Implementation of a GIS to Assess the Effects of Water Level Fluctuations on the Wetland Complex at Long Point, Ontario

by

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AUTHOR'S DECLARATION

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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ABSTRACT

The Long Point wetland complex is one of the most significant coastal wetland systems in the Great Lakes, containing a diverse mosaic of wetland vegetation communities that have developed in response to water level fluctuations due to natural climate variability. Natural short-term water level variations are important for promoting wetland productivity and diversity, but long-term water level changes resulting from human-induced climate change can have serious and long-term consequences on the integrity and health of wetlands. The historical response of the wetland to water level fluctuations was quantified and modelled to provide an indication of how the wetland may respond to future projected water level changes - water level fluctuations are used as a surrogate for climate change.

A spatiotemporal trend analysis was conducted within a geographic information system (GIS) to determine the effects of water level conditions on wetland vegetation and land cover at the wetland complex at Long Point, Ontario for seven years from 1945 to 1999. The spatiotemporal trend analysis documented changes in the structure and composition of the wetland complex in response to declining and rising water level conditions. During drier periods, there were significant increases in the amount of drier emergent and meadow vegetation, especially within the Inner Bay and northern portion of the outer peninsula. There was less fragmentation and complexity in the wetland as these drier communities expanded forming larger continuous patches of vegetation. During wetter periods, open water increased and there was a predominance of wetter emergent and meadow communities in the wetland. Drier vegetation communities became interspersed with water creating a more fragmented convoluted wetland landscape.

The historical response of the wetland vegetation and land cover to water level fluctuations was then simulated with three different wetland models developed in the GIS. A rule-based model, a probability model, and a transition model were developed to assess wetland response to future water level changes. The models were evaluated using simple statistical methods. The transition and rule-based models performed the best and were successful in predicting over 80 % of the wetland vegetation distribution correctly. The probability model was the least successful, predicting only 55 % of the response correctly.

The GIS proved successful in documenting wetland response to historical water level fluctuations and providing insight into the potential impacts of future climate change though water level fluctuations on the Long Point coastal wetland complex. The spatiotemporal analysis and wetland modelling advance the role of GIS in wetland management and analysis. They are practical methods within a GIS that can be used to assess the impacts of climate change on wetland systems and to document and model wetland change in other coastal wetlands of the Great Lakes.

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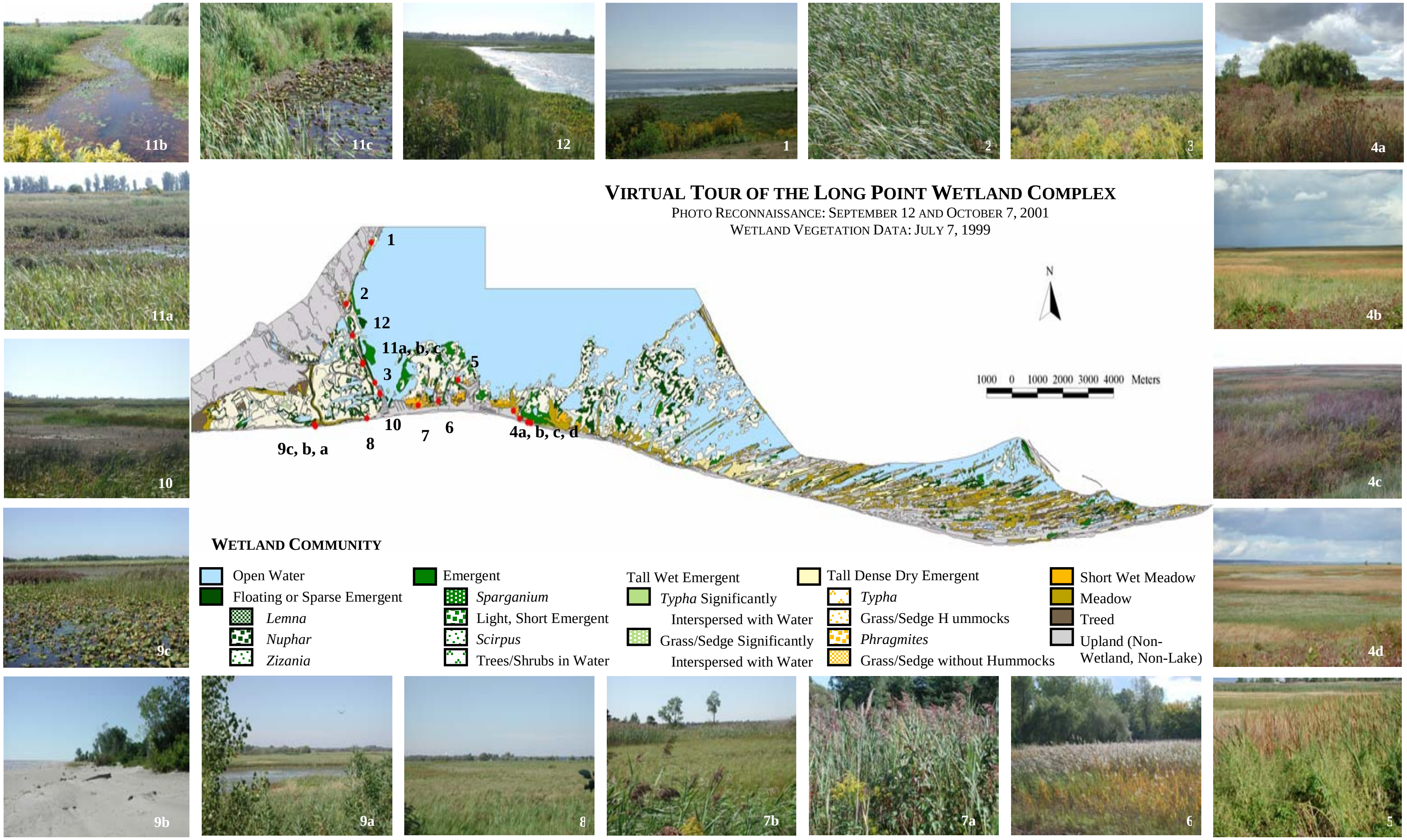
1.0 INTRODUCTION

Long Point, situated on the northern shore of Lake Erie within the Regional Municipality of Haldimand-Norfolk near the town of Port Rowan, Ontario, is one of the most significant wetland complexes in Southern Canada. The complex contains a rich mosaic of habitat from sandy beaches and dunes, grass-covered ridges, savannas, open ponds, wet meadows, and marshes to forests that supports a diversity of natural vegetation and wildlife (Figure 1-1). Local efforts by various private groups and government agencies have been extremely successful at preserving Long Point. The wetland complex has been designated as a World Biosphere Reserve and it remains one of the least developed coastal wetlands in the world (Long Point Bird Observatory, 2001).

Coastal wetlands, such as Long Point, are transitional zones between permanent terrestrial and aquatic environments along the shore of a lake and are highly influenced by lake processes including waves, seiches, and seasonal and long-term water level fluctuations (GLIN, 1998a). In general, wetlands are defined as lands that are periodically or permanently inundated with “water long enough to promote wetland or aquatic processes as indicated by poorly drained soils, hydrophytic vegetation, and various kinds of biological activity that are adapted to a wet environment” (National Wetlands Working Group, 1987). Wetlands are one of the most productive and diverse ecosystems in the world and provide numerous environmental, social, and economic benefits and functions.

Wetlands:

- provide habitat for mammals, birds (particularly shore birds and migratory waterfowl), fish, reptiles, amphibians, and invertebrates;
- provide protected areas for fish spawning and nurseries;
- provide wintering and nesting areas for migratory waterfowl and feeding for migratory stopovers;
- cycle and store nutrients and pollution, including nitrogen, sulphur, methane, and carbon dioxide;
- filter pollutants through waste assimilation and absorption of toxic substances;
- protect the shoreline from erosion by slowing and trapping the flow of sediments and act as a buffer for waves;
- regulate stream and lake levels through the storage of floodwater and groundwater recharge in smaller and medium size watersheds;
- provide educational and scientific opportunities; and,
- provide opportunities for consumptive and non-consumptive tourism and recreation activities such as hunting, fishing, boating, canoeing, bird and animal watching, and aesthetic enjoyment (CWS, 2002; Delesalle, 1998; Bolsenga and Herdendorf, 1993; Herdendorf, 1992; Herdendorf, 1987; Bayly, 1979b).



DESCRIPTION OF PHOTOGRAPHS (Number. Location: *gradation of vegetation from foreground to background*)

1. Port Rowan Lookout: *mix of meadow and Typha, Typha, emergent and floating emergent, open water/submergent*
2. Cemetery: *Typha*
3. Long Point Bay: *meadowy upland, tall dense dry emergent, floating emergent, open water/submergent, emergent with Typha to left*
4. Long Point Provincial Park: (a) *meadow, treed* (b) *shrub meadow, tall dense dry emergent, short wet meadow, tall dense dry emergent, Typha* (c) *meadow, tall dense dry emergent and short wet meadow* (d) *meadow, tall dense dry emergent, short wet meadow, tall dense dry emergent, open water with emergent, tall dense dry emergent*
5. Old Cut Boulevard: *meadow, Typha, short wet meadow, Phragmites*
6. Johnson Avenue: *Phragmites, treed*
7. Inner Long Point Bay: (a) *Phragmites, treed* (b) *meadow with Phragmites*
8. Hastings Drive: *grass/sedge hummocks*
9. End of Hastings Drive: (a) *meadow, tall dense dry emergent around rim of floating emergent, tall dense dry emergent, treed* (b) *sand with non-wetland treed* (c) *floating emergent (Nuphar), emergent (Decodon), floating emergent, tall dense dry emergent, treed*
10. Inland to Big Creek: *mudflat within tall dense dry emergent*
11. Big Creek National Wildlife Area: (a) *tall dense dry emergent, floating emergent in centre* (b) *Typha surrounding floating emergent* (c) *floating emergent rimmed by mix of meadow and Typha*
12. Mouth of Big Creek: *meadow, Typha, river, emergent along river edge, treed in horizon*

FIGURE 1-1: THE LONG POINT WETLAND COMPLEX

Although wetlands are an integral component of the earth's natural system, wetlands are extremely vulnerable to environmental change. Natural and anthropogenic factors have contributed to a progressive loss in the areal extent and quality of wetlands within the Great Lakes, particularly within the lower Great Lakes, over the last century (Herdendorf, 1987). Wetland vulnerability is primarily influenced by natural and anthropogenic climatic variability or change. Natural stressors related to climate variability, including water level fluctuations, sediment transport, ice, and storms are an essential component of change within wetlands. Human activities and biological factors also affect change within wetland systems. In fact, human development has resulted in a loss of 83 percent (%) of wetlands within the Great Lakes region alone (Herbert, 2000). The amount of wetland area has been severely diminished by draining, dredging, and filling of wetlands for urban, agricultural, and industrial land uses, and by the creation of dikes for water level control. In addition to human development, increasing amounts of siltation and nutrient runoff due to agricultural practices, pollution from urban and industrial activities, the introduction of non-native or exotic species, the expansion of invasive or opportunistic species, and disease have placed further stress on wetland systems (GLIN, 1998b). Wetland change from natural processes is reversible, but change from human interference may be permanent and detrimental to the functionality of wetland systems.

1.1 Problem Statement

Variations in climatic conditions have been instrumental in the development and existence of coastal wetlands within the Great Lakes (GLIN, 1998b). Long-term natural variations in climatic conditions are driven by changes in atmospheric circulation patterns, solar output, and the earth's orbit. Coastal wetlands are sensitive to many climatic conditions and changes to these conditions may directly or indirectly impact wetland systems. The most important climatic conditions affecting wetland systems are temperature and precipitation. Temperature influences the length of growing season, type of vegetation, primary productivity, and rate of chemical and biological reactions within the wetland. Temperature and precipitation, combined with other climatic conditions including solar radiation, wind, cloud cover, evaporation, and evapotranspiration are important regulators of water levels in the Great Lakes (Mortsch, 1998). These climatic variables affect the hydrology of the Great Lakes by influencing the amount of surface runoff, soil moisture, and groundwater storage in the system.

Coastal wetlands are located along the interface between the terrestrial and aquatic environments and are highly influenced by water level fluctuations; therefore changes in the water level regime will affect the quality and quantity of vegetation within the wetlands. Slight variations to the climatic conditions of the Great Lakes basin affect the magnitude, frequency, timing, and duration of water level fluctuations, which in term has a tremendous influence on the natural vegetation and wildlife within coastal wetland systems. Periodic water level fluctuations due to natural climatic variability are important for promoting and enhancing the productivity and diversity of wetlands by stimulating vegetation growth, providing nutrients, and eroding sediments. Short-term natural variations also influence, to some extent, the distribution of plants and the anaerobic conditions of the soil (Lyon *et al*., 1986).

Human-induced climate change may place additional stress on coastal wetland systems within the Great Lakes. Whereas, climatic changes from natural processes are inevitable and relatively slow on the time scale, the effects of human-induced climate change can have serious and long-term consequences on the integrity and health of wetland systems. Primarily the burning of fossil fuels as well as deforestation, agricultural, and industrial processes release carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), and chlorofluorocarbons (CFCs) into the atmosphere. These greenhouse gases become trapped in the atmosphere and contribute to the enhanced warming of the planet. There is evidence that global average surface temperatures have already increased 0.6 degrees Celsius (°C) over the twentieth century as a result of global warming (IPCC, 2001). Enhanced global warming will inherently alter the climate of the Great Lakes basin and thus consequently affect water levels that form and help maintain coastal wetlands in the Great Lakes (GLIN, 1998b).

Various climate change scenarios are projecting warmer mean annual temperatures and less annual and spring snowfall for the Great Lakes basin (Mortsch *et al*., 2000). Warmer summer temperatures, resulting in more evaporation and transpiration, combined with less precipitation in the summer reduce the amount of land runoff entering the basin. Warmer winter temperatures result in an increase in winter rainfall and a decrease in winter snowfall, therefore leading to a decrease in the amount of runoff from snowmelt in the spring (Mortsch, 1998). Subsequently, the net basin water supply in Lake Erie is projected to decrease and the mean lake level is expected to decline (GLIN, 1998b). Projected Lake Erie water level declines due to enhanced global warming will undoubtedly affect the quality and quantity of coastal wetlands along the shore of Lake Erie. Longer-term water level fluctuations, as a result of human-induced climate change, will affect the vegetation composition, structure, and areal extent of wetlands, which consequently can have serious implications to the environmental, social, and economic functions of wetland systems (Bukata *et al.*, 1988a; Lyon *et al.*, 1986).

1.2 OBJECTIVES OF THE THESIS

To understand how wetland vegetation may respond to projected climate change, it is important to quantify how the wetland has responded to historical climate change where water levels changes are used as a surrogate for climatic changes. Geographic information systems (GIS) have been identified as a useful tool for analyzing and modelling spatiotemporal trends in wetland systems (Johnson, 1990). Spatiotemporal trend analyses provide a key to how wetland vegetation responds over time to changes in water level conditions that can be used to infer how the wetland may respond to alterations in water levels due to projected climate change. Therefore, a GIS will be employed in this thesis to analyze historical wetland change at Long Point.

More specifically, the main goal is to utilize a GIS (1) to determine the effects of water level fluctuations on wetland vegetation and land cover in the wetland complex at Long Point, Ontario, both on a temporal and spatial scale, and then (2) attempt to model wetland vegetation distribution in response to water level fluctuations. The main objectives of this thesis are to:

- collect and digitize wetland classification maps of Long Point for years that represent periods of low, medium, and high water levels and water level trends that are rising and declining;
- identify the documented response of wetland vegetation to historical water level fluctuations and the potential of analyzing this response within a GIS;
- quantify and characterize spatial and temporal changes within the wetland vegetation communities at Long Point in relation to water level fluctuations within ARC/INFO (ESRI, 1999);
- develop several wetland models in ARC/INFO that are based on different modelling approaches to simulate wetland vegetation response to water level fluctuations; and,
- evaluate the accuracy of the models and assess the applicability of the approaches to future modelling efforts.

