>
<th>Land Cover Type</th>
<th>C Factor</th>
</tr>
</thead>
<tbody>
<tr>
<td>Natural vegetation</td>
<td>0.003</td>
</tr>
<tr>
<td>Agricultural land</td>
<td>0.45</td>
</tr>
<tr>
<td>Urban</td>
<td>0.85</td>
</tr>
<tr>
<td>Water</td>
<td>0</td>
</tr>
<tr>
<td>Bare land</td>
<td>1</td>
</tr>
<tr>
<td>Cleared Land</td>
<td>0.45</td>
</tr>
</tbody>
</table>

The support Practice (P factor): This corresponds to the soil conservation measures or other operations that control erosion, such as terraces, strip cropping and contour farming. However, because there was no available information for the study area, P was set to 1.

Finally, the soil erosion map (Figure 5.2) was produced using Raster Calculator to build the expression: \( R \times K \times LS \times C \times P \). Ground truthing was carried out in order to validate some of the high soil loss risk areas. The soil erosion map was reclassified in order to exclude the high soil erosion areas (> 10 t/ha/yr). The output map i.e. low erosion map (<10 t/ha/yr) was intersected with the predicted location maps from BIOCLIM in order to generate the suitable habitat maps for the wetland species with temperature increase (Figures 5.3-5.6).
5.3. RESULTS AND DISCUSSION

The study found the potential soil loss sites of less than 1 t ha\(^{-1}\) yr\(^{-1}\), 1-10 t ha\(^{-1}\) yr\(^{-1}\) and more than 10 t ha\(^{-1}\) yr\(^{-1}\) in the north-eastern region of NSW. The low erosion risk areas had a potential soil loss of less than 1 t ha\(^{-1}\) yr\(^{-1}\). The medium erosion risk areas had a potential soil loss in the range of 1-10 t ha\(^{-1}\) yr\(^{-1}\) while the high erosion risk areas had a potential soil loss of more than 10 t ha\(^{-1}\) yr\(^{-1}\).

**Figure 5.2:** Potential soil loss based on Revised Universal Soil Loss Equation (RUSLE) from Tweed Heads to Evans Head
The predicted potential distribution (Figure 5.3) of *Avicennia marina* (Grey Mangrove) indicates that only the suitable climate space found within the tidal range would provide suitable habitats for this species. This is because the in *Avicennia marina* inhabits areas within tidal range. The mean annual temperature for *Avicennia marina* is 17-26°C (Louppe et al., 2008). This therefore suggests that *Avicennia marina* would likely adapt to higher temperatures of up to + 7°C in this subtropical region. This is probably because it attains climax growth under tropical conditions and a + 7°C rise in mean annual temperatures would convert the subtropical climate of north-eastern NSW to a tropical climate with higher mean annual temperatures. Furthermore, *Avicennia marina* would likely to redistribute southwards in north-eastern NSW, Australia with an increasing temperature. This is probably because of the currently existing tropical conditions northward in Queensland and a rise in temperature would likely produce unsuitable climatic conditions in the presently higher temperature conditions northwards in Queensland.

The dunal wetland plant species *Banksia integrifolia* (Coastal Banksia) would likely redistribute to suitable climate space covered by sand dunes (Figure 5.4). This is because it commonly grows on sandy coastal areas and it could reach a height of about 20m (Leiper et al., 2009). *Banksia integrifolia* is also found in regions with tropical conditions and is moderately drought and frost tolerant. This therefore suggests its high adaptability potential in an increasing mean annual temperature due to climate change.
The forested wetland species *Melaleuca quinquenervia* (broad-leaved paperback) would also likely to redistribute southward and inland with an increasing mean annual temperatures due to the more availability of suitable climate space (Figure 5.5). This species occurs in both subtropical and tropical climates and can therefore tolerate dry conditions. It grows to a tree height of about 25m in areas often covered with water (Leiper et al., 2009). It would therefore tolerate prolonged flooding and a fluctuating water table.
Figure 5.3: The simulated suitable climate space for *Avicennia marina* with increasing temperature.
Figure 5.4: The simulated suitable climate space for *Banksia integrifolia* with increasing temperature
Figure 5.5: The simulated suitable climate space for *Melaleuca quinquenervia* with increasing temperature
Figure 5.6: The simulated suitable climate space for *Leptospermum liversidgei* with increasing temperature.
Coastal swamp wetland species *Leptospermum liversidgei* (Swamp May) was also found to most likely redistribute southwards and westward from the coastline with an increasing mean annual temperature in north-eastern NSW, Australia (Figure 5.6). It is an erect twiggy shrub which grows to a height of about 3 meters (Leiper *et al.*, 2009). It also occurs in tropical environments and would, therefore, likely adapt to an increasing mean annual temperature of up to 6°C as a result of climate change.

The decrease pattern of suitable climate space for these species i.e. *Avicennia marina*, *Banksia integrifolia*, *Melaleuca quinquenervia* and *Leptospermum liversidgei* with a rising mean annual temperature is similar to the results found by Brereton *et al.*, (1995), Williams (2007) and Newell *et al.*, (2001). Even though the climate space of

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**Figure 5.7:** Area covered by suitable climate space of species with increasing mean annual temperature
these species decreased at higher temperature increase, the area of suitable climate space differed among species with *Banksia integrifolia* found to have the least area of climate space (28Km$^2$) at no projected change in mean annual temperature while *Leptospermum liversidgei* had the most projected area of about 1618Km$^2$ (Figure 5.7).

However, due to the fact that these wetland species would occur in tropical conditions, it therefore suggests that there would still be available suitable habitats (climate space) for these species by the end of the century with the projected 6.4°C rise in global mean annual temperature (IPCC, 2001) and the 4.8°C projected rise in temperature within the region (Hennessy *et al.*, 2004). Nonetheless, a rise in mean annual temperature beyond 7°C would likely results to a complete loss of suitable habitats for the wetland species in north-eastern NSW. The loss of suitable habitats however, may not unequivocally leads to species extinction but will certainly make these species extremely vulnerable.

Even though BIOCLIM is considered an important tool in conservation and biological surveys (Busby, 1991; Lindenmayer *et al.*, 1991), the significant limitation is that the predicted distributions may not accurately represent movement of highly mobile species. Furthermore, this modeling does not consider the water table levels that may also determine the suitable habitat of wetland species. In addition, warmer conditions may also open up these suitable habitats to colonisation by more tropical species that may further displace these wetland species’ ranges through competition. BIOCLIM was
however used in this study because it plays an important role in predicting the potential spatial distribution of species (Baker *et al.*, 2000; Clarke, 2003)

Although BIOCLIM has been criticized for not including the biotic interaction and the dispersal ability of species (Davis *et al.*, 1998), it is however a ‘first filter’ for identifying locations and wetland species that may be most at risk of climate change (Chilcott *et al.*, 2003).

**5.4. CONCLUSIONS**

Climate change, particularly changes in temperature can be expected, to have a profound impact on the spatial distribution of wetland species. This study used BIOCLIM climate modeling program to predict the potential changes of spatial distribution of four wetland species in the north-eastern NSW, Australia. The climate model assesses the climatic suitability of habitats under current and future climate scenarios. The results showed that temperature increase from the current level to the extremes estimated by the end of the century would likely re-distribute some of the wetland species such as *Avicennia marina* and *Melaleuca quinquenervia* southwards in north-eastern NSW. The results further suggested that suitable environments for some wetland species such as *Banksia integrifolia* would still be available by the end of the century at the predicted temperature ranges. It is important to note that even though the model showed the potential areas of spatial change of wetland species, the availability of the areas would
depend on the anthropogenic factors and other unpredictable natural phenomena. However, climate modeling is probably the only tool available for us to understand the uncertainty of potential climate variability and its impact on wetland change.
CHAPTER SIX

6.0. EXAMINING THE POTENTIAL IMPACTS OF SEA LEVEL RISE ON COASTAL WETLANDS IN NORTH-EASTERN NSW, AUSTRALIA

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Please see full article i.e. (Akumu et al., 2011) in Appendix 2.

I would like to acknowledge the three reviewers for their constructive comments.

Experiencing the potential impacts of sea level rise on coastal wetlands in north-eastern NSW, Australia

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Components of this chapter relating to examining the potential impacts of sea level rise were done in entirety by C.E. Akumu in partial fulfilment of his PhD. C.E. Akumu led the writing of the paper. S.Pathirana, S.Baban and D. Bucher reviewed the manuscript prior to submission to the Journal of Coastal Conservation. The relative contributions of the four authors to the manuscript are indicated below.

Conception of the study: CEA (50%), SP (50%)
Design of the study: CEA (50%), SP (50%)
Collection of data: CEA (95%), SP (5%)
Analysis of data: CEA (90%), SP (5%), SB (5%)
Interpretation of data: CEA (90%), SP (5%), DB (5%)
Conclusions: CEA (95%), SP (5%)
Writing up of paper: CEA (100%)

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6.1. INTRODUCTION

In chapter five, the potential spatial distribution due to climate change for the wetland species commonly found in mangroves and saltmarshes, coastal swamps, dunal and forested wetlands was predicted. It was found that a rise in mean annual temperature of beyond 7°C would likely result to a complete loss of suitable habitats for the wetland species i.e. *Avicennia marina*, *Banksia integrifolia*, *Melaleuca quinquenervia* and *Leptospermum liversidgei* in north-eastern NSW. However, in addition to the projected impacts of a rising temperature on the coastal wetlands, there is also the need to examine their vulnerability to sea level rise which is likely due to their close proximity to the ocean and low elevation. This chapter examines the potential impact of sea level rise on the coastal wetlands in north-eastern NSW.

Sea level rise is projected to have significant impacts on the earth’s coastal systems and low-lying areas globally, regionally and locally. The anthropogenic impacts on coastal wetland communities such as urbanization, residential development and agricultural development would therefore be exacerbated by a rise in sea level. The current and anticipated sea level rise (SLR) is mainly caused as a result of thermal expansion of the ocean and melting of land based ice sheets. According to Intergovernmental Panel on Climate Change (IPCC) Fourth Assessment report, the global sea level was stabilized at about 2500 years ago. There was no significant change in global sea level until the late 19th century where the onset of sea level rise began according to the instrumental record of modern sea level change (IPCC, 2007a).
The rising sea level would have severe impacts on coastal wetland communities in terms of inundation and erosion. This will generally lead to a higher coastal water level and an increase in salinity (IPCC, 2007b). The coastal wetland communities would be affected depending on the level of change in the coastal ecosystems. Some communities may adapt to sea level rise while others may migrate inland provided that there is an unblocked or undeveloped and suitable area inland for migration. Nevertheless, wetlands that accrete sediments such as mangroves and saltmarshes (Cahoon et al., 2000; Rogers et al., 2005) may keep pace with sea level rise as a result of sedimentation (Rogers et al., 2005). In contrast, if the accretion rate is lower relative to the rate of sea level rise, inundation would occur and mangroves may migrate into adjacent wetland communities such as saltmarshes. This phenomenon of landward migration of mangroves has already been recorded in south-east Australia (Saintilan & Williams, 1999). In Louisiana, sea level rise and salt water intrusion have been identified as factors in the decline of coastal bald cypress (*Taxodium disticum*) forest (Krauss et al., 2000; Melillo et al., 2000).

Globally, there have been some studies carried out on the potential impacts of sea level rise on coastal wetlands including the recent sea level rise impact modeling on coastal wetlands in the Jefferson county and the Chesapeake Bay region (Clough and Larson, 2009; Glick et al., 2008). They found an increase in wetlands such as saltmarshes and a decrease in inland fresh marshes by the year 2100.

In Australia, there have also been several studies on the impacts of sea level rise on coastal wetland communities (Eliot *et al.*, 1999; Ellison, 1994; Semeniuk, 1994;
Woodroffe, 1990). These studies found that wetlands such as mangroves are very vulnerable to inundation and erosion as they are mainly limited to current tidal zone. Furthermore, some specific threats to wetlands from sea level rise have been studied as part of a national vulnerability assessment (Waterman, 1996). An example is the Kakadu National Park in northern Australia where the projected impacts on the fresh water wetlands in the park showed that, the wetlands could become saline, given the current projections of sea level rise (Bayliss et al., 1997; Eliot et al., 1999). However, in the north-eastern subtropical region of NSW with high biodiversity, there is no known research on the potential impacts of sea level rise on the coastal wetland communities. There are gaps in our understanding as to how the wetland communities in the region will be affected by sea level rise. What wetland types are likely to be created or lost and by what extent? This chapter aims to predict the potential impacts of sea level rise on the coastal wetland communities in north-eastern NSW by the end of the century.

6.2. MATERIALS AND METHOD

6.2.1. Sea level Rise Projection

There is a variability of sea level rise estimates reported over the years. Global average sea level rise for 1990-2100 is projected to be about 88cm according to the IPCC (2001) Third Assessment Report (TAR). Warrick et al. (1996) in the Second Assessment Report (SAR) predicted a similar value of about 0.86m by the year 2100 based on the 1990 sea level. However, according to IPCC (2007a) Fourth Assessment Report (FAR),
global sea level rise is projected in the range of 18 – 59cm between 1980 to 1999 and 2090 to 2099. This last IPCC assessment report did not include rapid ice flow changes in its projected sea-level ranges, arguing that they could not yet be modeled, and consequently did not present an upper limit of the expected rise. Furthermore, the estimates assume that ice flow from Greenland and Antarctica will continue at the same rates as observed from 1993-2003. Nevertheless, according to the recent study by Vermeera & Rahmstorf (2009), the sea level is projected to rise of about 0.75m to 1.9m by the year 2100. However, due to variations in sea level rise estimates and the uncertainty on ice sheet discharge based on recent evidence of acceleration in Greenland and Antarctica, this study assumes a rise of about 1m by the year 2100.

6.2.2. Sea Level Rise Impact Model

Sea Level Affecting Marshes Model (SLAMM) was used to predict the potential impact of sea level rise on the coastal wetland communities in north-eastern NSW by the year 2100. SLAMM was developed by Richard A. Park with United States (US) Environmental protection Agency (EPA) funding (Park et al., 1986). This model simulates the dominant processes involved in wetland conversions and shoreline modifications during long-term rise in sea level (Park et al., 1989). The SLAMM model considers six primary processes that affect the fate of wetlands under different sea level rise scenarios. These include: Inundation, Erosion, Overwash, Saturation, Accretion and Salinity (Clough et al., 2010).
**Inundation:** The elevations of each cell are being reduced as the sea levels rise as a means to track the rise of water level and the salt boundary, thus keeping the mean tide level (MTL) constant at zero. The effects on each cell are calculated based on the minimum elevation and slope of that cell (Clough *et al.*, 2010).

**Erosion:** This model considers erosion to be triggered based on the threshold of maximum fetch and proximity of the wetland to estuarine water or open ocean (Clough *et al.*, 2010). A qualitative relationship between the maximum fetch and erosion has been developed (Clough *et al.*, 2010) which affects other portions of the model i.e.

- Max. Fetch less than 9km  
  Erosion = None
- Max Fetch > 9km to 20km  
  Erosion = Heavy
- Max Fetch greater than 20km  
  Erosion = Severe (affects dry-land)

**Overwash:** Overwash is assumed to occur in barrier islands of under 500 meter width during each 25 year time-step period as a result of storms. Beach migration and transport of sediments are also calculated (Clough *et al.*, 2010).

**Saturation:** This model assumes that fresh water marshes and coastal swamps can migrate onto adjacent uplands as a response of the water table to rising sea level close to the coast (Clough *et al.*, 2010).

**Accretion:** It considers the upward movement of marshes as a result of sequestration of sediments and biogenic production (Clough *et al.*, 2010). Accretion rate could be
specified on a spatially variable basis or be calculated using the model of accretion as a function of elevation, salinity and / or distance to channel i.e. equation 6.1 (Clough et al., 2010).

\[ A_{\text{cell}} = A_{\text{Elev}} \times (D \times S) \]  

(6.1)

Where:

\( A_{\text{cell}} \) = Predicted accretion rate for a cell (mm/year)

\( A_{\text{Elev}} \) = Accretion rate for a cell as a function of elevation alone

\( D \) = Factor representing distance to river or tidal channel (unitless)

\( S \) = Salinity factor representing salinity effects (unitless)

However, the accretion rate for the study was specified on a spatially variable basis within the Site file.

**Salinity**: This is optional and it considers that a location with a defined fresh-water flow, land categories can migrate based on salinity changes and based on a relatively simple salt wedge model (Clough et al., 2010). Nevertheless, this was not considered in north-eastern NSW because, multiple time-variable freshwater flow for each river and tributary were not available.

The methodological approach used (Figure 6.1) involved the use the wetland map generated using Landsat TM data acquired in February 2009 (see chapter four). Digital Elevation Model, and a site specific information file in order to predict the potential
impacts of sea level rise on the coastal wetlands. The Landsat TM acquired in February 2009 wetland classification was generated through preprocessing (geometric and radiometric corrections) and supervised classification (Akumu et al., 2010a). Accuracy assessment of the wetland classification which included mangroves and saltmarshes, dunal wetlands, coastal swamps, coastal upland water bodies, estuarine water bodies and forested wetlands was also performed (see chapter four). The February 2009 wetland map was later imported into ArcGIS 9.3 software and the wetland map was clipped with a 25m pixel size Digital Elevation Model (DEM) of the study area in order to obtain the same extent and cell size. The wetland map was later reclassified using the spatial analyst extension in ArcGIS in order to obtain the SLAMM wetland categories for the model. The SLAMM wetland categories included Nontidal swamps (i.e. forested wetlands), Mangroves and saltmarshes; Inland fresh marsh (i.e. dunal wetlands and coastal swamps), Inland open water (i.e. coastal upland water bodies) and estuarine open water (Table 6.1).
Figure 6.1: A schematic representation of the procedure used
Table 6.1: Reclassified February 2009 wetland classification into SLAMM wetland categories

<table>
<thead>
<tr>
<th>SLAMM categories</th>
<th>February 2009 wetland classification</th>
</tr>
</thead>
<tbody>
<tr>
<td>Nontidal swamps</td>
<td>Forested Wetlands</td>
</tr>
<tr>
<td>Mangroves and saltmarshes</td>
<td>Mangroves and saltmarshes</td>
</tr>
<tr>
<td>Inland fresh marsh</td>
<td>Dunal wetlands</td>
</tr>
<tr>
<td></td>
<td>Coastal swamps</td>
</tr>
<tr>
<td>Inland open water</td>
<td>Coastal upland water bodies</td>
</tr>
<tr>
<td>Estuarine open water</td>
<td>Estuarine water bodies</td>
</tr>
</tbody>
</table>

Nontidal swamps mostly consisted of trees with broad-leaved paperbark (*Melaleuca quinquenervia*) being the dominant plant species. The inland fresh marshes supported plant species such as *Banksia integrifolia, Boronia falcifolia, Bauera capitata and Leptospermum liversidgei*. Mangroves and saltmarshes were estuarine areas affected by tides and support mangrove and saltmarsh vegetation. The dominant mangrove species was *Avicennia marina* subsp. *australasica* (grey mangrove). Inland open waters were large or small fresh or brackish water bodies and included lakes and rivers. They may sometimes have submerged growths of sea tassel, ribbonweed and water milfoils. Estuarine open waters were large open saline or brackish water bodies with a relatively narrow permanent or intermittent connection to the sea.
Using the spatial analyst extension, the DEM was used to generate slope in degrees. The DEM, Slope, and the wetland map in SLAMM categories were converted from raster data format to ASCII Text format using the data conversion tool in ArcGIS. These were later used as input in SLAMM including a Site File of the study area using SLAMM version 6.0. The Site File includes variables such as historic trend of sea level rise, tide range, mean high water spring, marsh erosion rate, swamp erosion rate, tidal flat erosion, mangroves and salt marsh accretion rates, beach sedimentation rate and frequency of overwash.

The historic trend of sea level rise in the region was set to 1.18mm/yr according to Mitchell et al. (2000). The default marsh erosion rate, swamp erosion rate and tidal flat erosion rate i.e. 2.0 horz. m/yr, 1.0 horz. m/yr and 0.5 horz. m/yr respectively were used. The mean accretion rate for mangroves and saltmarshes in the region was 1.4±1.2 mm/yr (Saintilan et al., 2009). The beach sedimentation of the Sydney Harbour i.e. 0.8mm/yr (McLaughlin, 2000) was adopted due to the unavailability of beach sedimentation information in the study area. The frequency of overwash in the region was assumed to occur within a 25 year period. The output of the model was further converted to raster data format and analyzed in ArcGIS 9.3.

6.3. RESULTS AND DISCUSSION

The maps (Figures 6.2-6.6) showed the potential impact of sea level rise of about 1m on the coastal wetland communities in north-eastern NSW by the year 2100. The
study found an increase in wetlands such as mangroves and saltmarshes and a loss in wetlands such as inland fresh marshes by the year 2100 according to the simulation performed (Table 6.2). Furthermore, wetlands such as Inland fresh marshes and Nontidal swamps were projected to migrate onto adjacent uplands due to saturation based on the model (Figures 6.2-6.6).

According to the SLAMM model, the total area covered by Nontidal swamps in February 2009 reduced by about 16.4% by the year 2100 (Figure 6.7). This can be expected to be due to the intrusion of salt water by sea level rise thereby causing a loss in the fresh water communities. Furthermore, the model predicted a decrease in Inland fresh marshes of about 25.5% by the end of the century. Transitional marshes were estuarine intertidal wetlands with scrub-shrub plants including *Myoporum acuminatum*. There were no transitional marshes in February 2009 wetland classification because they consisted of mixed pixels and were not easily delineated using Landsat TM. However, they increased to about 250.51 km$^2$ by the year 2100 according to SLAMM model due to the fact that most of the dry land in the region fell at a very low elevation for inundation. However, there is some uncertainty in the model as to whether the dry land should be converted to transitional marsh, brackish marsh, or beach. If the dry land is within proximity to open ocean it converts to beach, otherwise transitional marsh. Brackish marsh is created only when the salinity module is utilized. The total area covered by mangroves and saltmarshes increased more than double by the end of the century relative to the area covered in February 2009 based on the simulation. This implies mangroves and salt marshes in north-eastern NSW can be expected to likely adapt and further
establish as a result of sea level rise. The projected increase in mangroves and saltmarshes, is similar to the findings of Glick et al. (2008), Clough and Larson (2009). Mangroves can be expected to intrude into saltmarshes (Saintilan & Williams, 1999) and this could further expand the area covered by mangroves and saltmarshes. Nonetheless, the intrusion of mangroves into saltmarshes is not generally considered to be as a result of sea level rise but also other factors such land use change in the catchment (Lovelock, et al., 2007). The adaptability of mangroves and saltmarshes to sea level rise could be enhanced by an increasing vertical elevation due to sedimentation. There were no tidal flats found until the year 2025 and this increased to about 80.79 km\(^2\) in 2050 and decreased by about 99\% by the year 2100 according to SLAMM projection. Tidal flats were mostly estuarine intertidal unconsolidated shore mud and the variation in tidal flats by the year 2100 according to the model can be due to sediments deposition as a result of to sea level rise and the establishment of tide tolerant wetland communities.

The total area covered by inland open waters reduced by about 73\% by the end of the century relative to the area covered in February 2009 based on the simulation. Estuarine open waters increased from about 35.89 km\(^2\) in February 2009 to about 161.34 km\(^2\) in 2100 according to the model. The increase in estuarine open waters based on the modeling can be expected to be due to sea water intrusion and inundation. This can alter the hydrological regimes in fresh and brackish waters such as the nature and variability of hydro-periods and the number and severity of extreme events (Baldwin et al., 2001; Burkett & Kusler, 2000; Sun et al., 2002).
Figure 6.2: Wetlands in SLAMM categories of February, 2009, north-eastern NSW-Australia
Figure 6.3: Wetlands in SLAMM categories by the year 2025 assuming 1m SLR by the end of the century in north-eastern NSW, Australia
Figure 6.4: Wetlands in SLAMM categories by the year 2050 assuming a 1m SLR by the end of the century in north-eastern NSW, Australia
Figure 6.5: Wetlands in SLAMM categories by the year 2075 assuming 1m SLR by the end of the century in north-eastern NSW, Australia
Figure 6.6: Wetlands in SLAMM categories by the year 2100 assuming 1m SLR by the end of the century in north-eastern NSW, Australia
Figure 6.7: Projected change in wetland area with time, based on SLAMM simulation
Table 6.2: Area of coastal wetlands in north-eastern, NSW by the year 2100 from SLAMM

<table>
<thead>
<tr>
<th>SLAMM 6 Categories</th>
<th>AREA COVERED (Km$^2$)</th>
<th>Feb. 2009</th>
<th>2025</th>
<th>2050</th>
<th>2075</th>
<th>2100</th>
</tr>
</thead>
<tbody>
<tr>
<td>Nontidal Swamps (Forested Wetlands)</td>
<td>149.26</td>
<td>145.44</td>
<td>140.09</td>
<td>134.54</td>
<td>124.72</td>
<td></td>
</tr>
<tr>
<td>Inland Fresh Marshes (Coastal Swamps &amp; Dunal Wetlands)</td>
<td>225.67</td>
<td>229.72</td>
<td>222.85</td>
<td>199.26</td>
<td>168.04</td>
<td></td>
</tr>
<tr>
<td>Transitional Marsh</td>
<td>0</td>
<td>81.84</td>
<td>125.64</td>
<td>197.68</td>
<td>250.51</td>
<td></td>
</tr>
<tr>
<td>Mangroves &amp; Saltmarshes</td>
<td>36.58</td>
<td>40.87</td>
<td>39.84</td>
<td>54.39</td>
<td>101.64</td>
<td></td>
</tr>
<tr>
<td>Tidal Flat</td>
<td>0</td>
<td>0</td>
<td>80.79</td>
<td>61.14</td>
<td>0.77</td>
<td></td>
</tr>
<tr>
<td>Inland Open Waters (Coastal upland water bodies)</td>
<td>32.76</td>
<td>20.48</td>
<td>14.44</td>
<td>10.96</td>
<td>8.84</td>
<td></td>
</tr>
<tr>
<td>Estuarine Open Water</td>
<td>35.89</td>
<td>54.47</td>
<td>68.98</td>
<td>96.12</td>
<td>161.34</td>
<td></td>
</tr>
</tbody>
</table>

Although the study found that a rise in sea level can be expected to significantly affect the wetlands in north-eastern NSW, there are some limitations in the sea level affecting marshes model. The model assumes a constant accretion rate over time for each habitat type whereas inundation by SLR would likely increase accretion by stimulating biomass productivity thereby allowing more sediments to be trapped by marsh vegetation.
(Morris *et al*., 2002). Furthermore, the model is not hydrodynamic because water is assumed to flow to all cells unless they are classified as ‘diked’. Besides, the output of the model is enhanced by the resolution of the DEM. In addition, the SLAMM model may be unrealistic because it does not consider anthropogenic factors to wetlands change. Also, there are differences between the marshes in the USA and Australian marshes which might affect the output of the model. These include differences in Australian wetland plant communities, the lack of barrier islands and the absence of overwash in Australian wetlands. Nevertheless, it provides a useful insight on the possible impacts of sea level rise on the coastal wetland communities which are already being threatened by rapid urbanization based on the numerous parameters such as elevation, slope and accretion rates considered by the model.

### 6.4. CONCLUSIONS

Climate change induced sea level rise could significantly affect the coastal wetland communities in north-eastern NSW, Australia. According to SLAMM, wetlands such as mangroves and saltmarshes increased in extent from about 36.58 km$^2$ in February 2009 to around 101.64 km$^2$ by the year 2100. Transitional marshes also increased to about 250.51 km$^2$ while estuarine open waters increased to around 161.34 km$^2$ by the end of the century. Nontidal swamps decreased from a total area of about 149.26 km$^2$ in February 2009 to about 124.72 km$^2$ by the end of the century. The decreasing pattern was similar with Inland fresh marshes and Inland open waters. There was a high variation in the extent of tidal flats with no tidal flats until the year 2025 from the simulation performed. The area covered by tidal flats also decreased from around 80.79 km$^2$ in 2050 to about 0.77 km$^2$ by the year 2100 based on the modeling.
There are however limitations in the sea level affecting marshes model and the model may also be limited because it does not consider the human impacts such as residential developments, urbanization, agricultural developments and land use change that could greatly influence coastal wetland ecosystems functioning and extent. Nonetheless, the output from the model could be used to enhance planning and management policies by local authorities.
CHAPTER SEVEN

7.0. MODELING METHANE EMISSION FROM COASTAL WETLANDS IN NORTH-EASTERN NSW, AUSTRALIA.

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Please see full article i.e. (Akumu et al., 2010b) in Appendix 3.

I would like to acknowledge the five reviewers for their constructive comments.

Modeling Methane Emission from Wetlands in North-Eastern New South Wales, Australia Using Landsat ETM+

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Components of this chapter relating to modeling methane emission from wetlands were done in entirety by C.E. Akumu in partial fulfilment of his PhD. C.E. Akumu led the writing of the paper. S. Pathirana, S. Baban and D. Bucher reviewed the manuscript prior to submission to Remote Sensing. The relative contributions of the four authors to the manuscript are indicated below.

Conception of the study: CEA (50%), SP (50%)
Design of the study: CEA (50%), SP (50%)
Collection of data: CEA (90%), SP (10%)
Analysis of data: CEA (90%), SP (5%), SB (5%)
Interpretation of data: CEA (90%), SP (5%), DB (5%)
Conclusions: CEA (95%), SP (5%)
Writing up of paper: CEA (100%)
7.1. INTRODUCTION

The modeling of the potential impacts of sea level rise on the coastal wetlands in north-eastern NSW was carried out in chapter six. The study found the coastal wetlands to be vulnerable to sea level rise with fresh water wetlands such as forested wetlands and coastal swamps more likely to decline spatially in extent. In addition to the vulnerability of the coastal wetlands to climate change, these wetlands contribute to greenhouse effect through the emission of gases such as methane (CH$_4$) to the atmosphere. It is therefore limited to quantify the potential impacts of climate change on these wetlands without examining their contribution to greenhouse effect. Therefore, there is the need to estimate methane emission from these wetlands in order to improve our understanding of their contribution to global methane budget. In this chapter, methane emission estimates from the coastal wetlands will be modelled using satellite data.

Natural wetlands are a major source of methane to the atmosphere, accounting for approximately 32±9.4% to the total methane emission (Hein et al., 1997; IPCC, 2007a). Among the existing greenhouse gases, methane is very important because it contributes approximately 17.7±2.5% to the net greenhouse effect (IPCC, 2001; Lelieveld et al., 1998). Methane is formed in from wetlands through the biological process called methanogenesis. Methanogenesis occurs in anaerobic environments in which organic matter undergoes decomposition by a specific group of bacteria (Liu, 1996). These anaerobic bacteria convert products such as carbon dioxide (CO$_2$), hydrogen (H$_2$), esters and salts of metanoic acid (HCOOH) generated by other micro-organisms into methane (Cicerone & Oremland, 1988). Methane production from wetlands is dependent on
climatic conditions such as temperature and soil moisture content that affect the metabolic activities of soil microbes. Soil moisture content also affects methane production directly by creating a low redox potential and anaerobic soil environment for methanogens (Liu, 1996; Xu et al., 2003). Methane production could also be attributed to substrate availability. High variability in measured methane fluxes from mangroves in Queensland, was attributed to dependence on the differences in substrate availability (Kreuzwieser et al., 2003).

There are different ways to estimate and monitor greenhouse gases such as methane in the atmosphere including experimental and remote sensing approaches. The remote sensing perspective, such as the use of Greenhouse gases Observing Satellite (GOSAT), launched January 23, 2009 produces relatively more accurate estimates of the flux of greenhouse gases on a subcontinental basis. This, therefore, reduces errors by half in identifying the greenhouse gases source and sinks at a subcontinental scale (Hamazaki, Kaneko, & Kuze, 2004). Furthermore, scaling techniques of greenhouse gases using optical sensing data such as Landsat and Advanced Very High Resolution Radiometer (AVHRR) have been very useful especially for greenhouse gas estimation in regional and global scale. Nevertheless, optical remote sensing data may tend to underestimate emission in tropical environments because of their limitation to cloud cover. Radar systems such as synthetic aperture radar (SAR) sensors on board several satellites (ERS-1, JERS-1, Envisat) have the potential to improve estimates of greenhouse gases across tropical environments because they can penetrate cloud and provide data day and night (Asner, 2001).
There have been some studies related to land-cover analysis/classification using satellite data (Baker, Lawrence, Montagne, & Patten, 2006; Lowry, Hess, & Rosenqvist, 2009; Lucas et al., 2009) and the measurements of methane fluxes from wetlands (Allen, et al., 2007; Bohn et al., 2007). They identified and classified land cover types such as mangroves, open water, agriculture, forest and urban. Furthermore, they found that accurate and easily reproducible land-cover maps using remote sensing products, which specifically enhance their extent and characteristics would improve monitoring and land management decisions. Methane fluxes emitted from wetlands were also found to differ significantly between sampling seasons and may increase with climate change by the end of the century (Allen et al., 2007; Bohn et al., 2007).