The aim of this thesis is to provide a practical example of the application of a geographic information system to assess the impacts of climatic changes on the coastal wetland complex at Long Point, Ontario and create a systematic methodology within a geographic information system for documenting and modelling wetland vegetation response to water level fluctuations that may be applicable to other coastal wetland systems in the Great Lakes.

1.3 Structure of the Thesis

In Chapter 2, an examination of the physical, ecological, and social processes that have shaped the Long Point wetland complex is provided. The chapter also discusses coastal wetland systems, typical vegetation communities within marsh wetlands, and the response of these communities to water level fluctuations. Chapter 3 reviews the current status of wetland research. Research documenting and modelling wetland vegetation response to climatic changes and water level fluctuations, both in a spatial and non-spatial capacity, is reviewed. These chapters provide the framework for understanding the response of wetland vegetation communities to water level fluctuations and climate change within the Long Point coastal wetland complex. Chapter 4 outlines the research methodology employed to assess the implications of climate change on Long Point. Data collection and processing procedures are outlined and then the methods of analysis and model development are discussed. Chapters 5 and 6 present and discuss the results of the spatiotemporal and modelling analyses, respectively. Chapter 7 presents the conclusions of the thesis. This chapter includes a discussion of the caveats of the data and methodology and provides recommendations for further research.

2.0 HISTORY AND ECOLOGY OF LONG POINT

Examination of the physical and ecological characteristics of the Long Point wetland complex, along with the historical development of the area, is an integral component in understanding the temporal and spatial changes that have occurred and modelling the response of the wetland to climate change. The first section of this chapter provides a summary of the physical, historical, ecological characteristics of the Long Point wetland complex. The second section focuses on wetlands in general, the different vegetation communities within wetlands, and the impacts of climate change and water level fluctuations on wetland systems.

2.1 LONG POINT

An overview of the Long Point area is provided in this section. First, the evolution of Long Point is briefly discussed in terms of the physical processes and climatic characteristics that have helped form the wetland complex. Next, the historical development of Long Point is discussed. Then a summary of natural hazards and their impact on Long Point is provided. The section concludes by detailing the biological resources of the area.

2.1.1 EVOLUTION OF LONG POINT

The Long Point wetland complex has developed along a sandy peninsula that extends 37 kilometres (km) eastward into the deepest part of Lake Erie. The formation of the peninsula began 4,000 years ago as long-shore drift deposited sediments eroded from cliffs to the west into shallow areas at the point. The process continues today and is gradually extending the eastern tip of the peninsula further into the lake. Between 1853 and 1945 the tip grew at an average rate of 7 metres (m) a year; when the tip expanded out into the deepest part of the lake, the annual growth rate decreased to 5 m (Lawrence and Nelson, 1994; CWS, 1983). Although the eastern tip of the peninsula is growing, areas in the western section, particularly Big Creek at the base of the peninsula and the Long Point Company Marsh, receive little deposition and are slowly eroding away (CWS, 1983).

2.1.2 CLIMATE

Long Point has a moist continental climate that is strongly influenced by Lake Erie. Spring and summer temperatures are cooler than the surrounding mainland and fall and winter temperatures are warmer because of the moderating effects of the lake. The average temperature during the summer is 22 degrees Celsius (°C), whereas the winter temperature averages 1°C. The warm climate of Long Point results in a long growing season with an average of 200 frost-free days throughout the year. Annual precipitation at Long Point is 998 millimetres (mm); prevailing winds are south-westerly. Storms are frequent and intense throughout the year. During the summer, storm wind speeds average 25 kilometres per hour (km/h) and during the winter, wind speeds average 50 km/h (CBRA, 1999; CWS, 1983; McCracken *et al.*, 1981).

The climate at Long Point is, and has been, an important regulator of the natural processes that occur at Long Point. Shoreline erosion and deposition, geochemical cycling, seasonal and long-term water level fluctuations, vegetation succession, and animal migration patterns on Long Point depend on the climate. Changes in the climate can have serious implications to the physical, ecological, and socioeconomic patterns that exist in the region today. Before examining the ecological impacts of climate change on the wetland vegetation communities within Long Point, historical and natural changes of the area are examined. The biological resources of Long Point are also discussed.

2.1.3 HUMAN INFLUENCES

The earliest known recorded inhabitants, the Ontario Iroquois, settled in the mainland around Long Point between 900 and 1200 AD. The Neutral Iroquois displaced the Ontario Iroquois in 1400, but disease and famine reduced the Neutral population to less than half and the remaining population was dispersed by the invasion of the New York Iroquois in 1650. The Iroquois and the Mississauga utilized the area for seasonal hunting and trapping until European settlement in 1790 (CWS, 1983). Early European immigrant settlements first appeared near Turkey Point, where the soil was more fertile and productive, but with the construction of Lakeshore Road in 1800, and subsequently the introduction of the railroad to the area in 1888, settlement continued to develop south towards Long Point. A deed for the first private land holding along the peninsula was granted for Ryersons Island in 1808 (Skibicki, 1993; CWS, 1983).

By the middle to late nineteenth century, European development was having an adverse impact on the natural environment. Dense forest stands were being harvested for commercial logging or cleared for agricultural land. In addition, wild game and fish stocks were being heavily exploited by commercial and recreational hunting, trapping and fishing. The depleted state of the natural resources prompted a group of sportsman, who later became known as the Long Point Company, to purchase most of the peninsula in 1866. The company purchased 6,044 hectares (ha) of land from the province to protect the natural resources from further exploitation. The government kept 71 ha of land near the eastern tip of the peninsula for a lighthouse and in 1960 permitted the Long Point Bird Observatory to build a facility on the land to conduct bird migration studies (CWS, 1983).

The Anderson property, at the tip of the peninsula, was purchased from the government in 1890; the property has been leased out over the years and still exists today. In 1908, the government purchased 41 ha of land from the Long Point Company for the St. Williams Forestry Station. The station provided local farmers with nursery stocks that could be used to help minimize wind and soil erosion in their agricultural fields. In 1919, the company granted a long-term lease to the Bluffs Shooting and Hunting Club for 40 ha of land at Bluff Point; the lease expired in 1985 (Nelson and Wilcox, 1996; CWS, 1983).

Long Point Park, a 162 ha recreational park, was established in 1921 and was a catalyst for development along the peninsula. Several years later, parkland was being leased out for private use and cottages were being constructed. In 1928, a causeway was constructed from the mainland onto the peninsula providing easier automobile access to the peninsula. The causeway created an influx of recreational users to the park and prompted the government to purchase an additional 57 ha of land from the Long Point Company for expansion. Long Point Park became a provincial park in 1956, at which time there were a total of 450 private cottages and six permanent residents within the park boundaries. Subsequently, the government permitted homeowners and cottagers to purchase their properties from the crown; that action diminished the size of the park from 219 ha to 9.3 ha and led to the creation of a new provincial park just to the east. The Long Point Provincial Park was established in 1961 on 132 ha of land the government purchased from the Long Point Company. Pressure from recreational development during the 1960s was severe, and by 1970 there were over 900 privately owned cottages and recreational services in Long Point (Nelson and Wilcox, 1996; Skibicki, 1993).

During the 1970s, the Canadian Wildlife Service (CWS) obtained lands to create National Wildlife Areas (NWAs) within Long Point that protect and promote the sustainable use of waterfowl habitat. Along with purchasing land between 1971 and 1973 to create the Big Creek NWA, the CWS received 3,239 ha of land from the Long Point Company and Nature Conservancy in 1978 to establish the Long Point NWA. In 1985, a 70 ha impoundment was constructed within the Big Creek NWA to regulate water levels and increase the diversity of open water vegetation (Skibicki, 1993). Refer to Figure 2-1 for an illustration of the current land divisions and ownership within Long Point.

In 1982, Long Point was recognized under the Ramsar Convention as a Wetland of International Significance (Ontario Parks, 1997). The Ramsar Convention promotes the conservation and wise use of wetlands, and the recognition ensures that the ecological integrity of Long Point will be maintained through the efforts of the Government of Canada (Ramsar, 1998). In 1986, Long Point was designated as a World Biosphere Reserve under the United Nations Educational, Scientific and Cultural Organization (UNESCO) Man and the Biosphere (MAB) Program. The designation, which included a total of 27,000 ha of land, promotes the conservation and sustainable use of biodiversity (UNESCO, 2002). Most recently, Long Point was recognized as an International Monarch Butterfly Reserve in 1995 and as a Globally Important Bird Area by BirdLife International and the North American Commission for Environmental Cooperation in 1996 (IBAC, 2002).

2.1.4 NATURAL INFLUENCES

Natural events, such as fire and storms, have had an impact on the natural and physical environment of Long Point. Fire has been responsible for maintaining the open forest and meadow vegetation communities that has led to savannah like conditions in much of the eastern two-thirds of the peninsula (Catling and

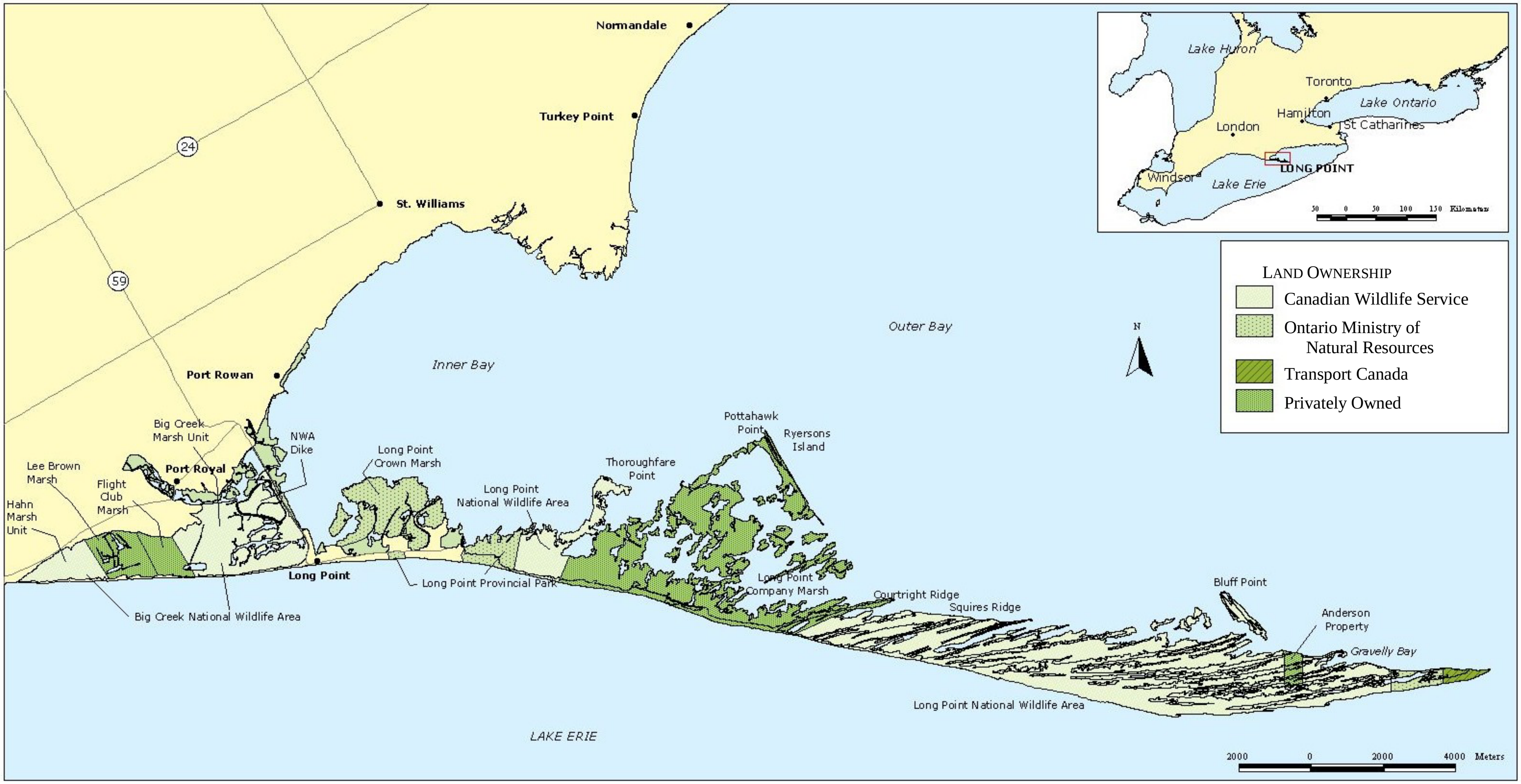


FIGURE 2-1: LONG POINT STUDY AREA

Reznicek, 1981). In 1881, fire burned a large portion of the Long Point Company Marsh along with three dune ridges. There were fires on Courtright and Squires Ridges in 1933, and again on Courtright Ridge in 1957. A large portion of company land between Squires Ridge and Gravelly Bay was burned by fire in 1962. The following year, there were reported fires at Cedar Creek Ridge and within the Long Point Provincial Park, where 140 ha of land burned. There were also fires at Little Creek Ridge in 1965 and at Squires Ridge in 1983, where 0.5 ha of land burned (Skibicki, 1993; CWS, 1983). Drought-like conditions in the 1930s and 1960s (due to high evaporation and low amounts of precipitation) coincided with, and may have contributed to, some of the fires (GLERL, 2001).

Storm activity has had an adverse effect on the natural environment at Long Point. The orientation of Long Point runs south-westerly to north-easterly, parallel to the long axis of Lake Erie. Thus, predominantly westerly storm systems can have a significant effect on Long Point. Storm systems advancing across the lake can build up tremendous wave and wind energy due to the long fetch and then unleash the energy along the shoreline of Long Point causing extensive erosion and alterations to the physical environment and damage to the natural vegetation. Storms have caused extensive flooding in low lying parts within the wetland complex and rapid erosion and deposition of sediments along the shoreline and within the wetland. Vegetation can be uprooted or completely eradicated during severe storms, and private property, such as cottages, can be entirely destroyed, especially during periods of high water levels.

In 1883, a storm breached Long Point creating Old Cut Channel. The channel was 360 m wide and up to 5.5 m deep and used for navigation until erosion and deposition forced its closure in 1885. The channel was subsequently reopened by a storm in 1901, but closed by another storm in 1906. In 1865, storms opened a channel just to the west of Old Cut Channel. Subsequent storms widened and deepened the channel to 800 m and 6 m, respectively, but shifting sand and sandbars closed the channel in 1916. Long Point suffered extensive damage from Hurricane Hazel in the fall of 1954 and from a severe storm in the spring of 1955. There were periods of flooding over Hastings Drive in 1969 and within Long Point Provincial Park in 1973. In 1975, a storm caused extensive flood and erosion damage along Hastings Drive and within land owned by the Long Point Company. Finally, in 1985 during a period of record high water levels, a severe storm with strong winds and high waves caused significant damage to cottages and private property along the shore and caused flooding over Hastings Drive (Lawrence and Nelson, 1994; CWS, 1983).

2.1.5 VEGETATION

The natural flora at Long Point is quite diverse and although there are many different species of vegetation, only a few have significant ecological roles (Bolsenga and Herdendorf, 1993). Long Point has over 20 distinct biotic communities containing over 700 species of vascular plants, of which 90 are rare in Ontario and four do not occur anywhere else in Canada (Nelson and Wilcox, 1996). These biotic communities are dispersed across three unique sections of the wetland complex that have developed under different hydrologic and physiographic conditions related to the moisture and organic matter content of the soil and the age of the substrate (Catling and Reznicek, 1981).

Lying at the base of the peninsula is the western section. Commonly referred to as Big Creek, the west section contains a number of marshes including the Hahn Marsh Unit, Lee Brown Marsh, Flight Club Marsh, and Big Creek Marsh Unit; the Hahn and Big Creek Marsh Units are part of the Big Creek National Wildlife Area. Big Creek is characterized by shallow marsh and wet meadow communities with mono-dominant stands of cattail (*Typha*) or other emergent vegetation such as sedge (*Carex*), reed grass (*Calamagrostis*) or wild rice (*Zizania*) (Francis *et al.,* 1985). The shallow marsh and wet meadow communities progress into dense marsh and wet woodland communities, further west, as the vegetation adapts to a drier, older, and more stable wetland environment (McCracken *et al.*, 1984). Big Creek is isolated from Long Point Bay by a causeway; therefore water level fluctuations in the bay have no direct impact on the wetland vegetation in this portion of the wetland complex. The vegetation is affected by spring flooding in the Big Creek River channel, high water levels in Lake Erie, and storm surges and seiches along the lake (McCracken *et al.,* 1984).

The middle section extends east from the base of the peninsula and contains many small inlets and marshes that have developed on a shallow embayment created by infilling of the Inner Bay. This section includes the Long Point Crown Marsh, Long Point Provincial Park, Long Point Company Marsh and the smaller division of the Long Point NWA. This section is characterized by deep-water marshes that have developed within the sheltered environment of the Inner Bay. The deep-water marshes contain a mix of emergent, submergent, and floating-leaved communities that are maintained in an early successional stage by seasonal and yearly lake level fluctuations (McCracken *et al.*, 1984). It has been estimated that ninety percent of the entire bottom of the Inner Bay is covered by submergent and floating-leaved vegetation; emergent communities are dominated by stands of *Typha* (Francis *et al.,* 1985).

The eastern section of Long Point extends east from Squires Ridge to the tip of the peninsula and includes the larger division of the Long Point NWA. In the east, there are sandy beaches with sparse vegetation along the south shore of the peninsula, but inland areas are covered by a series of alternating dune ridges and wetland swales formed by fluctuating lake levels. Newly formed dunes are initially colonized by beachgrass (*Ammophila* spp.); as the dunes become older, more stable, and drier, savannas and open grassland of cottonwood (*Populus deltoides*) and red cedar (*Juniperus virginiana*) develop eventually climaxing to woodland forests. Well developed forest ridges are covered with white pine (*Pinus strobus*), eastern white cedar (*Thuja occidentslis*), red oak (*Quercus rubra*), white birch (*Betula papyrifera*); a variety of grasses and herbaceous plants including Hound's tongue (*Cynoglossum officinale*) and butterfly milkweed (*Asclepias tuberosa*); and boreal species such as starflower (*Trientalis borealis*), yellow lady's slipper (*Cypripedium calceolus*), and early coral root (*Corallorhiza trifida*). The oldest ridges are covered with red oak (*Quercus rubra*), sugar maple (*Acer saccharum*), white oak (*Quercus alba*), red maple (*Acer rubrum*), black cherry (*Prunus serotina*), and southern species such as chinquapin oak (*Quercus muehlenbergii*) and sassafras (*Sassafras albidum*). Between the ridges are marshes, ponds, and swales with wet meadow communities containing sedge and rush vegetation, and lowland forests containing tamarack (*Larix laricina*), white pine (*Pinus strobus*), white birch (*Betula*), and white cedar (*Tabebuia heterophylla*) (CWS, 1983: McCracken *et al.*, 1981). The different types of wetland communities and associated vegetation species are discussed in more detail in Section 2.2.

2.1.6 WILDLIFE

The unique environment of Long Point combined with diverse vegetation and warm climate provides exceptional habitat for many wildlife species. The shallow waters of the Inner Bay are highly productive providing food and shelter for over 330 species of birds; 12 of which are provincially significant (Nelson and Wilcox, 1996; Koshida, 1988). Long Point is an important refuge and staging area for many of North America's shorebirds and migratory waterfowl. Centrally located along the Atlantic flyway between the Atlantic and Gulf coast wintering areas and the Prairie and Arctic breeding areas, Long Point is an ideal stopover location for many migratory species (Long Point Bird Observatory, 2001). Long Point is the most important staging area within the lower Great Lakes for the redhead (*Aythya americana*) and canvasback (*Aythya valisineria*) duck and to approximately 30,000 tundra swans (*Cygnus columbianus*) (CWS, 1983). Long Point is also an important staging area for the ring-necked (*Aythya collaris*), greater and lesser scaup (*Aythya marila, A. affinis*), bufflehead (*Bucephala albeola*), common goldeneye (*Bucephala clangula*), mallard (*Anas platyrhynchos*), and American black duck (*Anas rubripes*).

In addition to the waterfowl, many species of shorebirds, gulls, and raptors utilize Long Point during migration (CWS, 1983). Common species include the herring gull (*Larus argentatus*), ring-billed gull (*L. delawarensis*), Bonaparte's gull (*L. Philadelphia*), sharp-shinned hawk (*Accipiter striatus*), and saw-whet owl (*Aegolius acadicus*). Although Long Point is renowned worldwide as a stopover and staging area for many migratory species, there are over 170 species of birds with nesting records, or are suspected of breeding, at Long Point (Nelson and Wilcox, 1996). Rare and endangered breeding species include the Forester's tern (*Stema forsteri*), king rail (*Rallus elegans*), prothonotary warbler (*Protonotaria citrea*), piping plover (*Charadrius melodus*), and the recently reintroduced bald eagle (*Haliaeetus leucocephalus*).

The sheltered wetland environment and shallow waters at Long Point also provide important spawning and nursery habitat for many of Lake Erie's fish species including the largemouth and smallmouth bass (*Micropterus salmoides, M. dolomieu*), pike (*Esox*), smelt (*Osmerus*), yellow perch (*Perca flavescens*), rainbow trout (*Oncorhynchus mykiss*), coho salmon (*Oncorhynchus kisutch*), and several rare and threatened species including the lake chubsucker ((*Erimyzon sucetta*), yellow bullhead (*Ameiurus natalis*), spotted gar (*Lepisosteus oculatus*), pugnose shiner (*Notropis anogenus*), and tadpole madtom (*Noturus gyrinus*).

The diverse natural environment of Long Point is home to 46 species of mammal. Common species include the northern long-eared bat (*Myotis septentrenalis*), southern flying squirrel (*Glaucomys volans*), white-tailed deer (*Odocoileus virginianus*), muskrat (*Ondatra zibethicus*), badger (*Taxidea taxus*), raccoon (*Procyon lotor*), woodland and meadow vole (*Microtus pinetorum*, *M. pennsylvanicus*), least shrew (*Crypotis parva*), eastern pipistrelle (*Pipistrellus subflava*), deer mouse (*Peromyscus maniculatus*), white-footed mouse (*Peromyscus leucopus*), coyote (*Canis latrans*), and the gray and red fox (*Urocyon cinereoargenteus, Vulpes vulpes*).

There are 18 species of amphibians and 16 species of reptiles found at Long Point, many of which are rare or threatened including the spotted turtle (*Clemmys guttata*), Blanding's turtle (*Emydoidea blandingii*), eastern spiny softshell turtle (*Apalone spinifera spinifera*), Fowler's toad (*Bufo woodhousii fowleri*), eastern hognose snake (*Heterodon platyrhinos*), and eastern fox snake (*Elaphe vulpina gloydi*). Although research regarding invertebrate fauna at Long Point is limited, the area is important to the monarch butterfly (*Danaus plexippus*) during their autumn migration and to one of Canada's rarest invertebrates, the meadow crayfish (*Cambarus diogenes*) (CBRA, 1999).

2.2 Lake Erie Coastal Wetlands

Approximately one third of the wetlands within the Great Lakes are located along the shores of Lake Erie. The favourable climate, variable water levels, shallow sedimentary nearshore, and the dynamic process of erosion and deposition promote the development of wetlands (Nelson and Wilcox, 1996). In Canada, wetlands can be classified as five main types: bogs, fens, swamps, marshes, and shallow open water areas. Classification is based on the biotic and abiotic characteristics of the wetland, such as the type of vegetation, hydrology, soil type, acidity or alkalinity of the soil, and amount of peat (Herbert, 2002; GLIN, 1998a; National Wetlands Working Group, 1987). These five wetland classes can be further divided into 70 different geomorphic forms depending on the surface morphology of the wetland, presence of certain features in the landscape, position in the landscape, and proximity to water bodies; and, each of these forms can be characterized into different wetland types based on the form and structure of the vegetation (Herbert, 2002).

Bogs are poorly drained wetlands covered with mats of moss that eventually form layers of peat. In fact, peat accumulation in bogs is substantial (Herbert, 2002). Bog soils are wet, acidic, and nutrient poor. Bogs are dominated by Sphagnum moss but sedges (*Carex*), low growing shrubs, and some species of trees such as spruce (*Picea*) are also found (Newmaster *et al.*, 1997). Bogs are located along the landward margin of coastal wetlands, or occur within wet meadows where the water table is high, but can occur as floating mats in areas where the vegetation has adapted to fluctuating water levels. Bogs are typically found in the upper Great Lakes, although pockets of bogs can be found along Lake Erie (GLIN, 1998a).

Fens are located in periodically flooded areas above the water level. Fens consist of layers of poorly or moderately decomposed peat; the soil contains more nutrients and is generally less acidic compared to bogs (Newmaster *et al.*, 1997). Fens are diverse and dominated by sedge vegetation. Moss, reeds, grasses, and low to medium high shrubs are common. Although species of tamarack and cedar can be found, trees are sparse (Herbert, 2002; GLIN, 1998a). Fens are not common in the coastal wetlands of the Great Lakes, although some fens can be found in the northern Great Lakes.

Swamps are wooded marshes that are associated with streams, rivers, or lakes. Swamps occur on soils that are saturated for most, if not all, of the growing season and support herbaceous plants, mosses, tall shrub thickets, and dense coniferous and deciduous forests (Herbert, 2002). Generally intolerant of the wide water level fluctuations typical of the Great Lakes, swamps are located along the landward margin of the shore. Swamps are typically isolated from the lake and therefore only influenced by the lake during periods of high lake levels (GLIN, 1998a; Newmaster *et al.*, 1997).

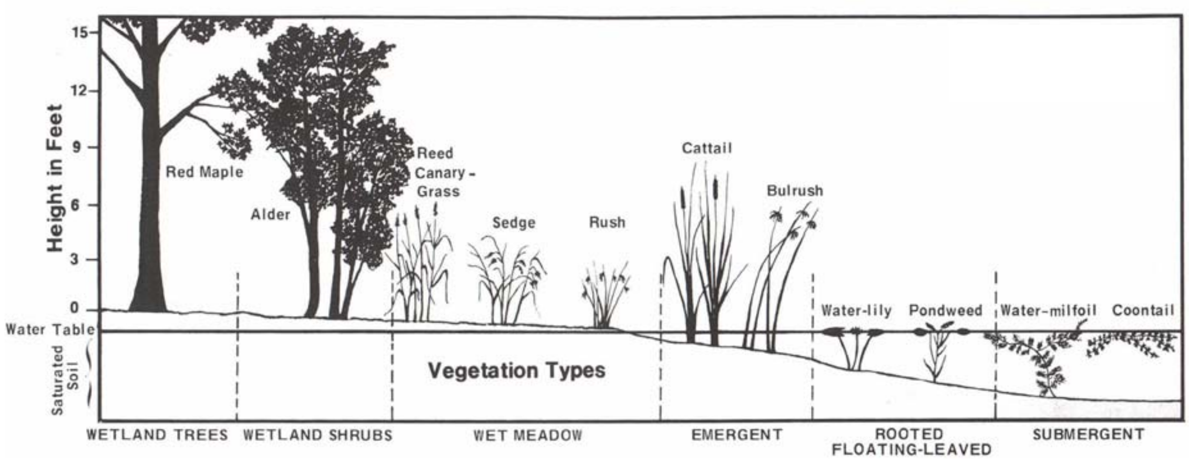
Marshes are the most common type of wetland along the shores of the Great Lakes. Marshes occupy the area that ranges from moist or saturated areas above the water level to shallow water areas; the maximum depth marsh vegetation occurs at is 1.5 m. Marsh vegetation is highly tolerant of water level fluctuations within the lake system and as a result, the vegetation occurs along distinct zones related to water depth gradients (GLIN, 1998a). Emergent vegetation is dominant and occupies drier zones in the marsh; submergent vegetation occupies areas of standing or slow moving water. Tall reeds and rushes, grasses and sedges, broad-leafed and floating-leaf plants are typical marsh vegetation (Herbert, 2002; Newmaster *et al.*, 1997). The soil in marshes is composed of organic material or sand and lacks a distinct peat layer.

Shallow open water areas are transitional zones between the lake and marsh environments. Shallow open water areas occur along lakes where the “mid-summer depth of the water is less than two metres and open expanses of water comprise at least 75 % of their area” (Herbert, 2002). Floating and submergent vegetation are common. Shallow open water areas are highly productive and the dense vegetation provides food and shelter for many species of amphibians, reptiles, crustaceans, and fish (Herbert, 2002). The soil of shallow open water wetlands is mucky or composed of mineral and organic material (Newmaster *et al.*, 1997).

The Long Point wetland complex is characterized by marsh and shallow open water wetlands. The wetland complex contains a variety of geomorphic coastal wetland forms including dune (ridge) and swale wetlands in the outer peninsula, a diked wetland in the Big Creek NWA, a small amount of delta and riverine wetlands in the Big Creek River Channel, lagoon and barrier wetlands in the Big Creek Marshes and embayed and shoreline wetlands in the Inner Bay. Although most of Long Point is classified as a marsh wetland, these geomorphic forms do influence how the wetland vegetation responds to water level fluctuations. Long Point also contains several different wetland vegetation types or communities. The wetland vegetation communities typically associated with marsh wetlands are outlined below. The types of plant species generally found within each of the communities and the arrangement of the communities within the wetland are discussed. Then the role of fluctuating water levels within coastal wetlands, the responses of the wetland vegetation communities to fluctuating water levels, and the resulting impacts on the wetlands are examined.

2.2.1 WETLAND VEGETATION CONTINUUM

A typical marsh wetland, such as those that occur at Long Point, consists of several communities or zones of vegetation that respond to different moisture conditions along an elevation (or slope) gradient. The vegetation communities are distributed along water depth gradients reflecting the different tolerances of the dominant aquatic plants to flooding conditions (Grosshans and Kenkel, 1997). The wettest community in the wetland vegetation continuum is the open water community. Open water vegetation includes submergent and floating-leaved plants that occupy deeper water areas along the shoreline. Emergent vegetation occupies shallower areas along the shore. As the moisture conditions in the continuum become drier, the wetland progresses from the emergent vegetation to grass and sedge communities then to wet meadows and treed communities, the driest community within the wetland continuum (Figure 2-2).



High water levels: landward shift of vegetation communities

Low water levels: lakeward shift of vegetation communities

FIGURE 2-2: CROSS SECTION OF A TYPICAL MARSH (Bolsenga and Herdendorf, 1993)

Wetlands are highly dynamic systems. The vegetation communities are continually shifting, expanding, and contracting along the continuum in response to seasonal and annual water level fluctuations. Generally, as lake levels rise, there is a landward migration of the wetland communities along the gradient.

Vegetation communities with limited inundation tolerances respond by shifting their distribution towards the dry end of the moisture gradient (Bauder, 2000). Steep slopes, dikes, and other man-made structures can, however, impede the natural landward succession of wetland communities during periods of high water levels (Koshida, 1988; Geis, 1985). Conversely, as lake levels decline, the vegetation communities migrate towards the lakeward extent of the wetland. The types of vegetation communities within the wetland continuum are described.

2.2.1.1 Submergent Vegetation

Submergent vegetation are rooted or non-rooted aquatic plants that occur in shallow to moderate depth water. The maximum depth in which submergent vegetation occurs in open water areas is two metres, although the vegetation can grow in deeper areas depending on water clarity. The vegetative growth of the plants, except for the flowering parts of some species, lies entirely beneath the water surface (Figure 2-3). Submergent vegetation are highly adaptive to water level fluctuations. Submergent vegetation has developed three important characteristics that allow the plants to respond to fluctuating water levels. First, submergent vegetation can move easily along the surface of the water in response to water level fluctuations because the plants are thin and flexible (Delesalle, 1998). Second, the compounded structure and cluster arrangement of the plant leaves are very efficient at gathering light for photosynthesis, even within turbid water caused by fluctuating water levels (Koshida, 1988). Third, the seeds of submergent plants remain viable for extended periods of dry conditions; as wet conditions return, the propagules germinate underwater and the plants reestablish themselves within deeper water zones (van der Valk and Davis, 1978; Harris and Marshall, 1963).