The quantitative estimation of greenhouse gases such as methane from natural sources using satellite data is very important. This is because natural emissions are closely connected to climate and ecological variables and this interaction constitutes potential feedbacks in the global system. Satellite remote sensing of wetland gaseous emission is very useful especially for large and inaccessible wetland types such as the floodplain wetlands in the Murray-Darling Basin, with a spatial extent of 5,826,600 hectares (Kingsford, 2010). This approach has been successfully used to estimate the amount of methane emission from wetlands in Siberia and India (Agarwal & Garg, 2007; Takeuchi et al., 2003). These studies estimated methane emission from wetlands such as forested bog, open bog, salt flats, mud flats and swamps. In Australia, especially in the north-eastern part of New South Wales (NSW), there has been no published research of methane estimation from wetlands using satellite data. There are gaps in our
understanding as to what the various wetlands contribute to global methane emission at both local and national scale.

The mean annual temperature is projected to increase by 0.2 to 1.6°C in coastal regions of NSW, Australia by the year 2030 (Hennessy et al., 2004). This would have severe impacts on ecosystem processes such as forests in tropical regions of Australia, which are highly sensitive to climate change within the range that is likely to occur in the next 50-100 years (Hilbert, Ostendorf, & Hopkins, 2001).

If temperature is a variable affecting methane production then quantifying the changes expected from a given temperature change can provide important information to the predictive models. This chapter aims to estimate the amount of methane emission from the coastal wetlands in north-eastern NSW and to estimate the amount of emission with temperature increase using Landsat ETM+.

7.2. MATERIALS AND METHODS

7.2.1. Satellite data

The spectral bands used to model methane emission from the wetlands in north-eastern NSW were the optical visible, near-infrared and thermal bands. Visible and Near-Infrared bands were used to extract wetland class information while the thermal band was used to extract land surface temperature which is an important determinant of methanogenesis. Landsat ETM+ data of June 2001 were used to estimate methane emission from the wetlands. This was due to the availability of cloud free image and also
because of its high spatial resolution in the thermal band i.e. 60 x 60 m compared to Landsat TM with a thermal band of 120 x 120 m. The spectral bands of Landsat ETM+ used (Table 7.1) were bands 1-4 and band 6.

Table 7.1: Spectral range and spatial resolution of Landsat ETM + used

<table>
<thead>
<tr>
<th>Bands</th>
<th>Spectral range (Microns)</th>
<th>Spatial resolution</th>
</tr>
</thead>
<tbody>
<tr>
<td>Band1</td>
<td>0.45-0.52</td>
<td>30 × 30 m</td>
</tr>
<tr>
<td>Band2</td>
<td>0.52-0.60</td>
<td>30 × 30 m</td>
</tr>
<tr>
<td>Band3</td>
<td>0.63-0.69</td>
<td>30 × 30 m</td>
</tr>
<tr>
<td>Band4</td>
<td>0.76-0.90</td>
<td>30 × 30 m</td>
</tr>
<tr>
<td>Band6</td>
<td>10.40-12.50</td>
<td>60 × 60 m</td>
</tr>
</tbody>
</table>

The methodology (Figure 7.1) involved the use of the optical visible and near-infrared bands of Landsat ETM+ to classify the wetlands and to generate Normalized Difference Vegetation Index (NDVI) for the wetland classes (see chapter four). The NDVI of the vegetation was used to estimate emissivity for the wetland classes. The emissivity of the wetland classes was needed in order to correct the spectral emissivity of the blackbody temperature generated. This is because NDVI values of vegetation have been found to be strongly correlated with thermal emissivity (Van de Griend & Owe, 1993). Pre-processing (e.g. geometric and radiometric corrections) and supervised classification of the Landsat ETM+ data (see chapter four) were performed and the wetland classes exported to geographic information system (GIS) for further analysis of area covered by each class. Accuracy assessment of the classified wetland map was carried out using orthophoto, Google Earth and groundtruthing. The classified wetland map was further reclassified using the spatial analyst tool in ArcGIS in order to assign the
mean emissivity values for each wetland type which was further used to correct the spectral emissivity of the blackbody temperature generated.
Figure 7.1: Schematic representation of the methodology
Band 6 was pre-processed and used to generate land surface temperature (LST) for the wetland classes. Climate data from the Bureau of Meteorology was used to validate the generated LST and to obtain precipitation and evaporation information for the study area.

Observed methane fluxes for the wetland classes were obtained by field and lab experiments. The productivity factor of methane emission from the wetland classes were estimated using net primary productivity values from literature. The area of the wetland classes, observed methane flux, productivity factor, temperature dependent factor (T-factor), precipitation and evaporation ratio were used to model methane emission estimates from wetlands in north-eastern NSW Australia.

7.2.2. Methane Emission Estimation

Methane production process from the wetland soil is dependent on soil moisture, wetland extent, vegetation type, water table depth and temperature (Burrows et al., 2010). Higher temperatures alone would increase methane production in saturated areas but would also cause these saturated areas to shrink resulting in a net methane reduction while higher precipitation alone would raise the soil moisture, water table, and expands the wetland extent thereby resulting to a net increase in methane. However, an exception to these relationships is for small increase in temperature (1°C) and precipitation (5%), for which the temperature increase can cause a slight increase in methane emission (Bohn et al., 2007). Furthermore, the vegetation type in an ecosystem would influence the net primary productivity (NPP) of the ecosystem and this is proportional to methane
production (Sheppard et al., 1982). The estimation of methane production from the soil in this study assumes the methane emission processes depend linearly on temperature. This was carried out using equation 7.1 modified from Agarwal and Garg (2007) and Liu (1996).

\[ E_{CH_4} = E_{obs} \cdot F_t \cdot A \cdot P \cdot fw \]  
(7.1)

Where:

\( E_{CH_4} \) = Estimated Methane Emission

\( A \) = Area of wetland classes

\( F_t \) = Temperature dependent factor (T factor)

\( E_{obs} \) = Observed methane flux from different wetland classes,

\( P \) = Productivity factor

\( fw = P/E \) = Precipitation /Evaporation ratio, Where: \( fw \left\{ \begin{array}{l} P/E \text{ if } P \leq E \\ 1 \text{ if } P > E \end{array} \right\}\)

7.2.2.1. Estimating Area of methane emitting wetlands

The area of methane emitting wetlands was estimated from classified wetland map produced using Landsat ETM+ satellite images (five scenes) acquired on June 21st 2001. This was performed in ER Mapper software 7.1 using the following image processing steps: a) Georeferencing b) Subsetting c) Masking d) Converting DN to Radiance e) Converting Radiance to Reflectance and f) Classification (See chapter four).

Accuracy assessment of the wetland classification was performed by comparing the classified map with scenes from Google Earth, color orthophoto and groundtruthing.
of the region. 312 samples were selected from the classified map through random sampling method from the wetland classes and compared with the referenced maps. Furthermore, groundtruthing was carried out in order to validate the wetland classes. It was performed by randomly selecting the samples in the field and using of a global positioning system (GPS) as a guide to identify the wetland classes. A k × k confusion matrix (error matrix) table was then developed and the producer accuracy, user accuracy, overall accuracy and kappa analysis were calculated. The classified wetland map was later exported to ArcGIS 9.3 and their respective areas generated. The methane emitting wetland types were identified from the classification and these included: mangroves and saltmarshes, forested wetlands, dunal wetlands, estuarine water bodies, coastal upland water bodies and coastal swamps.

Normalize Difference Vegetation Index (NDVI) of the reflectance image was also generated using band 3 (Visible Red) and 4 (Near-infrared) of the Landsat ETM+ image (see chapter four). The NDVI was generated for use in the estimation of thermal emissivity of the wetland classification. The NDVI values were obtained for the following wetland classes: mangroves and saltmarshes, forested wetlands, dunal wetlands and coastal swamps with overlapping NDVI values between classes. The observed mean NDVI values and standard error (SE) were 0.665±0.044 for mangroves and saltmarshes, 0.685±0.054 for forested wetlands, 0.669±0.067 for dunal wetlands, and 0.565±0.095 for coastal swamps.
7.2.2.2. Estimating temperature dependent factor (T factor)

The T factor was derived from equation 7.2 (Liu, 1996).

\[ \text{Tt} = \frac{F(Ts)}{F(Ts)} \quad \text{equation 7.2} \]

Where:

\[ F(Ts) = e^{0.334(Ts-23)} \quad \text{equation 7.3} \]

Ts = Land surface temperature in Degrees Celsius

\[ F(Ts) = \text{Mean of F(Ts) over wetlands. It was derived from the F(Ts) image. The F(Ts) image was converted from raster to vector format in ArcGIS 9.3 and their mean values estimated.} \]

The mean F (Ts) value over the wetlands and standard error was 0.04±0.01.

7.2.2.2.1. Estimating Land surface Temperature (Ts)

In this study, land surface temperature (LST) is an important input factor in the methane modelling process. It was derived from Landsat ETM+ using the thermal infrared band (10.40-12.50 μm), with a spatial resolution of 60m. The satellite images were georeferenced using an orthophoto and re-projected into the UTM-WGS 84 projection. The images were later subsetted to the study area and a DEM was used to
mask out all areas that are more than 20 m altitude from the coastline. The following processing steps were further carried out on the Landsat ETM + band 6 satellite data:

Conversion of digital number (DN) to Spectral Radiance

The DN of the thermal infrared band (band 6) was converted to Radiance using equation 7.4 (L.P.S.O., 2002).

\[ L_\lambda = 0.0370588 \times DN + 3.2 \] \hspace{1cm} (7.4)

Where:

\( L_\lambda \) = radiance, W/m\(^2\)/sr/\( \mu \)m

\( DN \) = digital number

Conversion of Spectral Radiance to At-Satellite Brightness Temperature (Blackbody temperature)

The spectral radiance was converted to at-satellite brightness temperature using equation 7.5 assuming uniform emissivity (L.P.S.O., 2002).

\[ T_B = \frac{K_n}{\ln\left(\frac{K_n}{L_\lambda} + 1\right)} \] \hspace{1cm} (7.5)

Where:

\( T_B \) = at-satellite temperature in Kelvin

\( L_\lambda \) = spectral radiance W/m\(^2\)/sr/\( \mu \)m
K₂ and K₁ are pre-launch calibration constants. For Landsat-7 ETM+,

K₂ = 1282.71 K
K₁ = 666.09 mW cm⁻² sr⁻¹ μm⁻¹

However, the temperature obtained (T_B) is with reference to a black body (perfect absorber and emitter of radiation) assuming a uniform emissivity. There is therefore a need to correct the spectral emissivity with respect to the land cover. Emissivity is the ratio between emittance from an object in relation to emittance from a blackbody at the same temperature (Campbell, 2006). The spectral emissivity for the various wetland types were derived using their NDVI values (Van de Griend & Owe, 1993). According to Van de Griend and Owe (1993), when the NDVI value of natural land surface is between 0.157 and ~ 0.727, then emissivity may be acquired approximately from NDVI index of the vegetation using equation 7.6 (Van de Griend & Owe, 1993).

\[ \varepsilon = 1.0094 + 0.047 \ln (\text{NDVI}) \]  
\[ \text{(7.6)} \]

Where:
\[ \varepsilon = \text{emissivity} \]
\[ \text{NDVI} = \text{Normalized Difference Vegetation Index} \]

A constant emissivity value of 0.986 for water (estuarine water bodies and coastal upland water bodies) was adopted from the emissivity classification scheme by Snyder et al. (1998).

The mean emissivity values for the wetland classes were as follows: mangroves and saltmarshes = 0.990±0.004, dunal wetlands = 0.991±0.007, forested wetlands =
0.992±0.004, coastal swamps = 0.983±0.009. This is, however, limited to wetland vegetation and would not be applicable to bare soil and water with NDVI values of less than 0.157. Furthermore, a change in the quality of the wetland vegetation would also change the thermal emissivity values of the wetland.

The wetland map was reclassified using these values and was used to correct the spectral emissivity of the blackbody temperatures (TB).

The emissivity corrected land surface temperatures (St) were then computed using equation 7.7 (Dinh et al., 2006).

\[
St = \frac{TB}{1 + (\lambda \times TB / \rho) \ln \varepsilon} \quad (7.7)
\]

Where:

St = emissivity corrected land surface temperatures (LST) in Kelvin

\( \lambda \) = wavelength of emitted radiance,

\( \rho = h \times c / \sigma \) (1.438769x10^{-2} m K, second radiation constant), \( \sigma = \) Boltzmann constant (1.3806503x10^{-23} J/K), \( h = \) Planck’s constant (6.626068x10^{-34} J s), \( c = \) velocity of light (2.99792x10^8 m/s).

TB = at-satellite temperature in Kelvin

\( \varepsilon = \) emissivity of wetland classes
The unit (Kelvin) of the emissivity corrected LST was converted to degree Celsius using equation 7.8.

\[
I_1 - 273.15 = \frac{1}{equation\ 7.8}
\]

Where:

\[I_1 = St\]

Land surface temperature values (Table 7.2) were then estimated for the different wetland classes. These values were validated from climate data obtained from the Bureau of Meteorology. Furthermore, according to the IPCC (2007a) report, the mean annual temperature will rise by 0.2 to 1.6 °C by the year 2030. This study assumes a 1°C rise in mean annual temperature by the year 2030 and this value was added to the LST generated.

The following estimated T factor values (Table 7.3) for the different wetland classes were then generated based on the estimated land surface temperatures. Furthermore, a projected T factor assuming a 1°C rise in mean annual temperature by the year 2030 was also generated using the projected LST. This was further used to predict methane emission estimates with 1°C rise in mean annual temperature assuming every other variable is kept constant.

7.2.2.3. Methane Flux

The methane flux values for the wetland classes (Table 7.4) were obtained from field and lab experiments. Static gas chambers with a diameter of 30cm and a height of
44cm and 0.07m$^2$ surface area from which the bases had been removed were used to trap methane flux from the wetlands. The top of the chambers were gas tight sealed with a lid and a rubber stopper and were gently pushed into the soil surface to a depth of 14cm leaving an air space volume at the top of 21,214cm$^3$. Chambers used for estuarine waters and coastal upland bodies were attached to 100mm PVC pipes and floated in the estuary and lake. Four PVC pipes with a length of 1.5m were attached at the joints using a 4 × 90° bends and the static gas chamber was attached in the centre of the PVC pipes using a plywood and cable ties. This was left to float in the estuary waters and coastal upland water bodies with a string attached. The chambers were sampled four times over a period of 30 minutes with an interval of 10 minutes and duplicate samples were collected at each time. The sampling was carried out by using a 25ml syringe and needle to collect 25ml of air from the chamber and stored in a non-sterilized exetainers. The gas samples were analyzed within a period of at most two days using a gas chromatograph equipped with a flame ionization detector and an electron capture detector (Wang & Wang, 2003). This was carried out in autumn with a total number of four points in the field for each wetland type and a total number of 16 samples for analysis for each wetland type. The geographic positions for the wetlands (Figure 7.2) were recorded with a global positioning system (GPS) within the study area. Samples from estuarine waters, mangroves and saltmarshes were collected in Tweed Heads. Dunal wetland samples were collected in Pottsville, coastal swamps in Lennox head, forested wetlands in Ballina and coastal upland water bodies from Lake Ainsworth. The total daily methane flux for each wetland type were directly computed by linear interpolation based on the assumption that the flux measured is a representative of the daily mean for each wetland type.
Figure 7.2: Geographic locations of wetland types for methane gas samples
7.2.2.4. Productivity Factor

The productivity factor was a ratio of net primary productivity for each wetland ecosystem to net primary productivity of tropical rainforest (equation 7.9) assuming that methane production in a tropical rainforest is constant throughout the year because of relatively constant temperatures and rainfall (Sheppard et al., 1982).

\[ R = \frac{\text{NPP (ecosystem)}}{\text{NPP (tropical rainforest)}} \]  \hspace{1cm} (7.9)

Where:

\( R \) = Productivity factor

\( \text{NPP} \) = Net primary productivity

Mean productivity factors for the wetland ecosystems (Table 7.5) were calculated using their net primary productivity values for the wetland classes adopted from (Cronk & Fennessy, 2001; Sheppard et al., 1982). The net primary productivity of the wetland types were obtained from the dry weight of above ground biomass.