Many wildlife species, particularly waterfowl, depend on submergent vegetation as a source of food. Typical submergent species include:

- pondweed (*Potamogeton epihydrus, P. gramineus, P. pectinatus, P. pusillus*);
- coontail (*Ceratophyllum demersum*);
- water milfoil (*Myriophyllum* spp.);
- wild celery (*Vallisneria americana*);
- waterweed (*Elodea canadensis*);
- muskgrass (*Chara vulgaris*);
- naiads (*Najas*);
- bladderworts (*Utricularia*); and,
- smartweed *(Polygonum coccineum*).

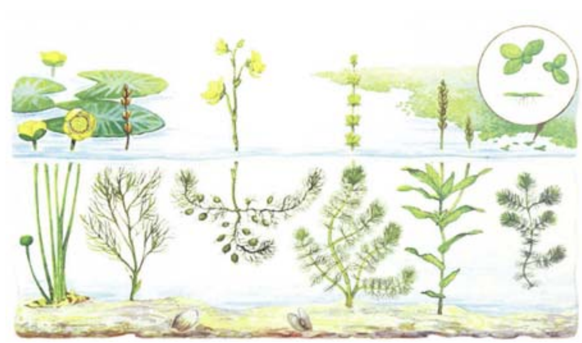


Figure 2-3: Submerged and Floating-Leaved Vegetation (Delesalle, 1998)

2.2.1.2 Floating-Leaved Vegetation

Floating-leaved vegetation is defined as aquatic plants with vegetative growth that floats on the surface of the water. Floating-leaved vegetation occurs within protected areas in the lake, such as in bays or within stands of emergent vegetation, along the shoreline (Northern Environmental, 2001). The adaptive responses of floating-leaved plants to water level fluctuations are similar to those of submergent vegetation. The broad leaf structure of the plants promotes photosynthesis at the water surface, and the slender and flexible stems of the plants allow easy transport of nutrients between the leaves and the massive tubers (Koshida, 1988).

There are two types of floating-leaved vegetation: free-floating or floating. Free-floating plants are non-rooted and the entire plant floats freely on the surface of the water; growth of these plants is not limited by water depth. Duckweed (*Lemna*) is a common example of a free-floating plant. Floating plants are rooted and have long flexible stems and leaves that float horizontally on the water surface. Typical plants include the white water lily (*Nymphaea odorata*), yellow pond lily (*Nuphar variegatum*), and some species of pondweed (*Potamogeton natans*). Floating-leaved vegetation is an important component of the wetland system as the vegetation supports many invertebrate and vertebrate species (Bayly, 1979b).

2.2.1.3 Emergent Vegetation

Emergent vegetation are aquatic plants rooted in shallow water and the vegetative growth emerges above the surface of the water (Figure 2-4). The base of the plant is either temporarily or permanently flooded. Emergent plants are tolerant of fluctuating water levels; the plants can survive short periods of drought, but thrive in moist conditions. Short periods of dry conditions provide for a dynamic and productive environment and are necessary for reestablishing young plants and maintaining healthy stands of emergent vegetation. Seeds from the plants spread during periods of standing water, and remain viable for years or decades until natural draw down occurs where the seeds can then germinate on exposed mudflats or in very shallow water (van der Valk and Davis, 1978). The plants propagate by rhizomes (underground roots/tubers), during periods of high water levels. Long periods of flooding, however, will result in the dieback of emergent vegetation (Delesalle, 1998).

Figure 2-4: Emergent Vegetation
(Delesalle, 1998)

There are three types of emergent vegetation: robust, broad-leaved, and narrow-leaved emergent. Robust emergent vegetation occupies deeper water depth gradients than broad and narrow-leaved emergent vegetation. These plants are sturdy and can grow between 1.5 and 3 m in height. Robust emergent vegetation include cattail (*Typha*), bulrush (*Scirpus acutus*), and reed grass (*Phragmites*). Broad-leaved emergent vegetation grows to heights of less than one metre and includes pickerelweed (*Pontederia cordata*), arrowheads (*Sagittaria*), and beggar's ticks (*Bidens*). Narrow-leaved emergent vegetation occupies moist or seasonally flooded areas and grows to less than 1.8 m in height (OMNR, 1984). Examples of narrow-leaved emergents are wild rice (*Zizania palustris*), burreed (*Sparganium*), sedge (*Carex*), cut grass (*Leersia oryzoides*), bluejoint (*Calamagrostis canadensis*), and canary grass (*Phalaris arundinacea*). Emergent vegetation within Long Point generally occurs in a mosaic of monodominant stands of individual species (Bolsenga and Herdendorf, 1993), but nevertheless, provides an important source of food for waterfowl, and nesting and habitat cover for other wildlife.

2.2.1.4 Meadow and Shrub

Meadow and shrub are herbaceous and woody vegetation less than six metres in height. Meadows contain non-woody and herbaceous grass and sedge vegetation; shrubs include woody vegetation with dense foliage and several stems growing to heights of less than one metre and taller woody plants with distinct crowns and trunks that grow from one to six metres in height (OMNR, 1984). Twigrush (*Machaerina),* goldenrod (*Solidago*), spikerush (*Eleocharis*), beakrush (*Rhynchospora*), and nutrush (*Scleria*) are common examples of wet meadow vegetation. Sweet gale (*Myrica gale*) is an example of a short shrub, and dogwood (*Cornus*), alder (*Alnus rugosa*), and buttonbush (*Cephalanthus occidentalis*) are examples of taller shrubs occupying this wetland community. Meadow and shrub is one of the least diverse communities within the wetland system, but provides important nesting areas for waterfowl and food and shelter for other wildlife. The quality and usefulness of the habitat is, however, severely diminished during periods of high water levels (ILERB, 1981).

2.2.1.6 Treed

Wetland trees are large perennial woody plants greater than six metres in height and have a main trunk with branches that form a crown (OMNR, 1984). The outer extent of the treed community is marked by the presence of deciduous trees that are tolerant of periodic flooding within the floodplain and lack of coniferous trees, which are common in upland forests (Geis and Kee, 1977). The boundary between unsaturated and saturated conditions at the surface of the soil during the late spring and summer when water levels are the highest has also been used to mark the boundary between the treed and upland community (Quinlan, 1985). Examples of treed vegetation include white pine (*Pinus Strobus*) and red oak (*Quercus rubra*), which were logged at Long Point during the 1950s (Koshida, 1988), willow (*Salix*), ash (*Fraxinus*), tamarack (*Larix laricina*), spruce (*Picea*), white oak (*Quercus alba*), red maple (*Acer rubrum*), sassafras (*Sassafras*), and black cherry (*Prunus serotina*) (OMNR, 1984). Fire, deer grazing, storms, and fluctuating water levels has limited the development of dense stands of treed vegetation in Long Point, but this community does provide habitat for many species of breeding birds and mammals (Catling and Reznicek, 1981).

2.2.1.7 Wetland Community Tolerance Ranges

The response of wetland vegetation to fluctuating water levels depends on the magnitude, frequency, timing, and duration of the water level fluctuation and the tolerance depths of the vegetation communities

(ILERB, 1981). The optimal water depth tolerance ranges for the wetland vegetation communities are summarized in Table 2-1. In addition to the wetland vegetation communities, two land cover types were included at the upper and lower water depth extents in the tolerance chart. Although these land cover types are not actual wetland vegetation communities per se, lake and upland are important communities associated with wetland areas. The tolerance depth ranges for the wetland communities were synthesized from the literature (United States Environmental Protection Agency, 2000; Newmaster *et al.*, 1997; Ould and Holbrow, 1987; Geis, 1985; Kadlec and Wentz, 1974; Dane, 1959). A detailed chart summarizing the tolerance ranges of individual plant species is provided in Appendix A.

TABLE 2-1: WETLAND COMMUNITY TOLERANCE RANGES

WETLAND COMMUNITY	WATER DEPTH RANGE (CM)*
Lake	> 200
Open Water/Submergent	60 to 200
Floating Emergent	30 to 60
Emergent	-30 to 30
Tall Emergent	-30 to –50
Meadow	-50 to –80
Treed	- 80 to –100
Upland	< -100

* Negative "water depth" values indicate height above lake level in centimetres (cm)

Vegetation response is a function of water depth but the composition of the vegetation within a community is a function of soil or substrate type, the reservoir of buried seeds in the seed bank, slope, wave action, and water chemistry. Fire and inter-specific competition between plant species along the gradient is also important in determining vegetation composition (Grosshans and Kenkel, 1997; Keddy and Reznicek, 1985). Now that the wetland vegetation communities have been characterized in detail, it is important to explore the impacts of water level fluctuations on these communities.

2.2.2 THE ROLE OF FLUCTUATING WATER LEVELS IN COASTAL WETLANDS

Water level fluctuations are essential for the long-term health of coastal wetland ecosystems. Periods of drought and flood conditions reflect natural climatic variation and are vital for maintaining diversity and productivity within wetlands by promoting constant rejuvenation of the wetland plant communities (Grosshans and Kenkel, 1997). Without natural disruptions to the water level regime of a wetland system, the wetland would grow towards a climax state dominated by denser emergent and terrestrial vegetation; vegetation that is not optimal for many wetland dependent wildlife. Wetlands require periodic disturbances to set back succession and restore earlier productive stages within the wetland (ILERB, 1981). In fact, many wetland systems have evolved over hundreds or thousands of years, and certain aquatic vegetation has adapted to the natural fluctuations to the point where the plants require fluctuations for continued survival and seed production (Harris and Marshall, 1963, 341). Wetlands experience (1) short-term water level fluctuations; (2) seasonal water level fluctuations; and (3) long-term water level fluctuations.

Short-term water level fluctuations are driven by wind or pressure generated events. Storm surges and seiches can temporarily alter lake levels as much as two metres within a couple of hours (Bolsenga and Herdendorf, 1993; Hartley and Barnes, 1981; Herdendorf *et al.*, 1981). The impacts of storm surges and seiches on wetlands are minimal, but severe events can damage or destroy wetland vegetation communities and erode or physically alter the shoreline and beach environment.

Seasonal fluctuations are due to annual changes in the amount of runoff and evaporation in the hydrologic cycle (Lawrence and Nelson, 1994). Snowmelt and spring precipitation contribute to higher lake levels in the late spring and summer; Lake Erie water levels are at a maximum in June. The pattern of rising water levels during the spring and early summer months promotes well-balanced interspersion of emergent and open water plant communities within the wetland. Waterfowl prefer these hemi-marsh conditions in the wetland, which are produced by stable water levels during the spring and early summer (Koshida, 1988). During the summer, lake levels decline due to increased rates of evaporation and less runoff. Lake levels continue to decline through the winter months until early spring (Koshida, 1988). Seasonal fluctuations range from 0.3 to 0.6 m (Lawrence and Nelson, 1994).

Seasonal fluctuations promote wetland productivity by restoring and redistributing nutrients essential to maintaining wetland communities (Delesalle, 1998). Areas of the wetland that experience the most water level fluctuation during the year have been noted to have more diverse vegetation zones with more diverse species within the zones than areas that experience less seasonal fluctuations (Keddy and Reznicek, 1986). Changes in the timing and magnitude of seasonal fluctuations can have serious ramifications on the distribution of wetland vegetation and wildlife that use the wetland. Vegetation response to seasonal fluctuations depends on the life cycle and physical structure of the plants. Annual vegetation that can complete their life cycle rapidly can respond to fluctuating water levels, whereas perennial plants must tolerate fluctuating conditions (Keddy and Reznicek, 1986).

Long-term water level fluctuations are driven by the interaction between precipitation, land and tributary storage, evaporation, and the hydrological characteristics of the upper Great Lakes and Lake Erie (CWS, 1983). Long-term water level fluctuations occur inter-annually over a multiple of years. Long-term fluctuations can vary lake levels by up to four metres (Lyon and Drobney, 1984). Long-term water level fluctuations are important for healthy wetland systems. Long-term fluctuations maintain wetland diversity and promote a more productive wetland compared to wetlands with stable water level conditions (GLIN, 1998a). Persistent long-term water level conditions can, however, cause stress and competition of wetland vegetation and thus produce less suitable habitat for wetland dependent wildlife (ILERB, 1981). Wetlands that experience persistent lake levels for longer periods of time can experience dramatic long-term changes in the wetland's soil structure, water chemistry, and plant and animal communities (Delesalle, 1998, 16). Wetland vegetation dies back or is eroded and the vegetation communities are displaced latterly along the wetland continuum (Bolsenga and Herdendorf, 1993; Herdendorf *et al.*, 1981). The survival of some species of vegetation during persistent long-term water levels depends solely on buried seeds, while other species temporarily exploit the changed environmental conditions (Keddy and Reznicek, 1986). The response of wetland vegetation communities to long-term water level fluctuations is discussed in greater detail in the following section.

2.2.3 THE RESPONSE OF WETLAND VEGETATION TO LONG-TERM FLUCTUATIONS

Water level fluctuations due to natural climatic variability initiate vegetation succession in wetland systems. Wetland succession occurs in irregular but natural cycles that last in length from 5 to 30 years (van der Valk and Davis, 1978). The cycle involves four stages of water level conditions (Figure 2-5). During periods of stable water levels where there are small seasonal variations, the marsh remains relatively productive and emergent vegetation growth is promoted. However, a prolonged period of stable

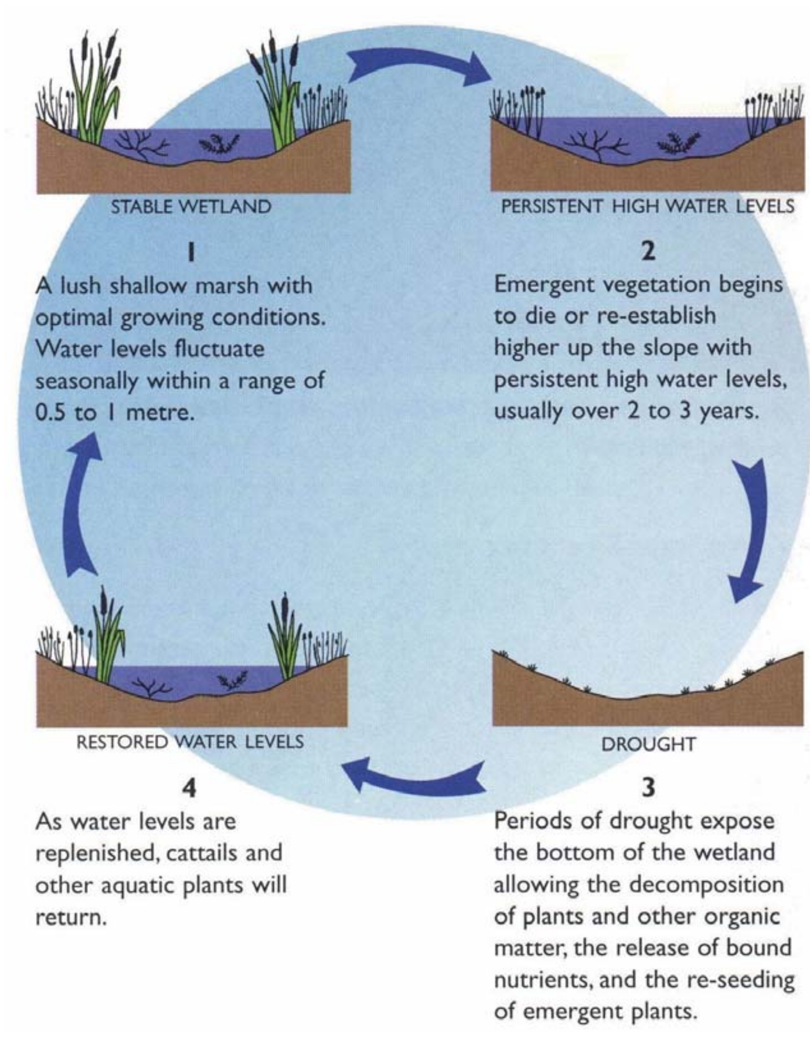


FIGURE 2-5: NATURAL CYCLE OF WETLAND SUCCESSION (Delesalle, 1998)

conditions with a reduced range of water level fluctuations reduces the amount of wetland area and encourages dense emergent growth (ILERB, 1981). Hence, the complexity, biodiversity and productivity of the wetland diminish and the system enters a state of degeneration or stagnation (Grosshans and Kenkel, 1997).