7.2.2.5. Precipitation and evaporation Ratio

Precipitation and evaporation values were obtained from the climate statistics in the region. This study assumes water saturation in the soil as a function of precipitation and evaporation ratio. A higher precipitation relative to evaporation is assumed that the
soils are water saturated for a given period and a value of 1 is assigned for every grid cell.

The mean precipitation and standard error (SE) in the month of June 2001 for the study area was 166.2±26.7mm while the mean evaporation and standard error for the month of June 2001 was 125±35.4mm. The precipitation and evaporation ratio was therefore set to 1 for every grid cell.
7.3. RESULTS AND DISCUSSION

The maximum observed mean NDVI value of 0.685±0.054 was for forested wetlands while coastal swamps had the minimum mean NDVI value of 0.565±0.095 (Figure 7.3). This implied the forested wetland vegetation were greener than the coastal swamps vegetation cover in June 2001.

Figure 7.3: NDVI values of wetland vegetation in north-eastern NSW, Australia
The land surface temperature values (Figure 7.4) in the winter month of June 2001 were found to be higher in water than on land. The higher land surface temperature values found in water than on land in the winter month of June 2001 is probably due to the fact that in winter, water would absorb solar energy and hold onto heat longer than land. The possibility of using land surface temperature in estimating water table depth is an area of further research.

Figure 7.4: Estimated Land surface temperature (LST)
Table 7.2: Estimated mean LST and projected LST for the wetland classes in the winter month of June including standard error of the mean (SEM)

| Wetland classes              | LST (°C)  | T°C from BOM | Projected LST (°C) assuming  
<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th>1°C rise in mean annual temperature by the year 2030</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mangroves and saltmarshes</td>
<td>12.23±0.64</td>
<td>12.98±0.56</td>
<td>13.23±0.64</td>
</tr>
<tr>
<td>Forested wetlands</td>
<td>10.78±0.51</td>
<td>10.56±0.61</td>
<td>11.78±0.51</td>
</tr>
<tr>
<td>Coastal swamps</td>
<td>11.77±0.78</td>
<td>11.11±0.72</td>
<td>12.77±0.78</td>
</tr>
<tr>
<td>Estuarine water bodies</td>
<td>14.13±0.72</td>
<td>15.23±0.69</td>
<td>15.13±0.72</td>
</tr>
<tr>
<td>Coastal upland water bodies</td>
<td>12.85±0.79</td>
<td>12.12±0.70</td>
<td>13.85±0.79</td>
</tr>
<tr>
<td>Dunal wetlands</td>
<td>12.22±0.75</td>
<td>13.31±0.54</td>
<td>13.22±0.75</td>
</tr>
</tbody>
</table>

The maximum T factor values (Figure 7.5) were found in estuarine water bodies while forested wetlands had the minimum T factor values. The higher T factor values found in estuarine water bodies was due to the higher temperatures found in estuarine waters at the time of satellite image acquisition. There would, however, be uncertainties in the T factor values due to the fluctuation in land surface temperatures within days, months, seasons and years.
Figure 7.5: Temperature dependent factor (T factor) for the wetland classes
Table 7.3: Temperature dependent factor (T factor) and standard errors for the wetland classes

<table>
<thead>
<tr>
<th>Wetland</th>
<th>T factor</th>
<th>Projected T factor-assuming 1°C rise in mean annual temperature by the year 2030</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mangroves and saltmarshes</td>
<td>0.69±0.15</td>
<td>0.96±0.20</td>
</tr>
<tr>
<td>Forested wetlands</td>
<td>0.45±0.08</td>
<td>0.64±0.12</td>
</tr>
<tr>
<td>Coastal swamps</td>
<td>0.57±0.16</td>
<td>0.80±0.22</td>
</tr>
<tr>
<td>Estuarine water bodies</td>
<td>1.03±0.25</td>
<td>1.44±0.34</td>
</tr>
<tr>
<td>Coastal upland water bodies</td>
<td>0.71±0.19</td>
<td>0.99±0.26</td>
</tr>
<tr>
<td>Dunal wetlands</td>
<td>0.70±0.18</td>
<td>0.98±0.25</td>
</tr>
</tbody>
</table>

Forested wetlands had the highest daily mean methane flux of 1.029±0.01 g/m²/day while coastal upland water bodies had the least mean daily methane flux of 0.015±0.004 g/m²/day. The variability of methane flux was high in the wetlands, especially in the forested wetlands. The uncertainties in methane flux were probably due to the changing environmental conditions such as soil temperature, rate of methane oxidation in the oxic soil between water table and soil surface and the difference in substrate availability. An increase in environmental conditions such as temperature and soil moisture content would increase methane flux from wetlands and vice versa (Wang & Wang, 2003). Methane fluxes from wetlands would furthermore be affected by seasonal conditions. This is because there are different seasonal conditions of variables such as rainfall, temperature, evaporation and vegetation types thereby increasing the variability of methane flux in a year.
Table 7.4: Estimated mean methane fluxes for the wetland classes

<table>
<thead>
<tr>
<th>Wetland class</th>
<th>Mean Flux ±SE (g/m²/day)</th>
<th>Mean Flux ±SE (g/m²/month)</th>
<th>Number of gas samples</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mangroves and saltmarshes</td>
<td>0.016±0.01</td>
<td>0.496±0.32</td>
<td>16</td>
</tr>
<tr>
<td>Forested wetlands</td>
<td>1.029±0.01</td>
<td>31.286±2.97</td>
<td>16</td>
</tr>
<tr>
<td>Coastal swamps</td>
<td>0.161±0.05</td>
<td>4.893±1.44</td>
<td>16</td>
</tr>
<tr>
<td>Estuarine water bodies</td>
<td>0.022±0.0001</td>
<td>0.683±0.004</td>
<td>16</td>
</tr>
<tr>
<td>Coastal upland water bodies</td>
<td>0.015±0.004</td>
<td>0.461±0.13</td>
<td>16</td>
</tr>
<tr>
<td>Dunal wetlands</td>
<td>0.037±0.02</td>
<td>1.123±0.54</td>
<td>16</td>
</tr>
</tbody>
</table>

The productivity factor (Figure 7.6) was highest in mangroves and saltmarshes while coastal upland water bodies had the lowest productivity factor. There were also variations in the productivity factor of the wetlands for methane emission. This is because it is dependent on the net primary productivity of the wetlands which further depends on the variables such as vegetation, hydrology, climate, soil type and nutrients availability (Cronk & Fennessy, 2001).
**Figure 7.6:** Productivity factor of methane emission for the wetland classes

**Table 7.5:** Mean productivity factors and standard error for the wetland classes

<table>
<thead>
<tr>
<th>Wetland class</th>
<th>Mean Productivity factor and SE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mangroves and saltmarshes</td>
<td>1.00±0.00</td>
</tr>
<tr>
<td>Forested wetlands</td>
<td>0.73±0.04</td>
</tr>
<tr>
<td>Coastal swamps</td>
<td>0.75±0.70</td>
</tr>
<tr>
<td>Estuarine water bodies</td>
<td>0.95±0.07</td>
</tr>
<tr>
<td>Coastal upland water bodies</td>
<td>0.25±0.00</td>
</tr>
<tr>
<td>Dunal wetlands</td>
<td>0.39±0.13</td>
</tr>
</tbody>
</table>
The thermal emissivity of the wetlands was used to correct the spectral emissivity of the blackbody temperature generated. The maximum mean thermal emissivity value (0.992±0.004) was for forested wetlands while coastal swamps had the minimum thermal emissivity value of 0.983±0.009 (Figure 7.7). The high thermal emissivity value for forested wetlands was due to their high NDVI values. There would however be variation of the thermal emissivity values for the wetlands especially within seasons. This is because the NDVI values are likely to change with seasons due to changes in environmental conditions and the resulting foliage replacement and flowering cycles of the plants.
Figure 7.7: Thermal emissivity values for the wetland classification
The wetland areas used in the methane emission model were generated from the wetland classification (Figure 7.8).

**Figure 7.8:** Wetland Classification in north-eastern NSW, Australia- Landsat ETM+, June 2001 (Source: chapter four).
The accuracy assessment carried out for the wetland classification (Table 7.6) showed a maximum user's accuracy (96.15%) for mangroves and saltmarshes and the minimum user's accuracy (78.00%) for forested wetlands. The kappa analysis from the classification showed a high correlation between the wetland classification and the reference data used.

**Table 7.6: Error Matrix Table for Landsat TM + 2001 wetland classification**

<table>
<thead>
<tr>
<th>Reference data</th>
<th>F</th>
<th>M</th>
<th>D</th>
<th>S</th>
<th>L</th>
<th>E</th>
<th>Row Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>F</td>
<td>39</td>
<td>0</td>
<td>8</td>
<td>3</td>
<td>0</td>
<td>0</td>
<td>50</td>
</tr>
<tr>
<td>M</td>
<td>0</td>
<td>50</td>
<td>0</td>
<td>2</td>
<td>0</td>
<td>0</td>
<td>52</td>
</tr>
<tr>
<td>D</td>
<td>2</td>
<td>0</td>
<td>46</td>
<td>3</td>
<td>0</td>
<td>0</td>
<td>51</td>
</tr>
<tr>
<td>S</td>
<td>8</td>
<td>1</td>
<td>0</td>
<td>44</td>
<td>0</td>
<td>0</td>
<td>53</td>
</tr>
<tr>
<td>L</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>43</td>
<td>7</td>
<td>50</td>
</tr>
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<td>E</td>
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<td>0</td>
<td>0</td>
<td>0</td>
<td>5</td>
<td>51</td>
<td>56</td>
</tr>
<tr>
<td>Column Total</td>
<td>49</td>
<td>51</td>
<td>54</td>
<td>52</td>
<td>48</td>
<td>58</td>
<td>312</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Remote sensing data</th>
<th>Producer’s Accuracy</th>
<th>User’s Accuracy</th>
<th>Overall Accuracy</th>
<th>Overall Kappa Statistics</th>
</tr>
</thead>
<tbody>
<tr>
<td>D = Dunal wetlands</td>
<td>85.19%</td>
<td>90.20%</td>
<td></td>
<td></td>
</tr>
<tr>
<td>F = Forested wetlands</td>
<td>79.59%</td>
<td>78.00%</td>
<td></td>
<td></td>
</tr>
<tr>
<td>S = Coastal swamps</td>
<td>84.62%</td>
<td>83.02%</td>
<td></td>
<td></td>
</tr>
<tr>
<td>L = Coastal upland water bodies</td>
<td>89.58%</td>
<td>86.00%</td>
<td></td>
<td></td>
</tr>
<tr>
<td>M = Mangroves and saltmarshes</td>
<td>98.03%</td>
<td>96.15%</td>
<td></td>
<td></td>
</tr>
<tr>
<td>E = Estuarine water bodies</td>
<td>87.93%</td>
<td>91.07%</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

|                  |                     |                 | 87.50%          | 85.00%                |

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The study also found forested wetlands to have the highest amount of methane emission (0.0016±0.00009 Tg) in the month of June, 2001 while coastal upland water bodies had the lowest amount of methane emission (0.0000019±0.0000005 Tg) in the month of June, 2001 (Table 7.7). According to the IPCC (2007a) report, the mean annual temperature will rise by 0.2 to 1.6 °C by the year 2030. In line with the IPCC (2007a) projection, an estimation of methane emission from the various wetlands by the year 2030, assuming 1°C rise in mean annual temperature was projected to increase in the month of June (Table 7.7).

**Table 7.7:** Methane emitting areas and emission estimates in the month of June 2001 including mean annual temperature increase by 1°C in north-eastern NSW, Australia

<table>
<thead>
<tr>
<th>Wetland type</th>
<th>Area covered by wetland (km²)</th>
<th>Methane emission estimate in June (Tg) ± SE</th>
<th>Methane emission estimate (Tg) in June assuming a 1°C rise in mean annual temperature ±SE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mangroves and saltmarshes</td>
<td>36.56</td>
<td>0.000013±0.000006</td>
<td>0.000018±0.000008</td>
</tr>
<tr>
<td>Forested wetlands</td>
<td>152.09</td>
<td>0.0016±0.00009</td>
<td>0.0022±0.0001</td>
</tr>
<tr>
<td>Coastal upland water bodies</td>
<td>32.74</td>
<td>0.000019±0.000005</td>
<td>0.000037±0.000007</td>
</tr>
<tr>
<td>Estuarine water bodies</td>
<td>35.97</td>
<td>0.000024±0.000001</td>
<td>0.000034±0.000001</td>
</tr>
<tr>
<td>Coastal swamps</td>
<td>150.56</td>
<td>0.00031±0.00002</td>
<td>0.00044±0.00001</td>
</tr>
<tr>
<td>Dunal wetlands</td>
<td>73.37</td>
<td>0.000022±0.000008</td>
<td>0.000031±0.00001</td>
</tr>
<tr>
<td>Total</td>
<td>481.29</td>
<td>0.0019±0.0001</td>
<td>0.0027±0.0002</td>
</tr>
</tbody>
</table>
The areas covered by the wetlands have contributed significantly to the estimated amount of methane emission. Forested wetlands had the maximum amount of methane emission due to a large area and methane flux. The estimation of area covered by the wetlands was however limited to the satellite data used and the environmental conditions of the wetlands at the time of image acquisitions. This is because the spatial resolution of the satellite data would influence the area covered by the wetlands and conditions such as floods and tides would also influence the wetland area. High tides would likely affect the area covered by estuarine water bodies, mangroves and saltmarshes due to their proximity to the sea. The estimated amount of methane emission from the wetlands classes was also influenced by their methane fluxes, which were obtained from field and lab experiments.

The methane fluxes observed from mangroves and estuarine water bodies in this study were lower than the calculated mean methane fluxes of 7.38 mg/m²/hr for Pichavaram mangrove and 15.41 mg/m²/hr for Adyar estuary in south India (Purvaja & Ramesh 2001). The low methane fluxes observed in mangroves sediments maybe a result of out-competition for substrates (Lovely & Klug, 1983) by sulphate reducers. This is because in marine environments with high salinity, methanogenesis may be inhibited by sulphate reducing bacteria (Abram & Nedwell, 1978; Verma, Subramanian, & Ramesh 2002). Forested wetlands generally had a higher methane flux compared to the estimated 23.5±11.3 g/m²/yr for forested wetland ecosystems calculated globally in Sheppard et al. (1982).
The study assumed that methane emission processes from wetlands depend linearly on temperature due to the fact that methanogenic activities are influenced by soil temperature. It further assumed soil moisture content based on the ratio of precipitation and evaporation within a region, which is, however, limited to the availability of climate data. In addition, it assumed the mean methane fluxes measured from the various wetlands as a representative of the daily mean for each wetland type. This would nonetheless change with changing environmental conditions. However, in order to estimate annual methane emission from the wetlands it is imperative to model methane fluxes for the various seasons. This is because methane emission is expected to change throughout the year due to changes in parameters such as precipitation, and temperatures during winter, summer, autumn and spring. During summer with high temperatures, methane emission is likely to increase but this could be limited by a high rate of evaporation thereby reducing the area covered by wetlands. Methane emission would increase in autumn due to high rainfall, thereby increasing wetland area and soil moisture content. During spring and winter, methane emission from wetlands would decrease due to less rainfall and lower temperatures in the north-eastern region of NSW, Australia. Nevertheless, there is a further limitation in the estimation of annual methane emission from wetlands using satellite data. This includes the lesser availability of cloud free satellite images on a monthly basis which is necessary in order to acquire wetlands information such as area cover and quality for methane estimation.

Climate change (temperature increase) would probably increase natural methane emission from global wetlands. Policies geared towards carbon sequestration and storage
could offset climate change and its impact to methane emission. The mitigation of methane emission from wetlands to the atmosphere is an area for further research. Remote sensing could play a vital role in identifying and monitoring the wetlands, and the acquisition of biophysical properties which could be used to improve our understanding of methane emission over time.

7.4. CONCLUSIONS

An estimation of methane emission from wetlands has been carried out using Landsat ETM+. This was modeled using the following parameters: productivity factor, temperature dependent factor (T-factor), wetland area, methane flux, precipitation and evaporation ratio. The estimation of methane emission from the wetlands has been carried out assuming methane production is linearly dependent on temperature. The study found a high variability of methane emission from the wetlands with the maximum amount of methane emission from forested wetlands and the minimum amount of emission from coastal upland water bodies. Methane emission is anticipated to increase with climate change which is most likely due to an increase in metabolic activities of the soil microbes. There are however uncertainties in the methane emission estimation due to changes in methane fluxes and environmental conditions such as temperature and rainfall over time from the wetlands. There are also limitations in the use of satellite data to estimate methane emission from wetlands such as the limitation in acquiring cloud free satellite images especially on a monthly basis and the factor of scale (spatial resolution) in delineating the wetland classes. Nevertheless, the empirical methane emission model
using Landsat ETM+ is suitable to estimate monthly and yearly methane budget from wetlands at both local and regional level. In the estimation of methane emission from wetlands, it is important to consider the changing pattern of climatic variables such as temperature and precipitation as these greatly influence methanogenesis.
CHAPTER EIGHT

8.0. DISCUSSION AND CONCLUSIONS

Chapters five, six and seven have projected the potential impacts of climate change and sea level rise on the coastal wetlands in north-eastern NSW, Australia. Nonetheless, these possible impacts could be curbed by the implementation of effective management, strategies by authorities charged with the management of the wetlands. This chapter would discuss the national and state wetland management policies in Australia. Furthermore, it would examine the current wetland management practices in the study area and will provide possible management, mitigation and adaptation strategies to limit the projected impacts. In addition, it would integrate and synthesize the significant findings of the research, limitations, and areas for further study.

8.1. POTENTIAL IMPACTS OF CLIMATE CHANGE ON THE COASTAL WETLANDS

This study has found that the suitable habitats for the coastal wetland species *Avicennia marina, Banksia integrifolia, Melaleuca quinquenervia* and *Leptospermum liversidgei* would decrease with a higher mean annual temperature. However, from the simulation performed, there would still be suitable climate space for these species with a 7°C rise in mean annual temperature. Nonetheless, a rise in temperature beyond 7°C could lead to a complete loss of suitable climate space for these wetland plant species.
Furthermore, the study found that a 1m rise in sea level can be expected to lead to the loss of some fresh water wetlands while wetlands such as mangroves and salt marshes can likely increase. This would occur provided that human shoreline modification works allow for inland migration of these wetlands (particularly saltmarshes, which are likely to become ‘squeezed’ between a modified and unyielding terrestrial boundary and an expanding mangrove habitat). Some new wetlands such as transitional marshes and tidal flats can likely be created in the region according to the SLAMM model by the end of the century. In addition, methane emission from these wetlands would likely increase with a rising temperature and forested wetlands would account for most of the emission. According to the methane emission modeling performed, forested wetlands emitted about $0.0016\pm0.00009$ Tg of methane in the month of June 2001 in the region. This would likely increase to about $0.0022\pm0.0001$ Tg in the month of June with a 1°C rise in global mean annual temperature.

8.2. WETLAND MANAGEMENT POLICIES

The Ramsar convention held on the 2\textsuperscript{nd} of February, 1971 in Iran developed guidelines for the sustainable use of wetlands, including the preparation of wetland policies by countries that are parties to the wetland convention. Based on the convention, the Commonwealth Government of Australia developed its own wetlands policy (EA, 1997) with the purpose to build wetland conservation, within the broader context of environmental management, into the daily business of the Commonwealth Government.
The main goal of the wetlands Policy of the Commonwealth Government of Australia is to conserve, repair and manage wetlands sustainably. In order to achieve this goal, the Commonwealth Government will strive, in cooperation with State/Territory and local Governments and the Australian people, to:

1) conserve Australia’s wetlands particularly through the promotion of their ecological, cultural, economic and social values. This therefore implies conservation of the coastal wetlands identified and mapped in this study is necessary in order to promote their socioeconomic values. This is possible provided action is taken to prevent the loss of these wetlands by land use change.

2) To manage wetlands in an ecologically sustainable way and within a framework of integrated catchment management. The implementation of this policy will be suitable for the coastal wetlands in north-eastern NSW. However, based on the research findings, these wetlands could be properly managed within a framework of coastal zone management rather than integrated catchment management.

3) To achieve informed community and private sector participation in the management of wetlands through appropriate mechanisms. This policy approach would be suitable to enhance the conservation and restoration of the coastal wetlands in north-eastern NSW. This could be achieved by investing through the open call for proposals in wetland projects.
4) To raise community and visitor awareness of the values, benefits and range of types of wetlands. This policy is suitable in the region and could limit the incursion of residential development into wetland areas.

5) To develop a shared vision between all spheres of government and promote the application of best practice in relation to wetland management and conservation. This policy will depend on the political will of each state and territory government in promoting wetland management as a priority.

6) To ensure a sound scientific and technological basis for the conservation, repair and ecologically sustainable development of wetlands. This implies the coastal wetlands in north-eastern NSW would be properly monitored and managed and their sensitivity to climate change could be easily detected.

7) Meet Australia’s commitments, as a signatory to relevant international treaties, in relation to the management of wetlands. This policy will also depend on the political will of the Federal government in committing to international obligations towards wetlands conservation.

In addition to the national wetland policy, some states and territories in Australia have developed wetland management policies including the state of New South Wales. The NSW Wetlands Management Policy was developed in 1996 with the purpose of “ecologically sustainable use, management and conservation of wetlands in NSW for the
benefit of present and future generations” (DLWC, 1996). The intended outcomes of policy enactment are to:

a) halt, and where possible, reverse:

- loss of wetland vegetation;
- declining water quality;
- declining natural productivity;
- loss of biological diversity; and
- declining natural flood mitigation.

Based on the findings in this study that some of the coastal wetlands in north-eastern NSW could be lost due to sea level rise and temperature increase, to achieve the above outcomes will require not only wetland management policy but an integrated policy with climate change, mitigation and adaptation.

b) To encourage projects and activities which will restore the quality of the State’s wetlands, such as:

- rehabilitating wetlands;
- re-establishing vegetation buffer zones around wetlands; and
- ensuring adequate water to restore wetland habitats.

These outcomes could be achieved provided the wetland area lost has not been occupied by residential or urban development. Based on the findings of this study, forested wetlands decreased from 1989 to 2009 probably due to urban development thereby limiting the rehabilitation of forested wetlands in some areas in region.
In order to achieve the NSW Wetland Management Policy goal, the policy adopts the following nine principles for the sustainable management of wetlands:

1) Water regimes needed to maintain or restore the physical, chemical and biological processes of wetlands will have formal recognition in water allocation and management plans. This implies the maintenance of coastal wetlands such as estuarine waters, mangroves and saltmarshes that are directly influenced by water regimes in the region of study will be enhanced.

2) Land use and management practices that maintain or rehabilitate wetland habitats and processes will be encouraged. This implies the continuous loss in wetland extent e.g. forested wetlands based on this research findings would be minimized.

3) New developments will require allowance for suitable water distribution to and from wetlands. This therefore suggest the use of buffer zones around the coastal wetlands identified in this study to accommodate for suitable water distribution to and from the wetlands.

4) Water entering natural wetlands will be of sufficient quality so as not to degrade the wetlands. This implies wetlands such as coastal upland water bodies and coastal swamps identified in this study would be prevented from toxic substances and eutrophication.
5) The construction of purpose-built wetlands on the site of viable natural ones will be discouraged. Based on this research findings that the coastal wetlands in north-eastern NSW would require nearby suitable space for migration with climate change, the avoidance of purpose-built wetlands on the sites of natural ones could create suitable wetland habitats for migration.

6) Natural wetlands should not be destroyed, but when social or economic imperatives require it, the rehabilitation or construction of a wetland should be required. This could lead to the expansion of existing wetland types found in the coastal region of north-eastern NSW.

7) Degraded wetlands and their habitats and processes will be actively rehabilitated as far as is practical. Based on this study, the quality of coastal swamps was lower relative to the other wetland types. This therefore implies active rehabilitation is required in the coastal swamps of north-eastern NSW relative to the other wetland types identified.

8) Wetlands of regional or national significance will be conserved. This principle justifies the need to conserve the coastal wetlands identified, mapped and monitored in this study i.e. mangroves and saltmarshes, forested wetlands, coastal swamps, dunal wetlands, estuarine water bodies and coastal upland water bodies.

9) The adoption of a stewardship ethos and co-operative action between land and water owners and managers, government authorities, non-government agencies, and the general community is necessary for effective wetland management. This implies the management
of the coastal wetlands in north-eastern NSW by a variety of stakeholders including land and water owners would be suitable to improve the effectiveness of the conservation process.

Neither the national nor state policies recognise the natural dynamics of wetlands nor the consequences of future environmental change, natural or human-induced. Management for future sustainability must include the maintenance of buffers and areas where wetlands may persist or where new wetlands may establish. Wetlands likely to persist under future environmental changes may be a higher management priority than wetlands where the future conditions cannot sustain them.
8.3. CURRENT WETLAND MANAGEMENT STRATEGIES IN NORTH-EASTERN NEW SOUTH WALES

There is no independent body charged with the management of the coastal wetlands in north-eastern NSW. The responsibility of management depends on the land tenure. Management is performed by local government, the national parks and wildlife services (if the wetland is within a formal nature reserve), the Land and Property Management Authority (management of crown land), WetlandCare i.e. a not-for-profit company or private land owners. In all cases, any development likely to impact on waterways will need state government departmental permission such as the Environmental Protection Agency (EPA).

The current management strategies commonly carried out by some local authorities such as the Byron shire council (PB, 2009) in the region include:

- Protection and buffering of existing habitats from development.
- Restoration of degraded habitats and linking of isolated or fragmented habitats with wildlife corridors.
- Control and reduction of threatening processes.
- Establishment of ongoing collaboration between different authorities and community groups involved in wetland management issues.
- Ensure the impacts of natural processes and hazards are given a high priority in the planning and management of coastal wetlands.
At this local level of management, there is recognition of future changes. This suggests that availability of high-quality GIS, satellite data and local expertise to make the best use of these resources is a high priority. Also, there is a need for regional and national coordination of consistent operational protocols.

8.4. MANAGEMENT, MITIGATION AND ADAPTATION STRATEGIES

The potentially suitable habitats for the wetlands species at higher temperatures (beyond 5°C) could be protected and included in planning by local authorities charged with the management of wetlands. This could be carried out using buffers of about 100m (Buchsbaum, 1994; Castelle, Johnson, & Canolly, 1994) on the potentially suitable habitats (Figures 8.1-8.4). This buffer size would generally be sufficient to protect these habitats. This is because a minimum buffer width of 15m would provide the basic physical and chemical buffering while a minimum buffer width of 30m would provide the maintenance of the biological components of wetlands in most circumstances (Castelle et al., 1994). However, there is no single universal width that can provide all the desired benefits of a buffer. Choosing a particular buffer width will generally depend on the conservation significance and the purpose of the buffer. Furthermore, as a management strategy, these suitable habitats could be classified as high priority areas for the conservation of these species.
Figure 8.1: Buffered suitable habitats for Banksia integrifolia

Figure 8.2: Buffered suitable habitats for Melaleuca quinquenervia

Figure 8.3: Buffered suitable habitats for Avicennia marina

Figure 8.4: Buffered suitable habitats for Leptospermum liversidgei

Legend
- Coastal upland water bodies
- Estuarine water bodies
- Suitable Habitat
- 100m Buffer

Scale: 0 5 10 20 kilometers
The potential impacts of sea level rise could be managed by identifying non-wetland areas that are currently at risk of 1m sea level rise (i.e. areas likely to be inundated and/or areas where new wetlands are likely to be created). The non-wetland areas at risk could also be protected and included in planning by local authorities. They include upland-developed dryland such as residential areas and upland-undeveloped dryland including agricultural areas. The identification and a 100 m buffering of the non-wetland areas currently at risk of 1m sea level rise in north-eastern NSW has been performed using the overlay and buffer functions in ArcGIS 9.3 (Figure 8.5). These non-wetland areas should be considered in coastal planning by policy makers.
Figure 8.5: Non-wetland areas at risk to 1m sea level rise in north-eastern NSW
Furthermore, methane emission from these wetlands would likely increase with a rising temperature. The reduction and avoidance of greenhouse gases such as methane from wetlands could help to mitigate climate change and thereby its impacts on methanogenesis. The mitigation strategies of climate change include:

**Carbon Sequestration**- This could be achieved through the afforestation, reforestation, agro forestry, Silviculture and promoting increased carbon stocks in biomass. This would be suitable to carry out in north-eastern NSW. The existing NSW policy on wetland rehabilitation could enhance afforestation of wetlands such as forested wetlands if properly implemented thereby sequestrating carbon.

**Carbon Conservation**- This could be carried out through the conservation of biomass and soil carbon in protected areas, improved forest and wetland management practices (e.g. reduced-impact logging), fire protection and more effective use of prescribed burning in forest and agricultural systems. This would also be suitable to carry out in the coastal region of study. Carbon conservation could be enhanced by the implementation of the existing NSW wetland management principle on conserving wetlands of regional or national significance.

**Carbon Substitution** - This includes the sustainable use of biofuels and the use of alternative sources of energy such as solar and wind energy rather than biofuels. This is a proposed policy that could be adopted in the state of NSW. This will encourage research on the use of alternative sources of energy and thereby reducing greenhouse gases to the
atmosphere. This can be expected to halt or reduce global warming thereby reducing methane emission found from coastal wetlands in this study.

Greenhouse gas reduction or avoidance - This could be achieved through biodigestion and energy related projects such as Emissions Trading Schemes (ETS). This is also a proposed policy that would be suitable to adopt in the state of NSW. This entails the pricing of carbon thereby discouraging the emission of carbon dioxide to the atmosphere by major polluters in the region.

In addition to mitigation, adaptation options would also provide a framework that would enhance the coastal wetland ecosystems' resilience to cope with changing climatic conditions. These include:

- Maintaining the natural biodiversity of the coastal wetland ecosystems in north-eastern NSW in order to reduce their vulnerability.
- Increasing vegetation cover or improving soil infiltration of the wetlands in order to increase water storage in catchments.
- Creating corridors for wildlife inhabiting wetlands.
- Preventing the effects of climate change through legislation, regulatory and policy measures.
- Retreat from the high risk areas of inundation caused by climate change induced sea level rise.
- Protect the inundated areas from sea level rise by building dykes and levees.
• Accommodate the impacts of climate change by introducing high resilience wetlands species.

• Development of more risk-based climate change impact assessment research.

However, in order to enhance the conservation of the coastal wetlands, it is necessary to integrate mitigation and adaptation strategies (Figure 8.6). This is because there is a synergy between many of the adaptation options and mitigation (Dang et al., 2003). The promotion of synergy between mitigation and adaptation will also advance sustainable development, since mitigation activities could contribute to reducing the vulnerability of natural ecosystems and socioeconomic systems (Ravindranath, 2007).
Figure 8.6: The Schematic linkages between mitigation and adaptation strategies of climate change and their possible scale of impacts on a wetland ecosystem.
8.5. SUMMARY AND CONCLUSIONS

There is growing evidence that the current and projected climate change is influenced by anthropogenic factors such as the emission of greenhouse gases to the atmosphere. There are predictions of increasing global surface temperature caused by the greenhouse effect and a rising mean sea level cause primarily by the melting land-based ice sheets and thermal expansion (IPCC, 2001, 2007a; Mitchell, 1989). This would have severe impacts on the earth’s ecosystems such as wetlands with numerous functions including the source of primary productivity. There is therefore the need to examine the potential impacts of climate change on the earth’s ecosystems in order to enhance their conservation. This study has assessed the potential impacts of climate change and sea level rise on the coastal wetlands in north-eastern New South Wales.

The coastal wetlands in the study area were located around latitudes 28°09′S and 29°06′S, and longitude 153°00′E. They extended from Tweed Heads to Evans Head within 20m elevation from the coastline of north-eastern NSW. They occurred in a subtropical environment with high biodiversity including plant species such as Argyrodendron spp., Araucaria cunninghamii, Flindersia australis, Ceratopetalum apetalum, Doryphora sassafras, Avicennia marina, and Banksia integrifolia (Morand, 2001). The objectives of the study were to: (1) provide an overview of wetland classifications in relation to their plant communities and their environmental variables with emphasis in NSW; (2) map the current and past wetland communities in north-eastern NSW, in order to identify any changes in quality and extent; (3) predict the
potential spatial distribution of selected wetland species (*Avicennia marina, Banksia integrifolia, Melaleuca quinquenervia* and *Leptospermum liversidgei*) as a result of climate change i.e. mean annual temperature increase; (4) predict the potential impact of sea level rise on the coastal wetland communities by the end of the century; (5) estimate the amount of methane emission from the coastal wetlands using satellite data and their emission with a temperature increase; and (6) provide management, mitigation and adaptation strategies that could be used to minimize the impacts of climate change on the coastal wetland ecosystems.