During periods of persistent high water levels, herbaceous and woody plants in the wet meadow and dominant emergent vegetation dieback but submergent and free-floating vegetation thrive in the deeper water. The high water levels initially stimulate growth in emergent vegetation but once the tolerance depth of the plants is exceeded the vegetation dies back (Poiani and Johnson, 1991). During periods of persistent low water levels, aquatic species recede and competitively dominant species in the wet meadow and emergent vegetation reestablish on exposed mudflats from the seed bank. As water levels are restored in the marsh, submergent and free-floating species germinate while annual wet meadow and emergent growth on the mudflats is eliminated (GLIN, 1998a; Poiani and Johnson, 1991; Keddy and Reznicek, 1986; van der Valk and Davis, 1978). The effects of persistent long-term water level fluctuations and alterations to the timing of seasonal fluctuations that may result from human-induced climate change and the implications of these changes to the wetland system are discussed.

2.2.4 THE EFFECTS OF WATER LEVEL FLUCTUATIONS ON WETLAND SYSTEMS

The timing, direction, magnitude, and intensity of water level fluctuations influences the physical and chemical conditions of the wetland, the composition and distribution of vegetation within the wetland, and ecology of the wetland in relation to wetland dependent wildlife. The effects of long-term high and low water level conditions and abnormal seasonal water level fluctuations are discussed.

2.2.4.1 PERSISTENT HIGH WATER LEVELS

Periods of rising water level conditions promote the expansion of the open water and floating-leaved communities landward and the contraction of emergent vegetation, meadow, and treed communities (van der Valk *et al.*, 1994; Jaworski and Raphael, 1976). As water levels rise, hemi-marsh conditions are produced in the wetland; these conditions occur when the wetland is equally composed of marsh vegetation and open water. Hemi-marsh conditions benefit many wildlife species including waterfowl, muskrats, black terns, and herons. There is also an increase in wildlife species diversity as habitat conditions are improved for invertebrates, reptiles, and amphibians and there are more opportunities for fish spawning, rearing, and forage (ILERB, 1981).

As water levels remain persistently high, emergent vegetation becomes less dense allowing waves to break closer to the shoreline and increasing erosion and turbidity in the nearshore (Jaworski and Raphael, 1976). Erosion is beneficial for maintaining open channels through the wetland and also can improve the interchange of water between the lake and wetland system. The water quality of the wetland improves; the water has higher pH values and concentrations of carbonate and dissolved oxygen, the aerobic surface decreases allowing phosphorus, iron, and manganese to be released in the water, water temperatures lower, and the soil becomes more acidic (Burton, 1985; Patterson and Whillans, 1985). These changes affect the species composition of submergent and floating-leaved vegetation communities and promote healthier fish habitat in the short term (Jaworski and Raphael, 1976, 291). Prolonged periods of increased erosion and turbidity will, however, reduce the amount of light penetrating the water leading to a decline in the growth of submergent vegetation and subsequently a decline in fish populations in the wetland (Koshida, 1988).

Persistent periods of high water levels reduce the amount of edge habitat for wildlife. Furthermore, the loss of emergent vegetation decreases the amount of viable habitat for small mammals and invertebrates. Emergent vegetation and invertebrates are the main staple for many waterfowl and marsh birds, and as the availability of food becomes scarce there are observed shifts in the bird communities that utilize the wetland. Prolonged high water level conditions kill woody trees, shrub vegetation and competitively dominant emergent species, such as *Typha*, which will dieback or form floating colonies as a result of increased turbidity and siltation (Keddy and Reznicek, 1986; Jaworski and Raphael, 1976). Subsequently, the marsh area increases and there are more areas within the wetland that can be occupied by non-woody herbaceous plants (Keddy and Reznicek, 1986). During extended periods of high water levels, however, there is a progressive loss of floating and rooted aquatics resulting in the eventual decline in wetland area (Koshida, 1988). Finally, persistent high water level conditions can also compound the damage caused by storms (GLIN, 1988b).

An example of the effects of rising water levels at Long Point is provided. Between 1966 and 1971, Lake Erie water levels increased 50 cm. During this period, invertebrates were flushed out from feeding areas and feeding by migratory shorebirds declined. From 1969 to 1973, water levels continued to increase resulting in more open water. The increase of open water led to excellent yields of wild rice and, combined with inland drought, resulted in the increased use of the marsh by dabbling ducks (Koshida, 1988, 94).

2.2.4.2 Persistent Low Water Levels

Declining water levels promote the growth of sedge meadow and dense emergent vegetation communities in the wetland. The diversity of the wetland decreases, as the denser and drier sedge meadow and emergent communities dominate and replace submergent and floating-leaved vegetation and emergent species that are less tolerant of drier conditions. Emergent vegetation occupies shallow water areas; seeds germinate from reserves or buried seeds that are exposed as the water levels recede and increase the amount of emergent vegetation within the wetland (Keddy and Reznicek, 1986). Furthermore, declining water levels allow non-wetland woody shrubs to invade the marsh. The loss of soil moisture in the wetland and increase in drier vegetation increase the risk of fire. Fire can be disruptive to coastal wetlands where seeds are buried higher in the soil (Keddy and Reznicek, 1986), but can also be beneficial to the wetland. Fire consumes organic material thus releasing nutrients into the soil and reversing vegetation succession (Koshida, 1988).

The extent of deeper water marsh communities decreases, but some submergent and floating vegetation remain in pools of standing water. The loss of submergent and floating-leaved plants decreases the amount of viable fish and wildlife habitat and food for migratory waterfowl (Koshida, 1988). Furthermore, the lower water levels disrupt wildlife habitat corridors. Declining water levels lead to a decrease in the primary productivity of the wetland (Patterson and Whillans, 1985). The wetland is composed of simpler vegetation communities with fewer species (CWS, 2002). As wildlife species diversity decreases and habitat conditions decline, the wetland becomes more favourable for opportunistic species including red-winged blackbirds, short-billed marsh wrens, blue-winged teal, rails, mallards, muskrats, dabbling ducks, white-tailed deer, cottontail rabbits, and small rodents (ILERB, 1981; Jaworski and Raphael, 1976).

Low water levels reduce wave energy and therefore erosion and turbidity in the nearshore decreases (Koshida, 1988). The sediment is exposed to aerobic conditions increasing the rates of oxidation and leading to the faster decomposition of organic materials. Nutrients transported by currents and wave action during higher water levels are deposited and become trapped under a layer of aerobic soil that develops on the wetland substrate during low water level periods; the nutrients are released from the sediments during periods of inundation. The water temperature increases and salinity levels rise (Koshida, 1988; Keddy and Reznicek, 1986; Burton, 1985).

2.2.4.3 Abnormal Seasonal Fluctuations

Abnormally high water levels in the winter can cause extensive ice damage if it is cold enough for ice to develop. The wetland surface freezes as thick ice sheets form closer to the shore, which can lead to extensive bottom lifting and erosion of sediments and vegetation in the wetland substrate. During spring thaw, sections of the shoreline can break off and move with the ice as it moves lakeward; this can be especially disruptive with flooding and refreezing in the spring (Geis, 1985). High winter water levels also lead to shoot suffocation of emergent plants therefore causing massive die-off of plants, especially *Typha* (Koshida, 1988). Low winter water levels, combined with low spring levels, promote the germination of emergent vegetation earlier in the spring and summer months resulting in denser emergent growth during the later summer months. Ice damage due to low water levels can also be extensive as moving ice scours the wetland substrate and uproots vegetation. Extremely low water levels allow frost to penetrate the sandy and mucky substrate, which can uplift the frozen substrate during periods of inundation (Koshida, 1988). The frozen roots and beds result in a severe decrease or loss of muskrat food and shelter. Therefore, muskrat populations are reduced due to starvation, disease, and predation (Mortsch, 1998). In addition, frozen shallow water leads to oxygen depletion and increased mortality of fish trapped under the frozen surface. Higher mortality rates of reptiles and amphibians are also expected as the shallow waters cause freezing conditions for hibernating species (Koshida, 1988, 54).

Lower than normal spring water levels may cause a shift in terrestrial vegetation closer to the shore, thus resulting in a loss of nearshore fish spawning and nursery ground. Lower spring levels may prevent fish from reaching spawning habitat or lead to the exposure of fish eggs in the water. Higher spring water levels result in more wetland substrate uplifting and greater sediment exposure. Abnormally low summer water levels promote the invasion of emergent vegetation, which cause more edge effects during periods of flooding and leads to declines in muskrat populations (GLIN, 1998b; Koshida, 1988; Geis, 1985).

2.2.5 IMPLICATIONS OF FUTURE CLIMATE CHANGE

As noted earlier, an enhanced Greenhouse Effect due to human-induced climate change will contribute to warming in the Great Lakes region. Projected climatic changes, the implications of these changes to water levels and the resulting impacts on the wetland systems are outlined for two possible scenarios: an increase in the frequency and duration of low water levels, and alterations to the seasonal distribution of water levels.

2.2.5.1 INCREASED FREQUENCY AND DURATION OF LOW WATER LEVELS

Climate scenarios are projecting that the frequency and duration of low water levels will increase with enhanced global warming. The Great Lakes basin is projected to experience (1) an increase in mean annual temperatures, leading to increased rates of evapotranspiration and (2) a decrease in summer precipitation. Consequently, these changes will reduce the amount of runoff entering the Great Lakes basin leading to a decline Lake Erie water levels (Mortsch, 1998). Persistent longer-term low water levels may negatively affect vegetation and wildlife species at Long Point.

Vegetation communities within the wetland complex may succeed to drier vegetation communities. The extent of submergent, floating-leaved, and wetter emergent vegetation communities in the wetland may decline or migrate lakeward. The spit may likely confine the lakeward migration of drier wetland vegetation communities as the sandy substrate may limit the growth of plant species (Wall, 1998). The extent of open water vegetation is expected to decrease as drier vegetation communities consisting of sedges, grasses, shrubs, and trees expand (Mortsch, 1998). As the water level recedes, exposed mudflats may be colonized by emergent vegetation thus resulting in some wetland expansion, but these areas will eventually tend to drier and denser emergent vegetation (Burkett and Kusler, 2000). Upland vegetation along the landward edge of the wetland may migrate into the wetland, and terrestrial plants may become more abundant and dominate drier areas of the wetland (Burkett and Kusler, 2000; Wall, 1998). The extent of the wetland may decline and, subsequently, the natural diversity of the vegetation and the utility of the wetland to wildlife species may decrease. The availability of food and shelter may become scarce for many wildlife species, particularly for waterfowl that use the wetland for migration, staging, and breeding areas, and there may also be fewer nearshore spawning and nursery areas for fish species. Therefore, competition and stress among wildlife species increases (Mortsch, 1998).

2.2.5.2 ALTERATIONS IN SEASONAL FLUCTUATIONS

Enhanced global warming may alter the seasonal cycle of water. Climate models are projecting changes in the amount of winter precipitation falling as snow to more rainfall. Subsequently, there may be less snow and snow cover in the winter. Warmer winter temperatures combined with a decreased snow base may lead to an earlier snowmelt and rise of water levels during the spring and an earlier onset of seasonal declines during the summer (GLIN, 1998b, Mortsch, 1998). Alterations in the timing of seasonal fluctuations due to projected climate change will affect the vegetation and wildlife within the Long Point wetland system

The alterations in the timing of seasonal fluctuations change the distribution of vegetation species in the wetland, which could have serious implications on waterfowl, mammals, and fish that use the wetland on a seasonal basis, especially during the breeding season when species are most vulnerable to the change. Higher water levels earlier in the spring may result in the massive die-off of emergent vegetation. The productivity of the wetland decreases and wildlife use is limited (Mortsch, 1998). An earlier summer peak and decline in water levels may promote more growth of emergent vegetation from the seed bank compared to a later summer peak (GLIN, 1998b). As emergent vegetation becomes denser, the productivity of the wetland also decreases and the wildlife use of the wetland declines (Mortsch, 1998).

2.2 SUMMARY

A precarious balance exists between the climatic conditions and the natural diversity at Long Point. The coastal wetland contains a rich mosaic of vegetation and landforms that have developed in response to water level fluctuations in Lake Erie due to natural climate variability over the past 4,000 years. Many wildlife species rely on the wetland for temporary or permanent food and shelter. Human-induced climate change threatens to upset this balance. Changes to the climatic system will alter the water level regime of the lake, thus changing the physical, ecological, and socio-economic structure of the wetland complex.

3.0 LITERATURE REVIEW

A review of the literature related to wetlands, geographic information systems analyses, and modelling is provided in this chapter. The current status of wetland research related to climate change and water level fluctuations are discussed in the context of the spatiotemporal and modelling analysis of Long Point. A general overview of the concept of geographic information systems is then provided and the applications of geographic information systems to wetland studies are discussed in terms of identifying and characterizing change in wetland systems. Next, the role of geographic information systems in wetland modelling is examined through a summary of existing wetland models and modelling techniques. Finally, the research niche of this analysis of Long Point is discussed in light of the current status of research.

3.1 WETLANDS AND CLIMATE CHANGE

The effects of climate change on coastal wetlands within the Great Lakes have been little studied (Lyon, 1995). A review of the existing literature is presented here. The first section examines studies that have addressed the issue of climate change at Long Point. The second section reviews studies that have documented the response of wetland vegetation to water level fluctuations in a qualitative manner. The third section discusses displacement models that were developed to characterize the response of wetland vegetation communities to water level changes. A summary of the literature is then provided.

3.1.1 CLIMATE CHANGE AT LONG POINT

Literature reviewing climate change at Long Point is scarce. A study that examined flooding and erosion hazards along the shoreline of Long Point noted that short-term climatic changes led to the record high Lake Erie water levels during 1985 and 1986 (Lawrence and Nelson, 1994). A combination of above average precipitation between 1970 and 1985 and decreased rates of evaporation due to lower atmospheric temperatures resulted in a 12 % increase in the water supply entering the lake. The record high levels caused extensive beach and dune erosion along the shoreline and flooding in low-lying areas. Approximately 50 % of beach area was lost as a result of the high water levels during this period (Lawrence and Nelson, 1994).

Staple (1993) determined the potential changes and implications of climate change at Long Point with a scenario of doubling the carbon dioxide ($2xCO_2$) concentration in the atmosphere. The Canadian Climate Center's (CCC) General Circulation Model (GMC) was used to determine the potential changes to the climate of the Inner Bay. Projected monthly changes were added to the current average monthly temperature and precipitation values, and water levels changes were derived from a hydrological model of the Great Lakes. With the CCC $2xCO_2$ scenario, Staple (1993) projected that winter and spring temperatures could increase as much as 10.5 °C and that summer and fall temperatures could increase by

2.1°C. Annual precipitation was expected to decrease by 6.3 % to 857 mm with a moderate increase during the winter. As a result, Lake Erie water levels were projected to decline 1.35 m. The implications of water level changes for vegetation, wildlife, recreation, and policy issues were then addressed.

Staple (1993) concluded that soil moisture availability in the substrate of Long Point would likely decrease thus limiting vegetation growth. Staple (1993) was unsure, however, whether the substrate type would remain sandy or change to muddy sand from alterations in erosion and depositional patterns due to less runoff and turbidity. There would be a definite shift in vegetation species from deeper water communities to drier communities, but the sandy spit would prevent the marsh from migrating lakeward. Submergent vegetation would likely still persist, but the vegetation's distribution would change and support less wildlife. Long Point would be dominated by marsh meadow and terrestrial vegetation, resulting in a decrease in wetland diversity. Marsh meadow and terrestrial vegetation may increase in the south and northwest areas of the point (Staple, 1993).

Water level declines resulting from climate change were modelled to assess the impact on the shorelines of several wetlands in Western Lake Erie, the Detroit River, and Lake St. Clair, including the Long Point wetland (Lee *et al.*, 1996). Projected lake level declines were combined with bathymetry data to map the areal extent of wetland change between current and altered shorelines. With the CCC 2xCO_2 scenario, Lake Erie water levels were projected to decline 1.48 m from IGLD 1985 levels. As a result, the extent of the Long Point wetland increased quite substantially; the wetland migrated into shallow lake areas along the shoreline and into areas currently open water in the Inner Bay (Figure 3-1).

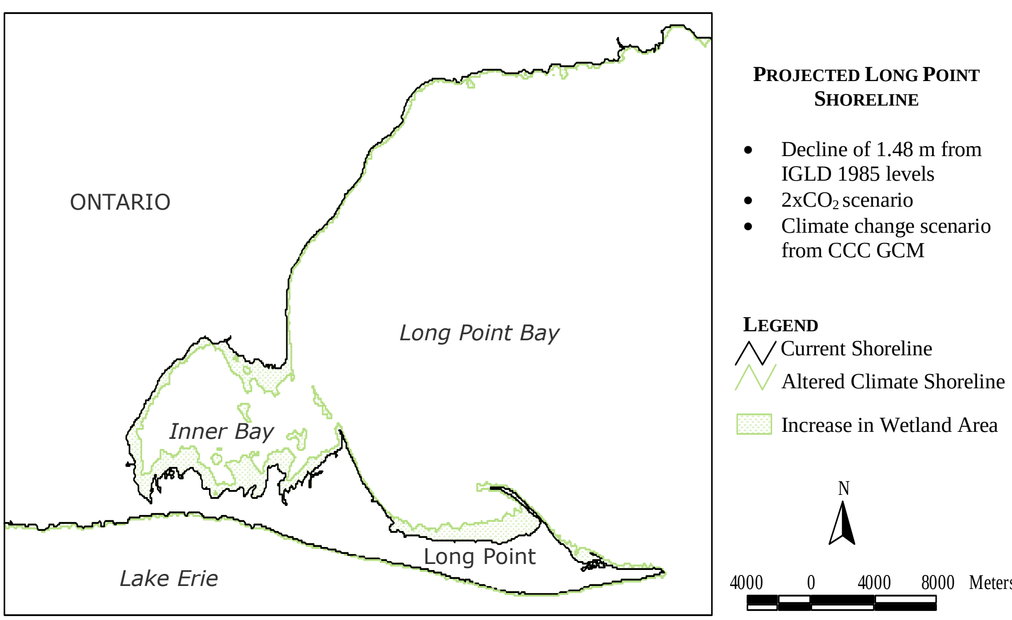


FIGURE 3-1: THE LONG POINT SHORELINE
(Lee *et al.*, 1996)

Few studies attempted to model future climate change at Long Point. But review of the existing literature has indicated that projected climate change, induced by humans, will have a dramatic impact on the wetland complex. The studies have quantitatively assessed projected changes to the climatic system and the implications of these changes on water levels. Changes in temperature and precipitation due to an enhanced global warming may substantially reduce Lake Erie water levels. Such a decline will have ramifications on the physical structure and vegetated composition of the wetland. To assess how the vegetation communities within the wetland may respond to projected declines in water levels, an examination of the literature that has documented the response of wetland vegetation to historical water level fluctuations is required.

3.1.2 EFFECTS OF HISTORICAL WATER LEVEL FLUCTUATIONS ON WETLAND VEGETATION

A number of studies have documented the effects of water level fluctuations on vegetation communities or individual plant species within wetland systems. Bellrose and Brown (1941) studied the effects of fluctuating water levels on emergent vegetation and related the changes to muskrat populations in the Illinois River Valley. Although the primary focus of the study was explaining muskrat declines, the study did observe that emergent vegetation was less abundant in stable water lakes than in semi-stable lakes. Another study examined the causes and effects of emergent vegetation declines in the Pointe Mouillée State Game Area on the eastern shoreline of Lake Erie (McDonald, 1955). Emergent vegetation were not immediately affected by high water levels, but massive and abrupt die-offs in emergent vegetation were noted following wet winters in 1945-46 and 1951-52. Higher water levels between 1943 and 1948 lead to the inland expansion of the marsh, the invasion of *Typha* into wet meadow and suffocation of *Typha* in deeper water, and the loss of barrier beach and treed vegetation along the shoreline.

Harris and Marshall (1963) examined the ecological effects of water level manipulation on emergent and submergent species including *Scirpus validus*, *Eleocharis palustris*, *Typha*, *Carex*, and *Salix* under one, two, three, four, and five years of draw down in a northern marsh in the Agassiz National Wildlife Refuge in northern Minnesota. Submergent species were eliminated during draw down on exposed areas but flourished in pools of standing water; the seeds survived extended dry periods and germinated after reflooding. During the first year of draw down, emergent species developed in exposed areas. Plant development depended on seed availability, soil type and moisture, season and duration of draw down, and amount of stranded algal debris. During the second year of draw down, more upland species and fewer emergent plants developed; development was influenced by residual vegetation, and soil type and moisture. During longer draw down periods, after a complete drying of the soil, stands of *Salix* developed. Reflooding of the marsh eliminated mudflat and annual species but submergent and emergent species re-established themselves (Harris and Marshall, 1963).

The impacts of regulating water levels on ten coastal marshes in Lake Erie and along the St. Lawrence River were assessed through field studies and visual interpretation of aerial photographs (Bayly, 1979a). The Long Point Crown Marsh and Long Point Company Marsh were among the ten coastal marshes studied. The Crown Marsh was characterized as an unrestricted bay marsh with uniform basin bathymetry. During periods of declining water levels, the vegetative fringe shifts lakeward; the marsh extends out a half a mile from the shore to a maximum depth of 1.3 m. During periods of rising water levels, the quality of the marsh slightly declines due to a temporary thinning of vegetation intolerant of water, but that usually adjusts after two years. Bayly (1979a) concluded that a Lake Erie water level of 571 feet would be ideal for the Crown Marsh, but the historic mean of 570.3 feet was fine. The Long Point Company Marsh represented an ideal marsh system having a perfect combination and distribution of emergent and aquatic vegetation. The bathymetry of the marsh is irregular and the contours slope inward towards the centre of the marsh from the marginal ridges. Persistent low water levels in the Company Marsh initiates the growth of *Typha* and *Phragmites* and allows succession to terrestrialization. Persistent high water levels leads to the progressive loss of emergent and aquatic vegetation and causes more extensive storm damage in the marsh. The study concluded that a lake level slightly higher than the mean of 570.3 feet would be optimal but the quality of the marsh at the mean level was excellent (Bayly, 1979a).

The International Lake Erie Regulation Board (ILERB) (1981) also assessed the impacts of water level regulations on wetland vegetation and wildlife in Lake Erie. The impacts of regulating the range in water level fluctuation to 30 cm above and below the mean lake level were studied. The study characterized the response of wetland vegetation to past water level fluctuations and observed that the productivity, biological composition, and size of the wetland varied with water level. The last episode of high water levels in Lake Erie caused the dieback of emergent vegetation and expansion of open water communities. As water levels receded, most vegetation communities reestablished themselves successfully, except *Typha*. The study concluded that a reduced range in water level fluctuations reduces the amount of wetland area and encourages dense growth of emergent vegetation (ILERB, 1981).

Koshida (1988) examined the impacts of water level fluctuations on wetland vegetation communities. Koshida states that fluctuations ranging from 30 cm above or below the current mean lake level would be optimal for Long Point, but anything greater will lead to a decrease in vegetation diversity within the wetland. The 30 cm reference that Koshida used was established by the study conducted by the ILERB (1981). An increase in water levels greater than 30 cm above the mean for three to five years would reduce or eliminate dense emergent vegetation. Floating and submergent communities that are more tolerant of flooding replace the emergent vegetation. A decrease in water levels greater than 30 cm below the mean would promote dense emergent growth. Longer periods of draw down, lasting more than three years, would initiate succession to drier meadow and treed communities (Koshida, 1988, 43).

Whillans (1985) examined the relationship between wetland vegetation and water level fluctuations in Inner Long Point Bay. Wetland vegetation classes were interpreted from aerial photographs dating between 1945 and 1978. Class area was correlated to water levels using Pearson Correlation coefficients. During periods of high water levels, robust emergent and wet meadow vegetation dominated and during periods of low water levels the areas of submergent and floating-leaf plants decreased (Whillans, 1985). Spatial statistics, including linear regression and correlation analysis, were successfully applied to relate wetland vegetation change to water level fluctuations. Similar techniques could also be applied to assess the relationships of the different wetland communities to water levels in the entire Long Point wetland complex; these relationships could also help derive a simplified wetland vegetation classification scheme for Long Point.

The impacts of long-term stable water levels on vegetation dynamics were examined in the Marsh Ecology Research Complex in Delta Marsh, Manitoba (Grosshans and Kenkel, 1997). The responses of emergent vegetation along the water depth gradient were observed, but the study noted that salinity and competitive interactions also influence vegetation response in the marsh. The experiment observed that *Typha* survived better in deeper water areas compared to other emergent species studied because of the plants tall thin leaves and great rhizome storage capacity, but the plant was less competitive in drier areas (Grosshans and Kenkel, 1997). Shay *et al.* (1999) also analyzed the response of emergent vegetation to fluctuating water levels in the complex. A correlation analysis was conducted to determine the relationship between marsh vegetation change and lake levels from 1948 to 1997. The study observed that *Typha* dominated during periods of declining water levels, *Phragmites* dominated during periods of low water levels, but emergent vegetation died during periods of high water levels.

The literature examining the effects of water level fluctuations on wetland vegetation focused on describing the response of a few individual wetland species or emergent and submergent communities in a broader context. The review of the literature did, however, provide some insight into how vegetation responds to variations in the magnitude and duration of water level fluctuations, and the impacts of water level regulations on vegetation response. Further analyses have attempted to model the responses of vegetation communities along the wetland continuum to water level fluctuation. These studies are discussed in the following section.

3.1.3 PLANT COMMUNITY DISPLACEMENT MODELS

Plant community displacement models characterize the response of wetland vegetation communities to water level fluctuations. The models are one-dimensional and show the elevational distribution of vegetation communities in relation to actual and inferred water levels (Geis, 1985). Development of these models has been based on ecological understanding of the biophysical conditions within a wetland system derived through field observations. Several displacement models have been developed to characterize community response within coastal wetlands of the Great Lakes. Gilman (1976) developed a model that summarized the preferred location of major plant species along a linear gradient of mean annual water levels for Campbell Marsh in New York. The model provides insight into the tolerance ranges of dominant submergent and emergent species along the wetland continuum (Painter and Keddy, 1992; McNaughton and Wolf, 1970).

A plant community displacement model was developed for Dickson Island in the St. Clair River Delta (Jaworski *et al.,* 1979). The model was developed as part of a study to document changes in the areal extent and plant community composition of seven Great Lakes shoreline wetlands in response to periodic water level fluctuations. Vegetation transects were established from open water to the upland community along the water depth gradient and maps were compiled during high, low, and average water level periods. As water levels declined to low conditions in 1964, sedge and wooded shrub increased while emergent, submersed and floating-leaved vegetation, and open water decreased; there was no change in meadow. As water levels rose to high conditions during the 1970s, the extent of submersed and floating-leaf communities increased causing a decline in emergent and sedge marsh communities, and meadows were drowned and displaced by sedges. The upper and lower extents of the plant community displacement in response to high and low water level conditions at Dickson Island were characterized (Figure 3-2).

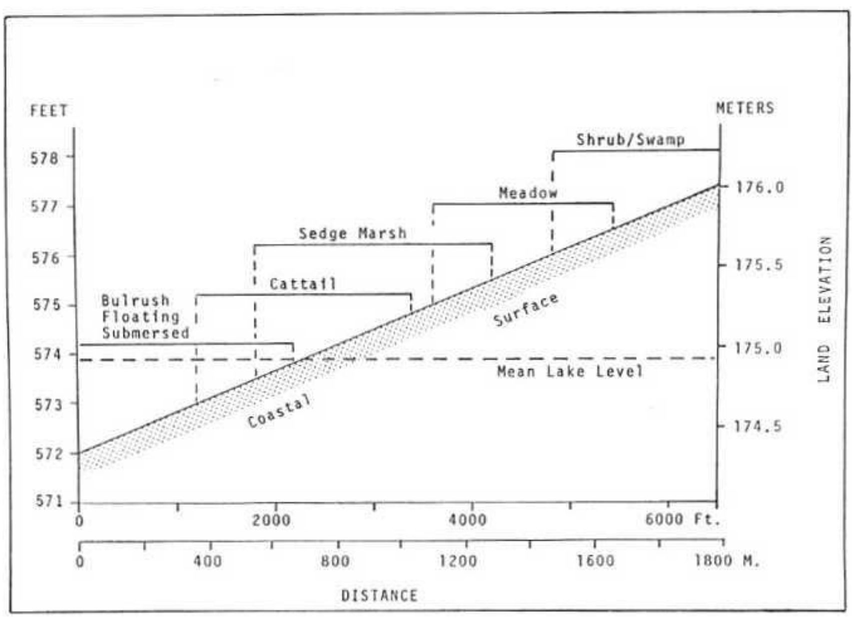


FIGURE 3-2: DICKSON ISLAND PLANT COMMUNITY DISPLACEMENT MODEL
(Herdendorf *et al*., 1981)

The plant community displacement model indicates that a narrow zone of sedge persists at any lake level between the upper limit of cattail and the lower limit of meadow, but deeper water communities may be displaced entirely. As water levels rise, the areas of sedge, emergent, and shrub forest decrease and open water increases. Sedge marsh experiences the greatest change in area, and shrub forest experiences the least amount of change. During average water level conditions, emergent vegetation is the most widespread, and as water levels decline the area of open water decreases and drier vegetation communities increase. A lag of two to three years was noted for a complete dieback of plant communities during higher water levels, and the lag time for maximum colonization of plants is longer for periods of low water levels (Herdendorf *et al.*, 1981, 297). Prior to this model, studies mainly focused on inland wetlands where water levels were managed within a confined system (Herdendorf *et al.*, 1981).

Keddy and Reznicek (1986) developed a vegetation displacement model for shoreline wetlands. The upper limit of the model consists of woody plant species that are intolerant of flooding. Wet meadows persist between the maximum and present water levels. During higher water levels, herbaceous and woody plants in the wet meadow are killed and the width of the wet meadow narrows. During periods of declining water levels, wet meadow reestablishes itself from buried seeds or by colonization of remaining species in the upper fringe of the community. Shallow marsh vegetation occurs between the present and extreme minimum water level. Emergent vegetation in the shallow marsh can survive long periods of flooding but requires drought conditions for seedlings to germinate and reflooding above the extreme low for seed recruitment. Aquatic vegetation occurs below the minimum low water level; emergent vegetation invades the aquatic zone during low water level periods, but they are eliminated when water levels rise. The model only considers the role of fluctuating water levels and could not predict the occurrence of communities or species in the wetland. The model does provide a conceptual framework for interpreting large-scale cyclic processes of vegetation (Keddy and Reznicek, 1986, 32).

Painter and Keddy (1992) modelled the response of the emergent marsh community to water level fluctuations in order to evaluate the potential impacts of regulation schemes on emergent vegetation in the Great Lakes. The model was based on general vegetation response to seasonal water level fluctuations and water levels of the previous year. High water events were used to determine the landward extent of the marsh and the "three months surrounding the peak during the growing season were chosen as the critical time period which would influence the location of the woody plant/marsh transition" (Painter and Keddy, 1992, 7). Low water events marked the lakeward extent of the marsh. The critical time period that influenced the location of the transition boundary between emergent and submergent vegetation was the mean water level for September. The model was then applied to analyze the impacts of various Great Lakes regulation schemes on marsh wetlands.