An overview of wetland classifications has been discussed and these included classification schemes developed by Cowardin *et al.* (1979), the Ramsar wetland classification, and the NSW wetland classification. The widely-used wetland classification scheme applicable to NSW has been modified to include artificial wetlands. The coastal wetlands in north-eastern NSW have been mapped and monitored using time series Landsat satellite data sets acquired on the 27th of February 2009, 16th of September 1989 and the 21st of June 2001. Bioclimatic modeling has been used to predict the potential spatial distribution of *Avicennia marina, Banksia integrifolia, Melaleuca quinquenervia* and *Leptospermum liversidgei* as a result of climate change i.e. mean annual temperature increase. The potential impact of sea level rise on the coastal wetland communities by the end of the century has been modeled using the sea level affecting marshes model. A process- based modeling technique using variables such as temperature dependent factor, observed methane flux and productivity has been used to estimate the amount of methane emission from the coastal wetlands and their emission with a
temperature increase. Furthermore, the possible mitigation and adaptation strategies that could be used to minimize the devastating impacts of climate change have been discussed.

This study has classified, mapped, and monitored the following wetland types: mangroves and saltmarshes, coastal upland water bodies, estuarine water bodies, forested wetlands, dunal wetlands and coastal swamps. It found a significant change in the quality of the wetland vegetation at the time of satellite image acquisition. The range of mean NDVI values was 0.610-0.670, 0.565-0.685 and 0.335-0.470 in the months of February 2009, June 2001 and September 1989 respectively. NDVI values vary between -1 and +1 and the wetland vegetation is greener when NDVI value is closer to 1. This is because live green plants absorb solar radiation in the photosynthetically active radiation (PAR) spectral region (red band), which they use in the process of photosynthesis to manufacture food. In contrast, live green plants also reflect and transmit solar radiation in the near-infrared spectral region (NIR band) because a strong absorption at these wavelengths would only result in over-heating the plant and possibly damaging the tissues since as the energy level per photon in this spectral region is not sufficient to be useful in photosynthesis. NDVI is a ratio of red and near-infrared spectral bands and is therefore a measure of plant greenness. The low rainfall value i.e. about 13.7mm in the month of September 1989 probably accounted to the low quality of the wetland vegetation in the September 1989 scenes. This is because NDVI apparently correlates with rainfall (Rowland et al., 1996). There was a significant change in the extent of some wetlands communities in the months of September 1989, June 2001 and February 2009.
The total area covered by forested wetlands in the month of September 1989 was about 162.01 Km\(^2\). This area decreased to about 152.09 Km\(^2\) and 149.26 Km\(^2\) in the months of June 2001 and February 2009 respectively. The continuous decrease in forested wetlands could be due to anthropogenic factors such as residential development, agricultural expansion and urban development. In addition to wetland change, this study also found that the representative plant species of the wetland types i.e. *Avicennia marina*, *Banksia integrifolia*, *Melaleuca quinquenervia* and *Leptospermum liversidgei* would likely redistribute southwards in north-eastern NSW with temperature increase. Also, there would still be available suitable climate space for these species to redistribute with a projected mean annual temperature increase of about 6.4°C (IPCC, 2001). This is probably because these wetland species also occur in tropical environments and would tolerate higher temperatures. Nevertheless, it was found that an increase in mean annual temperature beyond 7°C would likely lead to a complete loss of suitable habitats for these wetland species. Furthermore, this study found that climate change induced sea level rise could have a significant impact on the coastal wetland communities. According to the SLAMM modeling output, a meter rise in sea level by year 2100 could more than double the area covered by mangroves and saltmarshes relative to the area covered in February 2009. This implies mangroves and saltmarshes in north-eastern NSW could likely adapt and further establish as a result of sea level rise. In contrast, the area covered by inland fresh marshes (dunal wetland and coastal swamps) reduced by about 25.5% by the end of the century relative to the area covered in February 2009. This could be due to the intrusion of salt water by sea level rise thereby causing a loss in the fresh water communities. Furthermore, inland fresh marshes and nontidal swamps (forested
wetlands) were seen to migrate onto adjacent uplands due to saturation while new wetlands including transitional marshes were created according to the model. This study has also found significant variation in methane emission from the coastal wetlands. The highest daily methane flux i.e. 1.029±0.01 g/m²/day was found in forested wetlands while coastal upland water bodies had the lowest mean daily methane flux of 0.015±0.004 g/m²/day. The variability in methane flux was probably due to changing environmental conditions such as soil temperature, rate of methane oxidation in the oxic soil between water table and soil surface and the difference in substrate availability. Forested wetlands also had the highest amount of methane emission (0.0016±0.00009 Tg) in the month of June 2001 due to its large surface area. In contrast, coastal upland water bodies had the lowest amount of methane emission (0.0000019±0.0000005 Tg) in the month of June, 2001. It was found that a 1°C rise in mean annual temperature would increase the amount of methane emission from the coastal wetlands. This is probably due to an increase in metabolic activities of soil microbes and thereby an increase in methanogenesis.

Even though the potential impacts of climate change on coastal wetlands in north-eastern NSW has been successfully assessed using the geoinformatics approach, there however limitations in the modeling and assessment performed. The environmental information generated from the satellite data such as the wetland classification is limited to the factor of scale of the sensor. Furthermore, the species distribution modeling performed may not be suitable to highly mobile wetland species. The potential impacts of sea level rise on the coastal wetland communities may also not be realistic due to the numerous limitations in the SLAMM model including the assumption of a constant...
accretion rate. In addition, the estimation of methane from the wetlands is limited by the variability of methane flux which is influenced by daily, seasonal, and annual environmental conditions. Nonetheless, the findings of this research could be used in wetland management and planning of climate change impacts in order to enhance the conservation of these delicate and sensitive wetland ecosystems. Furthermore, in order to curb the potential impacts of climate change on the coastal wetlands in north-eastern NSW, it is necessary to implement effective management, mitigation and adaptation strategies such as buffering, carbon sequestration and storage, and by introducing high resilience wetland species or genotypes that could easily adapt in a changing climate. In addition, it is important to consider the natural succession of species from present-day warmer climates into the suitable area, which will undoubtedly provide a refuge for such species as they too are forced southwards as temperatures rise.

Future potential research areas should incorporate the modeling and analysis of seasonal variations of methane emission from wetlands over the region. The estimation of methane emission from wetlands at both state and national level is also an area of further study. Furthermore, the modeling and estimation of other greenhouse gases such as nitrous oxide from wetlands in the region is recommended. In addition, further climate change impact study on other coastal wetlands along NSW i.e. from Evans Head to Sydney would enhance our understandings of the processes and the extent of environmental change likely to occur in the coastal zone of NSW, Australia. Finally, the mapping and monitoring of wetlands at both state and national level using satellite data is
recommended. Satellite data will provide useful information about the temporal extent of wetland change at both state and national level in order to enhance conservation.
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APPENDIX 1

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APPENDIX 3
Modeling Methane Emission from Wetlands in North-Eastern New South Wales, Australia Using Landsat ETM+

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Abstract: Natural wetlands constitute a major source of methane emission to the atmosphere, accounting for approximately 32 ± 9.4% of the total methane emission. Estimation of methane emission from wetlands at both local and national scale using process-based models would improve our understanding of their contribution to global methane emission. The aim of the study is to estimate the amount of methane emission from the coastal wetlands in north-eastern New South Wales (NSW), Australia, using Landsat ETM+ and to estimate emission with a temperature increase. Supervised wetland classification was performed using the Maximum Likelihood Standard algorithm. The temperature dependent factor was obtained through land surface temperature (LST) estimation algorithms. Measurements of methane fluxes from the wetlands were performed using static chamber techniques and gas chromatography. A process-based methane emission model, which included productivity factor, wetland area, methane flux, precipitation and evaporation ratio, was used to estimate the amount of methane emission from the wetlands. Geographic information system (GIS) provided the framework for analysis. The variability of methane emission from the wetlands was high, with forested wetlands found to produce the highest amount of methane, i.e., 0.0016 ± 0.00009 teragrams (Tg) in the month of June, 2001. This would increase to 0.0022 ± 0.0001 Tg in the month of June with a 1 °C rise in mean annual temperature by the year 2030 in north-eastern NSW, Australia.
Keywords: methane; wetlands; emission estimation; satellite data

1. Introduction

Wetlands are areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water, the depth of which at low tide does not exceed six meters [1]. Wetlands are important because they are a source of primary productivity; they provide habitats to wildlife; enhance water quality and provide an arena for recreation [2]. There is a need for monitoring wetlands because it allows us to determine whether these ecosystems have changed over time in terms of size, extent and quality. Furthermore, their reliance on rainfall, surface runoff, groundwater levels and evaporation rates make them, and the ecological services they provide, vulnerable to even small climatic changes.

Natural wetlands are a major source of methane (CH$_4$) to the atmosphere, accounting for approximately 32 ± 9.4% to the total methane emission [3,4]. Among the existing greenhouse gases, methane is very important because it contributes approximately 17.7 ± 2.5% to the net greenhouse effect [5,6].

There are different ways to estimate and monitor greenhouse gases such as methane in the atmosphere including experimental and remote sensing approaches. The remote sensing perspective, such as the use of Greenhouse gases Observing Satellite (GOSAT) launched January 23, 2009, produces relatively more accurate estimates of the flux of greenhouse gases on a subcontinental basis. This, therefore, reduces errors by half in identifying the greenhouse gases source and sinks at a subcontinental scale [7]. Furthermore, scaling techniques of greenhouse gases using optical sensing data such as Landsat and Advanced Very High Resolution Radiometer (AVHRR) have been very useful especially for greenhouse gas estimation in regional and global scale. Nevertheless, optical remote sensing data may tend to underestimate emission in tropical environments because of their limitation to cloud cover. Radar systems such as synthetic aperture radar (SAR) sensors on board several satellites (ERS-1, JERS-1, Envisat) have the potential to improve estimates of greenhouse gases across tropical environments because they can penetrate cloud and provide data day and night [8].

There have been some studies related to land-cover analysis/classification using satellite data [9-11] and the measurements of methane fluxes from wetlands [12,13]. They identified and classified land cover types such as mangroves, open water, agriculture, forest and urban. Furthermore, they found that accurate and easily reproducible land-cover maps using remote sensing products, which specifically enhance their extent and characteristics, would improve monitoring and land management decisions. Methane fluxes emitted from wetlands were also found to differ significantly between sampling seasons and may increase with climate change by the end of the century [12,13].

Methane production from wetlands is dependent on climatic conditions such as temperature and soil moisture content that affect the metabolic activities of soil microbes. Soil moisture content also affects methane production directly by creating a low redox potential and anaerobic soil environment for methanogens [14,15]. Methane production could also be attributed to substrate availability. High
variability in measured methane fluxes from mangroves in Queensland was attributed to dependence on the differences in substrate availability [16].

The quantitative estimation of greenhouse gases such as methane from natural sources using satellite data is very important. This is because natural emissions are closely connected to climate and ecological variables and this interaction constitutes potential feedbacks in the global system. Satellite remote sensing of wetland gaseous emission is very useful especially for large and inaccessible wetland types such as the floodplain wetlands in the Murray-Darling Basin, with a spatial extent of 5,826,600 hectares [17]. This approach has been successfully used to estimate the amount of methane emission from wetlands in Siberia and India [18,19]. These studies estimated methane emission from wetlands such as forested bog, open bog, salt flats, mud flats and swamps. In Australia, especially in the north-eastern part of New South Wales (NSW), there has been no published research of methane estimation from wetlands using satellite data. There are gaps in our understanding as to what the various wetlands contribute to global methane emission at both local and national scale.

The mean annual temperature is projected to increase by 0.2 to 1.6 °C in coastal regions of NSW, Australia by the year 2030 [20]. This would have severe impacts on ecosystem processes such as forests in tropical regions of Australia, which are highly sensitive to climate change within the range that is likely to occur in the next 50–100 years [21].

If temperature is a variable affecting methane production then quantifying the changes expected from a given temperature change can provide important information to the predictive models. This study aims to estimate the amount of methane emission from the coastal wetlands in north-eastern NSW and to estimate the amount of emission with temperature increase using Landsat ETM+.

2. Materials and Methods

2.1. Study Area

The study area extends from Evans Head to Tweed Heads within 20 m elevation from the coastline of north-eastern NSW (Figure 1). It is a low land area bounded by latitudes 28°09′S and 29°06′S, and longitude 153°00′E Most of the area lies on the Mesozoic sediments of the Clearance-Moreton Basin. It consists of metamorphic rocks and the sediments have been overlain by tertiary volcanics of the Mt. Warning shield volcano. There are outcrops of volcanics in the region consisting of alkaline basalt, andesite and andesitic breccia [22]. Large deposits of Quaternary sand occur along the coast. These consist of marine and Aeolian quartz sands that have formed beaches, dunes and sandsheets. Older Pleistocene dune systems lie inland of the younger Holocene beaches, fore dunes and hind-dunes. The sand masses have been subjected to extensive mining. Estuarine muds and clays occur within creeks and lagoons and peats are present in swamps formed in swales and deflation depressions [22-24]. The region is subtropical with mean annual temperature of 20 °C and a mean annual precipitation range of 1,482 mm to 17,560 mm [25].

The wetland types found in this study area include mangroves and saltmarshes, forested wetlands, coastal upland water bodies, dunal wetlands, estuarine water bodies and coastal swamps. Mangroves and salt marshes were estuarine areas, which are subjected to tidal flooding and support mangrove and salt marsh vegetation [26]. Forested wetlands were dominated by trees and occurred on fertile soils, mostly at low altitude. The dominant forested vegetation in the area was broad-leaved paperbark
(Melaleuca quinquenervia). Coastal upland water bodies were large or small fresh or brackish water bodies along the coast and included lakes, rivers and ponds. Estuarine water bodies were large open saline or brackish water bodies with a relatively narrow permanent or intermittent connection to the sea. Coastal swamps were fresh water wetlands around the coast that consisted of shallow marshes, wet heaths, and meadows vegetated by sedges and aquatic herbs. Dunal wetlands were fresh water wetlands on coastal sand dunes or plains that support woodland, heathland, sedges, and rushes.

**Figure 1.** Study Area: north-eastern NSW Australia.

2.2. Satellite Data

The spectral bands used to model methane emission from the wetlands in north-eastern NSW were the optical visible, near-infrared and thermal bands. Visible and Near-Infrared bands were used to extract wetland class information while the thermal band was used to extract land surface temperature, which is an important determinant of methanogenesis. Landsat ETM+ data of June 2001 were used to estimate methane emission from the wetlands. This was due to the availability of cloud free images and also because of its high spatial resolution in the thermal band i.e., 60 × 60 m, compared to Landsat TM with a thermal band of 120 × 120 m. The spectral bands of Landsat ETM+ used (Table 1) were bands 1–4 and band 6.

**Table 1.** Spectral range and spatial resolution of Landsat ETM+ used.

<table>
<thead>
<tr>
<th>Bands</th>
<th>Spectral range (Microns)</th>
<th>Spatial resolution</th>
</tr>
</thead>
<tbody>
<tr>
<td>Band 1</td>
<td>0.45–0.52</td>
<td>30 × 30 m</td>
</tr>
<tr>
<td>Band 2</td>
<td>0.52–0.60</td>
<td>30 × 30 m</td>
</tr>
<tr>
<td>Band 3</td>
<td>0.63–0.69</td>
<td>30 × 30 m</td>
</tr>
<tr>
<td>Band 4</td>
<td>0.76–0.90</td>
<td>30 × 30 m</td>
</tr>
<tr>
<td>Band 6</td>
<td>10.40–12.50</td>
<td>60 × 60 m</td>
</tr>
</tbody>
</table>
The methodology (Figure 2) involved the use Landsat ETM+ satellite data to classify the wetlands and to generate Normalized Difference Vegetation Index (NDVI) for the wetland classes. The NDVI of the vegetation was used to estimate emissivity for the wetland classes. The emissivity of the wetland classes was needed in order to correct the spectral emissivity of the blackbody temperature generated. This is because NDVI values of vegetation have been found to be strongly correlated with thermal emissivity [27]. Pre-processing (e.g., geometric and radiometric corrections) and supervised classification of the Landsat ETM+ data were performed and the wetland classes exported to geographic information system (GIS) for further analysis of area covered by each class. Accuracy assessment of the classified wetland map was carried out using orthophoto, Google Earth and groundtruthing. The classified wetland map was further reclassified using the spatial analyst tool in ArcGIS in order to assign the mean emissivity values for each wetland type which was further used to correct the spectral emissivity of the blackbody temperature generated.