Although research characterizing the tolerances and linear displacement of wetland vegetation communities was limited, the review of the plant community displacement models has provided insight into the upper and lower water depth tolerances of wetland vegetation communities. The tolerance ranges of wetland vegetation communities identified in the literature could be applied to characterize the response of vegetation communities in the Long Point wetland complex.

3.1.4 SUMMARY

Burkett and Kusler (2000, 315) state that "predicting wetland response to climate change and variability is . . . limited by our understanding of how wetland flora and fauna respond to changes in temperature, precipitation, water level, water quality, and atmospheric carbon levels." Literature assessing the implications of climate change in wetland systems was limited, but review of the existing literature provides a general understanding how wetland vegetation respond to water level fluctuations and how the vegetation may respond to Lake Erie water level declines projected from human-induced climate change. Traditionally, wetland vegetation community response to water level fluctuations has been assessed through intensive fieldwork and biological inventories in a qualitative manner. With the advent of computer-based technology, such as geographic information systems, these responses can be characterized quantitatively. The role of geographic information systems in wetland analysis is discussed next.

3.2 GEOGRAPHIC INFORMATION SYSTEMS

A Geographic Information System (GIS) is a computer-based system that allows for the input, storage, management, analysis, and output of spatial data (Aronoff, 1995). Spatial data can be collected from a variety of different sources including topographic and land use maps, aerial photography, satellite imagery, and global positioning systems (GPS). Aerial photographs are the largest source of historical data available for studying changes within wetland environments. Aerial photographs offer long-term coverage (some photographs date back to as early as the 1920s), high spatial resolution, and large spatial extent compared to other data sources (Kadmon and Harari-Kremer, 1999). Within a GIS, data can be easily updated, linked with attribute data, used to generate new spatial datasets, or used as input for computer-based simulation models. In addition, a GIS can generate a number of reports and high quality graphical output to aid in planning and management strategies.

3.3 APPLICATIONS OF GIS TO WETLAND MANAGEMENT

The significant rate of wetland loss and resulting impacts on the surrounding environment, combined with a growing realization of the important values and functions that wetlands perform, has prompted an increase in wetland studies. Many of the studies have focused on the proper management, protection, and restoration of these highly productive and diverse ecosystems. Although the use of GIS in wetland studies is relatively new, they are becoming recognized as a valuable analytical tool for wetland management and analysis. Recent advancements in computer technology has facilitated the use of GIS as an efficient and effective management tool for monitoring and modelling wetlands that is fast, inexpensive, accurate, and user-friendly. The greatest potential of GIS in wetland analyses, however, is its powerful analytical capabilities (Woodcock *et al.*, 1990). According to Johnson (1990, 31), GIS can be employed to (1) analyze temporal change, (2) determine spatial relationships between physical and biological features, (3) determine spatial characteristics, (4) analyze the direction and magnitude of changes, and (5) interface with simulation models for predictive analyses. The applications of GIS to the field of wetland studies, from simple wetland inventories through to complex modelling of wetland processes, are reviewed.

3.3.1 WETLAND INVENTORY

GIS are useful in the development of wetland inventories for effective natural resource management practices. The literature, however, identified several problems associated with creating spatiotemporal databases. Often there is a lack of suitable data for wetland inventories. Limited access to certain wetland sites, financial and technical constraints, and lack of human resources can be problematic in creating wetland inventories (Haack, 1996). In addition, the amount of time required to create such inventories can be tremendous as interpretation of the aerial photographs and digitizing the information into a GIS is quite tedious and labour intensive. Furthermore, there are problems related to the positional accuracy, attribute accuracy, completeness, and consistency of the data (Langran, 1992). Errors can originate from inaccurate information of landscape elements on historical maps, or through the interpretation, digitization, or processing of the data (Johnson, 1990). But once these problems have been addressed, there are many useful applications of a GIS in wetland studies.

Spatial and attribute data stored in wetland inventories can be quantified to provide information relating to wetland cover types. Basic areal and linear measurements of land covers are easily derived in a GIS from the spatial data and when combined with attribute data, the number, type, and diversity of cover types can also be quantified. Several studies utilizing basic statistics to document change in wetland environments are discussed below.

The area and percentage of wetland vegetation in the Winous Point Marshes in Lake Erie were quantified from aerial photographs to determine the effects of high water levels on the wetland from 1973 to 1974. The authors noted dramatic vegetation change as a result of high water levels compared to draw down conditions in 1960 and 1965. Open water increased significantly and there were notable decreases in the amount of emergent vegetation in the marsh; the decline in muskrat population was attributed to the decrease of emergent vegetation (Farney and Bookhout, 1982). A spatial inventory of the Metzger Marsh in Lake Erie guided restoration efforts of the coastal marsh. Historical changes in wetland area from 1950 to 1994 were documented in relation to water level changes, littoral drift, and the condition of a protective barrier beach (Kowalski and Wilcox, 1999).

LANDSAT images were digitized into a GIS to study wetland changes in East Africa due to lake level fluctuations. The size of the wetland was measured and changes were descriptively related to lake levels and sedimentation (Haack, 1996). The study effectively documented change within the wetland between the dates of imagery, but could not conclude whether the changes were a result of climate change or destructive land use patterns. The aforementioned studies successfully applied the basic tools of a GIS to document change in wetland systems. But the studies neglected to utilize the more powerful analytical capabilities of a GIS to quantify changes in wetland structure and pattern over time or to empirically relate wetland changes to environmental conditions, physical constraints, or human influences.

3.3.2 STATISTICAL ANALYSIS

For proper wetland management, it is important to understand the impacts of environmental variables or natural and human-induced changes, such as climate change and water level fluctuations, on wetland vegetation and structure. A spatial inventory of wetland data can be integrated with spatial statistical analysis programs to relate environmental variables to wetland distribution or change. For example, climatic and physiographic conditions affecting wetland type and distribution in Manitoba were determined through statistical analysis of a spatial wetland inventory. A hybrid detrended canonical correspondence analysis was performed on the dataset; the analysis identified that wetland distribution was highly correlated with mean annual temperatures. Thermal seasonal aridity, annual precipitation, moisture deficit, bedrock geology, surface water flow deficit, texture, and hydraulic conductivity were also identified as important variables affecting wetland distribution in the province (Halsey *et al.*, 1997). Owen (1999) investigated the impacts of changing land use patterns on an urban wetland in Wisconsin. Hydrology, land use, and vegetation patterns were digitized from aerial photographs and historic maps. Vegetation species abundance was correlated to hydrological, chemical, and spatial variables using linear regression and multivariate ordination. The analysis concluded site elevation and water levels were the most important factors influencing the distribution of wetland vegetation.

There are several other studies that implemented statistical analysis to relate water level fluctuations to wetland change. The historical area and distribution of wetlands at Pointe Mouillée in Lake Erie were interpreted from aerial photographs. Historical changes in the wetlands between 1935 and 1980 were related to lake levels using linear regression (Lyon and Greene, 1992). A similar study was conducted for wetlands in the Straits of Mackinac (Lyon *et al.*, 1986).

Although these statistical analyses do not include a temporal assessment of change over time, the analyses do illustrate the usefulness of integrating a GIS with statistical software programs to assess the relationship of wetland vegetation with environmental variables and human influences (Stow, 1993). Subsequent to determining the factors that influence wetland change, it is important to quantify the structure or pattern of changes in wetland systems. Hence, an in-depth examination of the temporal and spatial trend analysis capabilities of a GIS follows. Examples in the literature that apply statistical analysis to relate wetland change, in the context of temporal and spatial trends, to environmental processes are also provided.

3.3.3 TEMPORAL ANALYSES

One of the important aspects in wetland management is studying the long-term changes in landscape structure. An understanding of the past is necessary for predicting future trends and for determining effective management strategies (Pan *et al.*, 1999). It is important to study spatial patterns in a landscape because these patterns influence ecological phenomenon such as flows of sediments and nutrients, spread of disturbances, and net primary productivity (Turner, 1990). As noted earlier, GIS are effective and efficient tools for developing wetland inventories and conducting basic inventory analyses of wetland data, but many GIS have limited functionality for conducting temporal analysis (Kienast, 1993). Spatial pattern analysis programs can be used in conjunction with GIS to quantify changes in wetland structure over time (Stow, 1993). Spatial indices derived from these programs can describe the size and shape of patches in a wetland system, the arrangement of patches within the landscape, and the rate and direction of change (Turner, 1987).

Numerous studies have successfully applied spatial indices to assess temporal change across various landscapes. Literature specifically examining temporal changes in wetland systems is limited, therefore additional examples of research conducted in a broader context of landscape ecology and resource management are provided. Landscape patterns and dynamic changes between 1972 and 1995 were characterized in rural China using a variety of structural, patch, and shape indices (Zaizhi, 2000). Structural indices including number of patches, total area, proportion of total area, and the mean, maximum, and minimum values of eleven land use classes were calculated. Patch indices included a landscape diversity index, dominance, and fragmentation index. Fractal dimension was the only shape index calculated. Rural landscape changes in Georgia were also quantified with indices that measured the number, area, amount of edge, and fractal dimension of each land use in the landscape (Johnson, 1990).

Temporal changes in land cover as a result of English settlement in the Herbert River Catchment in Queensland, New Zealand were characterized from 1860 to 1996 (Johnson *et al.*, 2000). The analysis quantified changes in the shape, size, and pattern of vegetation using edge to area ratios, dissection indices, island distribution index, evenness and diversity indices, and density of islands. Changes in the landscape pattern of a watershed in central Honduras were quantified from 1955 to 1995 with several structural indices along with a shape complexity index. Physical, ecological, and socio-economic factors influencing land use changes in the region were examined qualitatively (Kammerbauer and Ardon, 1999). Landscape indices have also been applied to measure forest fragmentation (Wu *et al.*, 2000; Ripple *et al.*, 1991), patterns of diversity within a forest resulting from human settlement and fire (Romme, 1982), and land use changes in watersheds and plains that result from agricultural and human development (Scott and Udouj, 1999; Simpson *et al.*, 1994).

There were few studies found in the literature that applied landscape indices to quantify changes in wetland systems. Landscape indices measuring diversity, edge, and interspersion were computed for six wetlands in the Great Lakes to assess changes in wetland vegetation over time. Area statistics were summarized to indicate directions of change, and then a variance analysis and detrended correspondence analysis were performed on the data to determine the relationship of wetland vegetation abundance to water level fluctuations. Fourteen wetland vegetation classes were included in the analysis, which is significant because many studies generally use from five to eight classes in their analyses. The study noted two limitations with their approach however. No wetland data were captured below the historical mean water level, hence the short term impacts of lower water levels could not be determined, and there were not enough years of data available to distinguish between the short and long term effects of water level fluctuations (Ecological Services for Planning Limited, 1992).

Long-term changes in wetland habitat are quantified for the Winous Point Marshes from 1873 to 1991 (Gottgens *et al.,* 1998). A significant amount of research focused on restoration efforts and the ecological consequences of restoration effects in the marsh, but there was little research quantifying how the marsh had changed over time in relation to environmental and human influences. Aerial photographs and wetland maps were digitized and statistics related to class area, patchiness, and edge habitat were calculated. A regression analysis was then conducted to determine the correlation of vegetation types with water levels. Changes in the marshes are well documented and correlated with the historical development at the marsh (Gottgens *et al.,* 1998).

Quinlan (1985) analyzed the impacts of various water levels on wetland vegetation community structure in three marshes along the Toronto waterfront. Aerial photographs were interpreted and 13 wetland community classes were mapped; a total of ten years were mapped from 1927 to 1983. An historical analysis determined the percent change in marsh areal extent, nature, and magnitude of change, edge diversity, and amount of interspersion between plants and water levels. During periods of extended low levels, sedge and grass dominated, open water was restricted, and there was poor interspersion between the open water and plant communities. During periods of higher water levels, the area of open water increased and there was greater interspersion between the communities. A predictive plant community model was then developed by plotting the percentage of marsh area for each cover type by the lake level. This model could be applied to model similar types of marsh, i.e. stable barrier beach marshes that are influenced by both lake and river water to determine the direction and relative magnitude of change. Although only lake levels were considered in the analysis, Quinlan (1985, 119) notes that vegetation response depends on the differences in species composition, surrounding land use, and basin bathymetry.

Another study evaluated three different approaches to assess the impacts of water level fluctuations on wetland vegetation in the Delta Marsh, Manitoba (van der Valk *et al.*, 1994). The first method employed a GIS to quantify several structural and vegetation indicators for the wetland. The second method involved quantifying species richness, total shoot density, Shannon’s index, and Simpson’s index of diversity in permanent quadrants in each marsh cell. Finally, Euclidean distance and Bray-Curtis similarity index were calculated for emergent vegetation in each permanent quadrant. The similarity test was unsuccessful in detecting changes in the wetland quadrants due to water level changes. The method of using vegetation data measured from field studies proved best for detecting the impacts of high water levels on the vegetation. The digitized vegetation maps were less successful in detecting change, but the authors concluded that the integration of vegetation data and maps would be the more informative and reliable approach to study impacts (van der Valk *et al.,* 1994).

Examination of the literature related to temporal trend analysis identified that a variety of different metrics can be used to quantify changes in the structure and pattern of natural and human landscapes over time. The studies reviewed have effectively demonstrated the application of spatial pattern statistics to measure change in the diversity, quality, and integrity of various environments, but there has been a limited application of these metrics to quantify change in wetland environments. The majority of studies pertaining to wetland environments applied statistical analysis techniques to relate wetland vegetation change to water level fluctuations so that changes in the structure of the wetland could be explained. Exploring changes in spatial pattern and structure temporally is one component of characterizing historical changes within a landscape. Historical changes can also be characterized through a spatial analysis, which quantifies the type and location of change within the landscape.

3.3.4 SPATIAL ANALYSES

The overlay capabilities within a GIS are powerful tools for spatially documenting landscape change over time. Spatial overlays are simple to perform and provide an excellent indication of overall land use structure and change (Pan *et al.,* 1999). Spatial overlays have been applied in numerous studies to monitor change over time, particularly within the built environment. Land use change studies may not be directly related to wetlands but such studies do illustrate the potential of spatial overlays for wetland analysis. For example, Adeniyi (1980) encoded land use data interpreted from aerial photography into raster grids and then applied spatial overlays in a GIS to determine the magnitude, type, and location of change resulting from urban growth in Lagos, Nigeria. In the analysis, Adeniyi (1980) quantitatively documented (1) the magnitude and rate of change between land use classes by comparing area statistics over time and (2) the type of change by constructing a land use change matrix of inter-categorical changes. The spatial locations of from and to changes were also illustrated with a land use change detection map. The comprehensive change analysis performed for Lagos could also be applied to assess change in other environments as well, including the wetland complex at Long Point. In fact, this paper provided the framework for documenting wetland changes within Long Point.

There are numerous examples in the literature that also apply spatial analysis to document change in wetland environments. Many studies have integrated remotely sensed imagery into spatial change detection analysis. Satellite imagery were overlaid in a GIS to quantify changes within wetland systems in Zambia from 1984 to 1994 (Munyati, 2000) and the Tamil Nadu and Andaman and Nicobar group of islands in India from 1989 1996 (Ramachandran *et al.,* 1998). Spatial overlays were applied to LANDSAT images to identify changes in aquatic macrophytes in the Everglades. Aquatic macrophytes, or broad-leaf emergent plant species, were classified in the images and then a post classification change detection was applied to determine changes in the spatial distribution of the vegetation from 1973 to 1991 (Jensen *et al.*, 1995). Satellite images of wetlands across Australia were overlaid to determine the spatial and temporal distribution of wetlands from 1986 to 1997. Climatic factors that influenced wetland area were also examined. A linear regression analysis of the change in wetland area was performed and the results were compared with rainfall data to determine if there were any relationships between the two variables. A significant relationship between wetland area and rainfall existed (Roshier *et al.*, 2001).

Yeung (1993) determined land use and wetland change at Long Point utilizing LANDSAT Multi-Spectral Scanner (MSS) images from 1974 to 1984. A supervised classification was conducted on the images to classify land use as agriculture, deciduous woodland, coniferous woodland, marsh, built up areas, and open water. The images were overlaid to determine areas of change. There was a noticeable declining trend in marsh area during the period of study but the resolution of the images was too coarse and the classification too broad, however, to show the complexity of change within the marsh wetland itself.