**Figure 2.** Schematic representation of methodology.
Band 6 was pre-processed and used to generate land surface temperature (LST) for the wetland classes. Climate data from the Bureau of Meteorology was used to validate the generated LST and to obtain precipitation and evaporation information for the study area. Observed methane fluxes for the wetland classes were obtained by field and lab experiments. The productivity factor of methane emission from the wetland classes were estimated using net primary productivity values from literature. The area of the wetland classes, observed methane flux, productivity factor, T-factor, precipitation and evaporation ratio were used to model methane emission estimates from wetlands in north-eastern NSW Australia.

2.3. Methane Emission Estimation

Methane production process from the wetland soil is dependent on soil moisture, wetland extent, vegetation type, water table depth and temperature [28]. Higher temperatures alone would increase methane production in saturated areas, but would also cause these saturated areas to shrink resulting in a net methane reduction, while higher precipitation alone would raise the soil moisture, water table, and expand the wetland extent thereby resulting in a net increase in methane. However, an exception to these relationships is for small increase in temperature (1 °C) and precipitation (5%), for which the temperature increase can cause a slight increase in methane emission [13]. Furthermore, the vegetation type in an ecosystem would influence the net primary productivity (NPP) of the ecosystem and this is proportional to methane production [29]. The estimation of methane production from the soil in this study assumes the methane emission processes depend linearly on temperature.

This was carried out using equation 1 modified from Agarwal and Garg [18] and Liu [14].

\[
E_{\text{CH}_4} = E_{\text{obs}} \cdot F_t \cdot A \cdot P \cdot f_w
\]

where:
- \(E_{\text{CH}_4}\) = Estimated Methane Emission
- \(A\) = Area of wetland classes
- \(F_t\) = T factor
- \(E_{\text{obs}}\) = observed methane flux from different wetland classes,
- \(P\) = Productivity factor
- \(f_w = P/E = \text{Precipitation /Evaporation ratio}\), Where: \(f_w \begin{cases} P/E & \text{if } P \leq E \\ 1 & \text{if } P > E \end{cases}\).

2.3.1. Estimating Area of Methane Emitting Wetlands

The area of methane emitting wetlands was estimated from classified wetland map produced using Landsat ETM+ data. This was carried out using the following image processing steps: (a) Georeferencing (b) Subsetting (c) Masking (d) Converting DN to Radiance (e) Converting Radiance to Reflectance (f) Supervised Classification and (g) Accuracy Assessment.

The Landsat ETM+ satellite images (five scenes) acquired on June 21st 2001 were converted from Binary BIL format into ER Mapper data format in ER Mapper software version 7.1. The Landsat scenes were georeferenced based on an orthophoto and re-projected into the UTM-WGS 84 projection. Twenty-four ground control points were used in the rectification process with an overall RMS error of less than 1 pixel. The images were later subsetted to the study area and a digital elevation model...
(DEM) was used to mask out all areas that are more than 20 m altitude from the coastline. This is because the coastal zone was defined as all areas less than 20 m elevation from the coastline.

Radiometric correction was carried out in the visible and near infrared regions of the satellite images. Radiometric correction entails the correction of image pixel values for sun elevation angle variation and the calibration of images to account for degradation of the sensors over time. The changes in sensors calibration factors will obscure real changes on the ground [30]. This process involves the conversion of digital numbers to at-satellite radiances and at-satellite radiance to at-surface reflectance. This radiometric correction process includes atmospheric and illumination corrections.

2.3.1.1. Conversion of DN to Radiance—Landsat ETM+

This was carried out using Equation 2:

\[
L_{\text{rad}} = \text{Bias} + (\text{Gain} \times \text{DN})
\]

where:

- \( L_{\text{rad}} \): Spectral radiance, W/m²/sr/µm
- \( \text{Bias} \)
- \( \text{Gain} \)
- \( \text{DN} \): Digital number.

The spectral values of gain and bias for Landsat ETM+ data (Table 2) were obtained from the image header file.

<table>
<thead>
<tr>
<th>Bands</th>
<th>Gain</th>
<th>Bias</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>0.7756863</td>
<td>-6.1999969</td>
</tr>
<tr>
<td>2</td>
<td>0.7956862</td>
<td>-6.3999939</td>
</tr>
<tr>
<td>3</td>
<td>0.6192157</td>
<td>-5.0000000</td>
</tr>
<tr>
<td>4</td>
<td>0.6372549</td>
<td>-5.1000061</td>
</tr>
</tbody>
</table>

2.3.1.2. Conversion of Radiance to Reflectance—Landsat ETM+

This was performed using Equation 3:

\[
R_{\text{TOA}} = \left( \pi \times L_{\text{rad}} \times d^2 \right)/ (\text{ESUN}_i \times \cos (z))
\]

where:

- \( R_{\text{TOA}} \): the planetary reflectance
- \( L_{\text{rad}} \): is the spectral radiance at the sensor’s aperture;
- \( \pi \approx 3.14159 \)
- \( \text{ESUN}_i \): the mean solar exoatmospheric irradiance of each band
- \( d \): the earth-sun distance, in astronomical units, which is calculated using the following EXCEL equation [31,32]
  \[
d = (1 - 0.01672 \times \cos (\text{RADIANS} (0.9856 \times (\text{Julian\_Day} - 4))))
\]
- \( z \): solar zenith angle (zenith angle = 90 – solar elevation angle), solar elevation angle is within the header file of the satellite images.
2.3.1.3. Classification

Supervised classification was carried on the Landsat ETM+ image in ER Mapper version 7.1. This was performed on the reflectance images using a set of user-defined classes. This requires digitizing training sites into user-defined polygons based on knowledge of the wetlands classes obtained from regular field visits. Statistics of the training sites were generated and evaluated. The supervised classification was performed using the Maximum Likelihood Standard algorithm. This is because it considers both the means and the variances of the training data in order to approximate the probability that a given pixel belongs to a particular class. Furthermore, it produces better results compared to minimum distance to means or parallelepiped classifiers [33]. Non-wetland areas such as urban and agricultural areas were masked out from the classification.

Accuracy assessment of the wetland classification was performed by comparing the classified map with scenes from Google Earth, color orthophoto and groundtruthing of the region. 312 samples were selected from the classified map through random sampling method from the wetland classes and compared with the referenced maps. Furthermore, groundtruthing was carried out in order to validate the wetland classes. It was performed by randomly selecting the samples in the field and using of a global positioning system (GPS) as a guide to identify the wetland classes. A $k \times k$ confusion matrix (error matrix) table was developed and the producer accuracy, user accuracy, overall accuracy and kappa analysis were calculated.

The producer’s accuracy of the wetland classification was performed by dividing the total correct sample units of each wetland category by the total number of the wetland class sample units as indicated by the reference data (i.e., column total). The user’s accuracy was also performed by dividing the total number of correct pixels in each wetland category by the total number of pixels classified in that category (i.e., row total). The overall accuracy of the classification (87.50%) was computed by using the sum of the major diagonal (i.e., the correctly classified sample units) divided by the total number of sample units in the entire error matrix [34]. The kappa statistical value (85.00%) for the classification is a measure of how well the remotely sensed classification agrees with the reference data. The kappa analysis from the classification showed a high correlation between the wetland classification and the reference data used.

The classified wetland map was exported to ArcGIS and their respective areas generated. The following methane emitting wetland types were identified: mangroves and saltmarshes, forested wetlands, dunal wetlands, estuarine water bodies, coastal upland water bodies and coastal swamps.

Normalize Difference Vegetation Index (NDVI) of the reflectance image was also generated. The NDVI was generated for use in the estimation of thermal emissivity of the wetland classification. This was carried out using Equation 4 [35].

$$\text{NDVI} = \frac{\text{Near-infrared} - \text{Visible red}}{\text{Near-infrared} + \text{Visible red}}$$ (4)

The NDVI values were obtained for the following wetland classes: mangroves and saltmarshes, forested wetlands, dunal wetlands and coastal swamps with overlapping NDVI values between classes. The observed mean NDVI values and standard error (SE) were $0.665 \pm 0.044$ for mangroves and saltmarshes, $0.685 \pm 0.054$ for forested wetlands, $0.669 \pm 0.067$ for dunal wetlands, and $0.565 \pm 0.095$ for coastal swamps.
2.3.2. Estimating T factor

The T factor was derived from Equation 5 [14].

\[
F_t = \frac{F(Ts)}{\overline{F(Ts)}}
\]  

(5)

where:

\( F_t \) = T factor

\( F(Ts) = e^{0.334(Ts - 23)} 
\)

(6)

\( Ts \) = Land surface temperature in Degrees Celsius

\( \overline{F(Ts)} \) = Mean of F (Ts) over wetlands. It was derived from the F (Ts) image. The F (Ts) image was converted from raster to vector format in ArcGIS and their mean values estimated.

The mean F (Ts) value over the wetlands and standard error was 0.04 ± 0.01.

2.3.2.1. Estimating Land surface Temperature (Ts)

In this study, land surface temperature (LST) is an important input factor in the methane modeling process. It was derived from Landsat ETM+ using the thermal infrared band (10.40–12.50 µm), with a spatial resolution of 60 m. The satellite images were georeferenced using an orthophoto and re-projected into the UTM-WGS 84 projection. The images were later subsetted to the study area and a DEM was used to mask out all areas that are more than 20 m altitude from the coastline. The following processing steps were further carried out on the Landsat ETM+ band 6 satellite data:

**Conversion of Digital Number (DN) to Spectral Radiance**

The DN of the thermal infrared band (band 6) was converted to Radiance using Equation 7 [36].

\[
L_\lambda = 0.0370588 \times DN + 3.2
\]  

(7)

where:

\( L_\lambda \) = radiance, W/m²/sr/µm

\( DN \) = digital number

**Conversion of Spectral Radiance to At-Satellite Brightness Temperature (Blackbody temperature)**

The spectral radiance was converted to at-satellite brightness temperature using equation 8, assuming uniform emissivity [36].

\[
T_B = \frac{K_2}{\ln\left(\frac{K_2}{L_\lambda} + 1\right)}
\]  

(8)

where:

\( T_B \) = at-satellite temperature in Kelvin

\( L_\lambda \) = spectral radiance W/m²/sr/µm

\( K_2 \) and \( K_1 \) are pre-launch calibration constants. For Landsat-7 ETM+
\( K_2 = 1,282.71 \) K
\( K_1 = 666.09 \) mW cm⁻² sr⁻¹ µm⁻¹.
However, the temperature obtained ($T_B$) is with reference to a black body (perfect absorber and emitter of radiation) assuming a uniform emissivity. There is therefore a need to correct the spectral emissivity with respect to the land cover. Emissivity is the ratio between emittance from an object in relation to emittance from a blackbody at the same temperature [35]. The spectral emissivity for the various wetland types were derived using their NDVI values [27]. According to Van de Griens and Owe [27], when the NDVI value of natural land surface is between 0.157 and ~0.727, then emissivity may be acquired approximately from NDVI index of the vegetation using Equation 9 [27].

$$\varepsilon = 1.0094 + 0.047 \ln (\text{NDVI})$$  \hspace{1cm} (9)

where:

$\varepsilon$ = emissivity

NDVI = Normalized Difference Vegetation Index.

A constant emissivity value of 0.986 for water (estuarine water bodies and coastal upland water bodies) was adopted from the emissivity classification scheme by Snyder et al. [37].

The observed mean emissivity values for the wetland classes were as follows: mangroves and saltmarshes = 0.990 ± 0.004, dunal wetlands = 0.991 ± 0.007, forested wetlands = 0.992 ± 0.004, coastal swamps = 0.983 ± 0.009. This is, however, limited to wetland vegetation and would not be applicable to bare soil and water with NDVI values of less than 0.157. Furthermore, a change in the quality of the wetland vegetation would also change the thermal emissivity values of the wetland. The wetland map was reclassified using these values and was used to correct the spectral emissivity of the blackbody temperatures ($T_B$).

The emissivity corrected land surface temperatures ($St$) were then computed using Equation 10 [38].

$$St = \frac{T_B}{1 + (\lambda \times T_B/\rho) \ln \varepsilon}$$  \hspace{1cm} (10)

where:

$St$ = emissivity corrected land surface temperatures (LST) in Kelvin

$\lambda$ = wavelength of emitted radiance,

$\rho = h \times c/\sigma$ (1.438769 × 10$^{-2}$ m K, second radiation constant), $\sigma$ = Boltzmann constant (1.3806503 × 10$^{-23}$ J/K), $h$ = Planck's constant (6.626068 × 10$^{-34}$ J s), $c$ = velocity of light (2.99792 × 10$^8$ m/s).

$T_B$ = at-satellite temperature in Kelvin

$\varepsilon$ = emissivity of wetland classes

The unit (Kelvin) of the emissivity corrected LST was converted to degree Celsius using Equation 11.

$$I_1 - 273.15$$  \hspace{1cm} (11)

Where: $I_1 = St$.

Land surface temperature values (Table 3) were then estimated for the different wetland classes. These values were validated from climate data obtained from the Bureau of Meteorology. Furthermore, according to the IPCC [4] report, the mean annual temperature will rise by 0.2 to 1.6 °C by the year 2030. This study assumes a 1 °C rise in mean annual temperature by the year 2030 and this value was added to the LST generated. The following estimated T factor values (Table 4) for the different wetland classes were then generated based on the estimated land surface temperatures.
Furthermore, a projected T factor assuming a 1 °C rise in mean annual temperature by the year 2030 was also generated using the projected LST. This was further used to predict methane emission estimates with 1 °C rise in mean annual temperature assuming every other variable is kept constant.

Table 3. Estimated mean LST and projected LST for the wetland classes in the winter month of June including standard error of the mean (SEM).

<table>
<thead>
<tr>
<th>Wetland classe</th>
<th>LST (°C)</th>
<th>T °C from BOM</th>
<th>Projected LST (°C) assuming 1 °C rise in mean annual temperature by the year 2030</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mangroves and saltmarshes</td>
<td>12.23 ± 0.64</td>
<td>12.98 ± 0.56</td>
<td>13.23 ± 0.64</td>
</tr>
<tr>
<td>Forested wetlands</td>
<td>10.78 ± 0.51</td>
<td>10.56 ± 0.61</td>
<td>11.78 ± 0.51</td>
</tr>
<tr>
<td>Coastal swamps</td>
<td>11.77 ± 0.78</td>
<td>11.11 ± 0.72</td>
<td>12.77 ± 0.78</td>
</tr>
<tr>
<td>Estuarine water bodies</td>
<td>14.13 ± 0.72</td>
<td>15.23 ± 0.69</td>
<td>15.13 ± 0.72</td>
</tr>
<tr>
<td>Coastal upland water bodies</td>
<td>12.85 ± 0.79</td>
<td>12.12 ± 0.70</td>
<td>13.85 ± 0.79</td>
</tr>
<tr>
<td>Dunal wetlands</td>
<td>12.22 ± 0.75</td>
<td>13.31 ± 0.54</td>
<td>13.22 ± 0.75</td>
</tr>
</tbody>
</table>

2.3.3. Methane Flux

The methane flux values for the wetland classes (Table 5) were obtained from field and lab experiments. Static gas chambers with a diameter of 30 cm and a height of 44 cm and 0.07 m² surface area from which the bases had been removed were used to trap methane flux from the wetlands. The top of the chambers were gas tight sealed with a lid and a rubber stopper and were gently pushed into the soil surface to a depth of 14 cm, leaving an air space volume at the top of 21,214 cm³. Chambers used for estuarine waters and coastal upland bodies were attached to100 mm PVC pipes and floated in the estuary and lake. Four PVC pipes with a length of 1.5 m were attached at the joints using a 4 × 90° bends and the static gas chamber was attached in the centre of the PVC pipes using a plywood and cable ties. This was left to float in the estuary waters and coastal upland water bodies with a string attached. The chambers were sampled four times over a period of 30 minutes with an interval of 10 minutes and duplicate samples were collected at each time. The sampling was carried out by using a 25 ml syringe and needle to collect 25 ml of air from the chamber and stored in a non- sterilized extainers. The gas samples were analyzed within a period of at most two days using a gas chromatograph equipped with a flame ionization detector and an electron capture detector [39]. This was carried out in autumn with a total number of four points in the field for each wetland type and a total number of 16 samples for analysis for each wetland type. The geographic positions for the wetlands (Figure 3) were recorded with a global positioning system (GPS) within the study area. Samples from estuarine waters, mangroves and saltmarshes were collected in Tweed Heads. Dunal wetland samples were collected in Pottsville, coastal swamps in Lennox head, forested wetlands in Ballina and coastal upland water bodies from Lake Ainsworth. The total daily methane flux for each wetland type were directly computed by linear interpolation based on the assumption that the flux measured is a representative of the daily mean for each wetland type.
2.3.4. Productivity Factor

The productivity factor was a ratio of net primary productivity for each wetland ecosystem to net primary productivity of tropical rainforest (Equation 12) assuming that methane production in a tropical rainforest is constant throughout the year because of relatively constant temperatures and rainfall [29], *i.e.*, 

\[
R = \frac{NPP(\text{ecosystem})}{NPP(\text{tropical rainforest})}
\]  

(12)

where:
- \( R = \) Productivity factor
- \( NPP = \) Net primary productivity.

Mean productivity factors for the wetland ecosystems (Table 6) were calculated using their net primary productivity values for the wetland classes adopted from [29,40]. The net primary productivity of the wetland types were obtained from the dry weight of above ground biomass.

2.3.5. Precipitation and Evaporation Ratio

Precipitation and evaporation values were obtained from the climate statistics in the region. This study assumes water saturation in the soil as a function of precipitation and evaporation ratio. A higher precipitation relative to evaporation is assumed that the soils are water saturated for a given period and a value of 1 is assigned for every grid cell. The mean precipitation and standard error (SE) in the month of June 2001 for the study area was 166.2 ± 26.7 mm while the mean evaporation and standard error for the month of June 2001 was 125 ± 35.4 mm. The precipitation and evaporation ratio was therefore set to 1 for every grid cell.
3. Results

The maximum observed mean NDVI value of $0.685 \pm 0.054$ was for forested wetlands while coastal swamps had the minimum mean NDVI value of $0.565 \pm 0.095$ (Figure 4).

**Figure 4.** NDVI values of wetland vegetation in north-eastern NSW, Australia.

The land surface temperature values (Figure 5) in the winter month of June 2001 were found to be higher in water than on land.

**Figure 5.** Estimated Land surface temperature (LST).

The study identified and mapped the following wetland classes after the supervised classification: mangroves and saltmarshes, forested wetlands, coastal upland water bodies, estuarine water bodies, coastal swamps and dunal wetlands (Figure 6).
Figure 6. Wetland Classification in north-eastern NSW, Australia- Landsat ETM+, June 2001.

The maximum T factor values (Figure 7) were found in estuarine water bodies, while forested wetlands had the minimum T factor values.

Figure 7. T Factor values for the wetland classes.
Table 4. T factor and standard errors for the wetland classes.

<table>
<thead>
<tr>
<th>Wetland class</th>
<th>T factor ± SE</th>
<th>Projected T factor ± SE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mangroves and saltmarshes</td>
<td>0.69 ± 0.15</td>
<td>0.96 ± 0.20</td>
</tr>
<tr>
<td>Forested wetlands</td>
<td>0.45 ± 0.08</td>
<td>0.64 ± 0.12</td>
</tr>
<tr>
<td>Coastal swamps</td>
<td>0.57 ± 0.16</td>
<td>0.80 ± 0.22</td>
</tr>
<tr>
<td>Estuarine water bodies</td>
<td>1.03 ± 0.25</td>
<td>1.44 ± 0.34</td>
</tr>
<tr>
<td>Coastal upland water bodies</td>
<td>0.71 ± 0.19</td>
<td>0.99 ± 0.26</td>
</tr>
<tr>
<td>Dunal wetlands</td>
<td>0.70 ± 0.18</td>
<td>0.98 ± 0.25</td>
</tr>
</tbody>
</table>

Forest ed wetlands had the highest daily mean methane flux of $1.029 \pm 0.01 \text{ g/m}^2/\text{day}$ while coastal upland water bodies had the least mean daily methane flux of $0.015 \pm 0.004 \text{ g/m}^2/\text{day}$.

Table 5. Estimated mean methane fluxes for the wetland classes.

<table>
<thead>
<tr>
<th>Wetland class</th>
<th>Mean Flux ± SE (g/m$^2$/day)</th>
<th>Mean Flux ± SE (g/m$^2$/month)</th>
<th>Number of gas samples</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mangroves and saltmarshes</td>
<td>0.016 ± 0.01</td>
<td>0.496 ± 0.32</td>
<td>16</td>
</tr>
<tr>
<td>Forested wetlands</td>
<td>1.029 ± 0.01</td>
<td>31.286 ± 2.97</td>
<td>16</td>
</tr>
<tr>
<td>Coastal swamps</td>
<td>0.161 ± 0.05</td>
<td>4.893 ± 1.44</td>
<td>16</td>
</tr>
<tr>
<td>Estuarine water bodies</td>
<td>0.022 ± 0.0001</td>
<td>0.683 ± 0.004</td>
<td>16</td>
</tr>
<tr>
<td>Coastal upland water bodies</td>
<td>0.015 ± 0.004</td>
<td>0.461 ± 0.13</td>
<td>16</td>
</tr>
<tr>
<td>Dunal wetlands</td>
<td>0.037 ± 0.02</td>
<td>1.123 ± 0.54</td>
<td>16</td>
</tr>
</tbody>
</table>

The productivity factor (Figure 8) was highest in mangroves and saltmarshes while coastal upland water bodies had the lowest productivity factor.

Figure 8. Productivity factor of methane emission for the wetland classification.
Table 6. Mean productivity factors and standard error for the wetland classes.

<table>
<thead>
<tr>
<th>Wetland class</th>
<th>Mean Productivity factor and SE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mangroves and saltmarshes</td>
<td>1.00 ± 0.00</td>
</tr>
<tr>
<td>Forested wetlands</td>
<td>0.73 ± 0.04</td>
</tr>
<tr>
<td>Coastal swamps</td>
<td>0.75 ± 0.70</td>
</tr>
<tr>
<td>Estuarine water bodies</td>
<td>0.95 ± 0.07</td>
</tr>
<tr>
<td>Coastal upland water bodies</td>
<td>0.25 ± 0.00</td>
</tr>
<tr>
<td>Dunal wetlands</td>
<td>0.39 ± 0.13</td>
</tr>
</tbody>
</table>

The maximum mean thermal emissivity value (0.992 ± 0.004) was for forested wetlands while coastal swamps had the minimum thermal emissivity value of 0.983 ± 0.009 (Figure 9).

**Figure 9.** Thermal emissivity values for the wetland classes.

The accuracy assessment carried out for the wetland classification (Table 7) showed a maximum user's accuracy (96.15%) for mangroves and saltmarshes and the minimum user's accuracy (78.00%) for forested wetlands.
Table 7. Error Matrix Table for Landsat TM + 2001 wetland classification.

<table>
<thead>
<tr>
<th></th>
<th>F</th>
<th>M</th>
<th>D</th>
<th>S</th>
<th>L</th>
<th>E</th>
<th>Row Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>F</td>
<td>39</td>
<td>0</td>
<td>8</td>
<td>3</td>
<td>0</td>
<td>0</td>
<td>50</td>
</tr>
<tr>
<td>M</td>
<td>0</td>
<td>50</td>
<td>0</td>
<td>2</td>
<td>0</td>
<td>0</td>
<td>52</td>
</tr>
<tr>
<td>D</td>
<td>2</td>
<td>0</td>
<td>46</td>
<td>3</td>
<td>0</td>
<td>0</td>
<td>51</td>
</tr>
<tr>
<td>S</td>
<td>8</td>
<td>1</td>
<td>0</td>
<td>44</td>
<td>0</td>
<td>0</td>
<td>53</td>
</tr>
<tr>
<td>L</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>43</td>
<td>7</td>
<td>0</td>
<td>50</td>
</tr>
<tr>
<td>E</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>5</td>
<td>51</td>
<td>0</td>
<td>56</td>
</tr>
<tr>
<td>Column Total</td>
<td>49</td>
<td>51</td>
<td>54</td>
<td>52</td>
<td>48</td>
<td>58</td>
<td>312</td>
</tr>
</tbody>
</table>

Producer’s Accuracy | User’s Accuracy | Overall Accuracy | Overall Kappa Statistics  
D = Dunal wetlands  | 85.19%         | 90.20%         | 87.50% | 85.00% |
F = Forested wetlands | 79.59%       | 78.00%       | 81.00% |        |
S = Coastal swamps    | 84.62%       | 83.02%       | 83.00% |        |
L = Coastal upland water bodies | 89.58%   | 86.00% |       |        |
M = Mangroves and saltmarshes | 98.03%  | 96.15%    |       |        |
E = Estuarine water bodies | 87.93%     | 91.07%     |       |        |

The study also found forested wetlands to have the highest amount of methane emission (0.0016 ± 0.00009 Tg) in the month of June, 2001 while coastal upland water bodies had the minimum amount of methane emission (0.0000019 ± 0.0000005 Tg) in the month of June, 2001 (Table 8). According to the IPCC [4] report, the mean annual temperature will rise by 0.2 to 1.6 °C by the year 2030. In line with the IPCC [4] projection, an estimation of methane emission from the various wetlands by the year 2030, assuming 1 °C rise in mean annual temperature was projected to increase in the month of June (Table 8).

Table 8. Methane emitting areas and emission estimates in the month of June 2001 including mean annual temperature increase by 1°C in north-eastern NSW, Australia.

<table>
<thead>
<tr>
<th>Wetland type</th>
<th>Area covered by wetland (km²)</th>
<th>Methane emission estimate in June (Tg) ± SE</th>
<th>Methane emission estimate (Tg) in June assuming a 1 °C rise in mean annual temperature ± SE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mangroves and saltmarshes</td>
<td>36.56</td>
<td>0.000013 ± 0.000006</td>
<td>0.000018 ± 0.000008</td>
</tr>
<tr>
<td>Forested wetlands</td>
<td>152.09</td>
<td>0.0016 ± 0.00009</td>
<td>0.0022 ± 0.0001</td>
</tr>
<tr>
<td>Coastal upland water bodies</td>
<td>32.74</td>
<td>0.0000019 ± 0.000005</td>
<td>0.0000037 ± 0.000007</td>
</tr>
<tr>
<td>Estuarine water bodies</td>
<td>35.97</td>
<td>0.000024 ± 0.0000001</td>
<td>0.000034 ± 0.000001</td>
</tr>
<tr>
<td>Coastal swamps</td>
<td>150.56</td>
<td>0.00031 ± 0.00002</td>
<td>0.00044 ± 0.00001</td>
</tr>
<tr>
<td>Dunal wetlands</td>
<td>73.37</td>
<td>0.000022 ± 0.000008</td>
<td>0.000031 ± 0.000001</td>
</tr>
<tr>
<td>Total</td>
<td>481.29</td>
<td>0.0019 ± 0.0001</td>
<td>0.0027 ± 0.0002</td>
</tr>
</tbody>
</table>
4. Discussion

The higher land surface temperature values found in water than on land in the winter month of June 2001 is probably due to the fact that in winter, water would absorb solar energy and hold onto heat longer than land. The possibility of using land surface temperature in estimating water table depth is an area of further research. Thermal emissivity of the wetlands was used to correct the spectral emissivity of the blackbody temperature generated. The high thermal emissivity value for forested wetlands was due to their high NDVI values. Nevertheless, there would be variation of the thermal emissivity values for the wetlands especially within seasons. This is because the NDVI values are likely to change with seasons due to changes in environmental conditions and the resulting foliage replacement and flowering cycles of the plants.

The higher T factor values found in estuarine water bodies was due to the higher temperatures found in estuarine waters at the time of satellite image acquisition. There would, however, be uncertainties in the T factor values due to the fluctuation in land surface temperatures within days, months, seasons and years. There were also variations in the productivity factor of the wetlands for methane emission. This is because it is dependent on the net primary productivity of the wetlands, which further depends on the variables such as vegetation, hydrology, climate, soil type and nutrients availability [40]. The area covered by the wetlands also contributed to the estimated amount of methane emission. Forested wetlands had the maximum amount of methane emission due to a large area and methane flux. The estimation of area covered by the wetlands was limited to the satellite data used and the environmental conditions of the wetlands at the time of image acquisitions. This is because the spatial resolution of the satellite data would influence the area covered by the wetlands and conditions such as floods and tides would also influence the wetland area. High tides would likely affect the area covered by estuarine water bodies, mangroves and saltmarshes due to their proximity to the sea. The estimated amount of methane emission from the wetlands classes was also influenced by their methane fluxes, which were obtained from field and lab experiments. The methane fluxes observed from mangroves and estuarine water bodies in this study were lower than the calculated mean methane fluxes of 7.38 mg/m²/hr for Pichavaram mangrove and 15.41 mg/m²/hr for Adyar estuary in south India [41]. The lower methane fluxes observed in mangrove sediments maybe a result of out-competition for substrates [42] by sulfate reducers. This is because in marine environments with high salinity, methanogenesis may be inhibited by sulfate-reducing bacteria [43,44]. Forested wetlands generally had a higher methane flux compared to the estimated 23.5 ± 11.3 g/m²/yr for forested wetland ecosystems calculated globally in Sheppard [29].

The variability of methane flux was high in the wetlands, especially in the forested wetlands. The uncertainties in methane flux were probably due to the changing environmental conditions such as soil temperature, rate of methane oxidation in the oxic soil between water table and soil surface and the difference in substrate availability. An increase in environmental conditions such as temperature and soil moisture content would increase methane flux from wetlands and vice versa [39]. Methane fluxes from wetlands would furthermore be affected by seasonal conditions. This is because there are different seasonal conditions of variables such as rainfall, temperature, evaporation and vegetation types, thereby increasing the variability of methane flux in a year.
The study assumed that methane emission processes from wetlands depend linearly on temperature due to the fact that methanogenic activities are influenced by soil temperature. It further assumed soil moisture content based on the ratio of precipitation and evaporation within a region, which is, however, limited to the availability of climate data. In addition, it assumed the mean methane fluxes measured from the various wetlands as a representative of the daily mean for each wetland type. This would nonetheless change with changing environmental conditions. However, in order to estimate annual methane emission from the wetlands it is imperative to model methane fluxes for the various seasons. This is because methane emission is expected to change throughout the year due to changes in parameters such as precipitation, and temperatures during winter, summer, autumn and spring. During summer with high temperatures, methane emission is likely to increase but this could be limited by a high rate of evaporation thereby reducing the area covered by wetlands. Methane emission would increase in autumn due to high rainfall, thereby increasing wetland area and soil moisture content. During spring and winter, methane emission from wetlands would decrease due to less rainfall and lower temperatures in the north-eastern region of NSW, Australia. Nevertheless, there is a further limitation in the estimation of annual methane emission from wetlands using satellite data. This includes the lesser availability of cloud free satellite images on a monthly basis which is necessary in order to acquire wetlands information such as area cover and quality for methane estimation.

Climate change (temperature increase) would probably increase natural methane emission from global wetlands. Policies geared towards carbon sequestration and storage could offset climate change and its impact to methane emission. The mitigation of methane emission from wetlands to the atmosphere is an area for further research. Remote sensing could play a vital role in identifying and monitoring the wetlands, and the acquisition of biophysical properties which could be used to improve our understanding of methane emission over time.

5. Conclusions

An estimation of methane emission from wetlands has been carried out using Landsat ETM+. This was modeled using the following parameters: productivity factor, temperature dependent factor (T-factor), wetland area, methane flux, precipitation and evaporation ratio. The estimation of methane emission from the wetlands has been carried out assuming methane production is linearly dependent on temperature. The study found a high variability of methane emission from the wetlands with the maximum amount of methane emission from forested wetlands and the minimum amount of emission from coastal upland water bodies. Methane emission is anticipated to increase with climate change which is most likely due to an increase in metabolic activities of the soil microbes. There are however uncertainties in the methane emission estimation due to changes in methane fluxes and environmental conditions such as temperature and rainfall over time from the wetlands. There are also limitations in the use of satellite data to estimate methane emission from wetlands such as the limitation in acquiring cloud free satellite images especially on a monthly basis and the factor of scale (spatial resolution) in delineating the wetland classes. Nevertheless, the empirical methane emission model using Landsat ETM+ is suitable to estimate monthly and yearly methane budget from wetlands at both local and regional level.
Acknowledgements

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