There are several documented studies examining wetland changes in Louisiana. The total area and percent change of wetland classes were calculated and change maps were produced through overlays of wetland data in the Baptiste Collette Bayou Sub-delta of the Mississippi River delta (Ader and Johnston, 1982) and Barataria Basin (Evers *et al.*, 1992). In addition, Ramsey and Laine (1997) compared the use of remotely sensed data versus high-resolution photography in identifying coastal wetland change in Louisiana prior to and post Hurricane Andrew. The study concluded that classification errors and boundary pixels, associated with satellite imagery, limits the applicability of the LANDSAT Thematic Mapper (TM) image to detect wetland change. The resolution of the image was also problematic for detecting change; the resolution of the TM image was 25 m by 25 m, compared to 1 m by 1 m resolution of the aerial photography.

Spatial overlay analysis was implemented to characterize wetland vegetation succession in floating wetlands in the Netherlands. Aerial photographs between 1937 and 1989 were interpreted for vegetation type, digitized, and converted into raster images. The area of each vegetation type was calculated, and transitions between vegetation types were quantified with overlays. The turnover times of areas changing into another vegetation type were also calculated to determine rates of succession (Bakker *et al.*, 1994). Major changes in four vegetation types were determined and major patterns of vegetation succession in the wetlands were successfully described.

Land use and land cover changes from 1955 to 1990 were examined within the Long Point Biosphere Reserve (Lawrence and Beazley, 1994). Land covers were interpreted from aerial photographs taken in

1955, 1978, and 1990 and mapped into a GIS. The data layers were overlaid to determine the amount and location of change in the Reserve. The analysis observed that (1) the amount of open water decreased to less than one percent of the total area because of infilling and (2) the amount of emergent vegetation has increased due to a 0.2 m decline in lake levels between 1955 and 1988. The authors note that dredging, land development, expansion of the marina, and shoreline protection has influenced change in the wetland throughout the period of study.

Spatial change analyses were conducted using overlay operations to summarize changes in marsh and upland vegetation on the Sapelo Island off the Georgia coast and Lake Marion in South Carolina. Distribution maps of aquatic macrophytes were derived from aerial photography. Water quality conditions were also mapped and the data were converted into raster grids. The grids were overlaid and areas of change for emergent, submergent, free-floating, and mixed vegetation were quantified (Welch *et al.*, 1992). The vegetation grids were also overlaid with bathymetry data to derive water depth ranges of the vegetation types, and plotted against water quality parameters to determine if there were any coincidences with the pattern of plant growth in the marshes. The analyses found that phosphorus greatly influenced the growth of submergent vegetation in the marsh (Welch *et al.*, 1988).

The long-term effects of water level fluctuations were determined by analyzing aerial photographs for wetlands along the St. Marys River in Michigan between 1939 and 1985. A transition matrix was derived from cross-tabulations in a GIS to quantify the amount and direction of change between vegetation classes. Standard linear regression analysis of vegetation class area and moving averages of water levels were used to determine the effects of lag times. Several observations were drawn from the analysis. Emergent vegetation had a five-year response rate to water level fluctuations. The amount of emergent vegetation increased when water levels declined and decreased as water levels rose. Scrub-shrub decreased as water levels rose and increased as water levels declined. Forest increased during periods of rising water levels; this relationship was unexpected since trees are less tolerant of flooding than herbaceous plants, but the authors did note that wetland trees are more tolerant of flooding than upland vegetation (Williams and Lyon, 1997; Williams and Lyon, 1991).

The application of spatial analysis to wetland data has proved successful in detecting changes in wetland vegetation area. The literature has identified that spatial overlays are simple to perform and are effective for documenting the amount, type, and location of vegetation changes. The literature also demonstrated that spatial analyses could be performed with satellite imagery or raster grids created from wetland data interpreted from aerial photography. Few of the studies have applied statistical analysis to relate wetland change to environmental conditions.

3.3.5 INTEGRATION OF TEMPORAL AND SPATIAL ANALYSIS

Temporal and spatial trend analyses quantify different phenomena within a landscape. Temporal analyses quantify change in the spatial structure and pattern of the landscape over time, whereas spatial analyses quantify the amount, type, and direction of change within the landscape. Nevertheless, the integration of a temporal trend analysis with a spatial overlay analysis would provide a unique examination of landscape change over time. Literature integrating the two types of analyses, however, was scarce. One study examined landscape changes in a rural environment.

Temporal and spatial landscape patterns in rural Haut-Saint-Laurent, Quebec were quantified from 1958 to 1993 at the field, patch, and landscape levels (Pan *et al.*, 1999). Aerial photographs were interpreted to determine land use classes and then digitized into a GIS. Structural indices measuring patch area, patch number, average patch size, and edge length were calculated along with landscape dominance, diversity, contagion, and fractal dimension. Rates of land use change were also derived from transformation matrices created by overlaying successive years of data. Furthermore, canonical correspondence analysis was utilized to assess the relationship of these changes to physical attributes. Pan *et al.* (1999) notes that many existing studies use “similar method[s] to quantify structural patterns . . . by comparing various landscape structural indices through time”, but many studies have failed to relate physical constraints or environmental conditions to these processes. The strength of Pan *et al.*’s analysis is that a temporal and spatial analysis is combined to describe the change and relate those changes to other environmental factors.

3.3.6 SUMMARY

A review of the literature identified that there are a variety of methods for documenting change in wetland environments. Changes within wetlands can be quantified temporally by examining changes in the structure and pattern of the system or can be quantified spatially in terms of documenting the location and type of wetland vegetation change. An integrated analysis that quantitatively explores wetland change is optimal, but most analyses have failed to integrate the temporal and spatial components. Statistical analyses can also be integrated into the change analysis to determine the underlying factors affecting wetland change. The application of statistical analyses to this investigation is limited however by the availability of spatial data for Long Point; only historical wetland vegetation and land use data along with topographic data are available. The literature review stresses the importance of understanding historical changes in a landscape to predict future landscape dynamics and devise more effective resource and landscape management strategies (Pan *et al.*, 1999). Modelling approaches that can predict wetland dynamics are reviewed in the next section.

3.4 WETLAND MODELLING

Mitsch *et al.* (1982) provides a review of the different types of freshwater wetland models that have been developed in the United States and Canada (Mitsch *et al.*, 1982). There is, however, an absence of spatially integrated models, as many of the existing wetland models do not include "spatial components in the selection of variables, nor do they have spatially distributed determinations of model results" (Lyon and McCarthy, 1995a, 5). The more advanced capabilities of GIS allow wetland models to be developed in a spatial context. The main advantage of a GIS is that it deals with large volumes of diverse and spatially orientated data that geographically anchor processes occurring across space and time (Payn *et al.*, 1999, 189). A review of the literature related to spatial wetland modelling is provided below. Wetland models were summarized according to the type of modelling approach used.

3.4.1 REGRESSION MODELLING

Simple regression models were developed to study the impacts of short-term water level fluctuations on wetlands in the Great Lakes (Bukata *et al.*, 1988b). Models based on the geometric shape of the wetland were developed that related the changes in the areal extent of wetlands to water level fluctuations. The models considered linear wetland shapes, convex or concave shapes, and elliptical geometric shoreline shapes. The models were based solely on terrain characteristics, and many other factors affecting wetland change were not considered.

Multiple regression analysis was used to predict vegetation response to water level changes of shoreline communities in a hydroelectric reservoir in Northern Sweden (Nilsson and Keddy, 1988). The initial state of the vegetation in the preceding year was the most important variable for predicting the magnitude and direction of change. Duration of flooding was identified as the most important water level variable affecting change and correlated well with greater abundance and richness in the wetland; timing was the least important. Periods of longer, and less frequent, floods killed emergent vegetation thus reducing abundance and diversity in the wetland, while shorter floods increased productivity and number of species in the wetland. Duration of flooding was the only variable used to predict vegetation response. The model was successful at predicting 41 % of the change, and therefore additional abiotic and biotic factors need to be incorporated in future versions of the model to improve the accuracy and sampling errors need to be considered in the evaluation process (Nilsson and Keddy, 1988).

Linear regression analysis has been successfully applied to model the impacts of water level fluctuations on wetland and beach areas along the shores of the Great Lakes. The effects of short and long-term water fluctuations were determined for the Straits of Mackinac in Lake Michigan (Lyon *et al.*, 1986). Short-term effects were determined by measuring the amount of flooding, relative soil chemistry, and presence of plants. Long-term effects were assessed through interpretation of aerial photography and regression modelling. A wetland vegetation community classification was derived through field surveys and statistical analysis that included tests of similarity, pair-wise analysis, and Kellman tau-b rank correlations. Pearson coefficients were calculated to relate water levels to wetland vegetation extent then applied to a regression analysis to predict wetland area based on given water levels. The areas predicted by the model were similar to the wetland areas measured from aerial photographs. A similar technique was used to develop a model that related wetland presence to water levels at Pointe Mouillée Michigan (Lyon and Drobney, 1984). Pearson coefficients and regression analysis was used to determine the relationship between water levels and wetland and beach area in the wetland. Changes in wetland and beach area were predicted for given water levels with a regression model. Lyon and Adkins (1995) developed a model for predicting changes in wetland communities at the St. Clair Flats in Michigan. The regression model considered soil type, bathymetry, and previous wetland vegetation. The study concluded that elevation was important for distinguishing between wetland and upland communities, and that water levels affected species composition within the wetland (Lyon and Adkins, 1995).

Several ecological models used logistic regression to predict the presence and absence of wetland vegetation or wildlife species. Variables related to the probability of species occurring are identified; correlation coefficients for these variables are derived and then used to create an algorithm to predict whether a species occurs or does not occur. The models output a probability distribution map that shows the likelihood of a particular species occurring across the study area. This technique was used to determine the future distribution of aquatic macrophytes in relation to four biophysical conditions in the Par Pond Reservoir in South Carolina (Narumalani *et al.*, 1997); predict the distribution of a marsh-nesting bird in coastal wetlands of Lake Erie (Özesmi and Mitsch, 1997); and, predict the distribution of wetland plant species in the Netherlands (van Horssen *et al.*, 1999). The application of logistic regression analysis in ecological modelling is, however, mainly used to model the distribution of a single entity, whether that entity is a particular plant or wildlife species or a wetland vegetation community. Each entity requires its own unique algorithm to predict the probability of that entity occurring across the landscape and the model generates separate probability distribution maps for each entity. Therefore, logistic regression modelling would not be optimal to predict the occurrence of all the wetland communities at Long Point since there are too many wetland vegetation classes to map. But the method could be used to predict the occurrence of one or two wetland vegetation communities that are of particular interest. Van de Rijt *et al.*, (1996) implemented logistic regression analysis to predict the distribution of vegetation types occurring in a former tidal area in the Netherlands. Rather than predicting the occurrence of individual species, vegetation types were used instead; this reduced the amount of output generated from the model and also provided a better indication of the environmental factors affecting wetland. However, the study did not attempt to validate the predicted output with actual data so it is unclear how well the model performed; validation was identified as a focus for future research.

3.4.2 CARTOGRAPHIC MODELLING

A cartographic approach was used to model the spatial distribution of aquatic macrophytes in Lake Marion, South Carolina (Remillard and Welch, 1993). Environmental variables were overlaid with macrophyte vegetation distribution maps in ARC/INFO. A Chi-square analysis was performed to statistically relate the environmental variables with the distribution of macrophyte vegetation then the strength of the relationships were determined with Pearson's Correlation and Cramer's V coefficients. The distribution of macrophytes was predicted for a new area of the lake using the Chi-square results. Grid cells that met the Chi-square deviation criteria were reselected out into a new grid; these cells represented cells where aquatic vegetation was expected to grow. The predicted distribution grids were overlaid with the actual vegetation maps and compared. The actual and predicted areas of emergent and submergent vegetation were summarized and the percentage of actual vegetation correctly predicted was calculated. A validated model that considered only water depth and amount of sedimentation was approximately 90 % successful in predicting aquatic macrophyte growth in the lake. To apply this modelling approach to other wetland systems, additional spatial data representing environmental conditions is required. But the analysis does provide a simple method for assessing the accuracy of a predictive model.

Cartographic modelling integrating remotely sensed data with geographic data was used to predict wetland loss due to projected sea level rise at Fort Moultrie, South Carolina (Jensen *et al.*, 1993). A digital elevation model was analyzed in a GIS to determine the areas that were flooded or not flooded based on various sea level scenarios. Polygons representing flood and no flood conditions were used as masks to document wetland loss on classified satellite images. The land covers affected by sea level rise were identified. Although vegetation patterns are not predicted, the analysis provides a simple method of determining and quantifying areas of wetland loss or gain due to changes in water levels through the use of a digital elevation model (DEM).

3.4.3 PROCESS-BASED MODELLING

Process-based models describe flows and processes across space. A process-based model compartmentalizes the spatial landscape into some geometric design (usually as grid cells) and then describes the flows within the compartments and the spatial processes between the compartments according to location-specific algorithms (Sklar and Costanza, 1991, 265). Process-based modelling has been employed to predict the response of coastal wetlands to sea level rise on the northeast coast of Florida (Lee *et al.*, 1992) and in Puget Sound, Washington (Park *et al.*, 1993). These models consider the effects of flooding depending on the water depth of a cell, and the spatial relationships of the cell with regards to fetch and erosion processes and water table response. In addition, a grid-based model coupling hydrodynamic and ecological components was developed to simulate 30 years of habitat change related to the impacts of proposed projects in the Mississippi Delta (Martin *et al.*, 2000).

Ellison and Bedford (1995) developed a spatial simulation model to simulate the responses of plant communities to disturbances in a Wisconsin wetland. A cell in the model was updated according to the cells current state, the state of its neighbours, and a set of transition rules. Within the model, a set of probability transition rules for the growth and death of the vegetation were established. The rules considered the water depth of a cell, season, seed bank germination, and seed dispersal. A sensitivity analysis performed on the model indicated that seed dispersal and germination had the greatest impact on the relative abundance of species in the model (Ellison and Bedford, 1995)

Spatial and temporal succession in a marsh complex in South Louisiana was modelled using dynamic spatial modelling techniques (Costanza *et al.*, 1990; Costanza *et al.*, 1986; Sklar *et al.*, 1985). The model was developed to assess the impacts of proposed projects on the river delta, to quantify the spatial distribution of change, to identify changes in habitat type and health, and to model alternative management options. The model simulates wetland succession with forcing functions and dynamic processes related to water level and flow, subsidence, river and tidal inputs, salinity, sedimentation, and plant production. Differential equations were developed using the Euler numerical integration technique to determine changes in the conditions of a cell due to water exchanges across the boundaries of the cell. The model did not consider any barriers to flows in the system, except for the outer boundary of the study area, which is unrealistic in real systems. Succession occurs when one habitat becomes more productive than another because of changing conditions in that cell. Goodness of fit was originally used to validate the models; however, the authors noted that this measurement is only good for ordinal or interval variables, not categorical variables (Sklar *et al.*, 1985). Therefore, the percentage of cells that were correctly classified was used as their 'goodness of fit' measurement. The results of the model were highly aggregated and generalized, and therefore could not be applied to quantify any real world systems nor could the parameters in the model be optimized to suit different management options (Sklar *et al.*, 1985).

A similar spatial simulation model was applied to model historical changes in another wetland complex in Louisiana; the Coastal Ecological Landscape Spatial Simulation (CELSS) Model effectively simulated ecological and physical processes as well as long-term changes to the system (Costanza *et al.*, 1988). Finally, another study investigated the applicability of explicit and implicit spatial models to model wetland change in South Carolina (Sklar *et al.*, 1994). An implicit model was used to assess the impacts of land use changes on plant productivity and nutrient cycling in a tidal freshwater marsh and an explicit model was developed to determine the impacts of sea level changes on hydrology and salinity in tidal saltwater marshes. Spatially implicit landscape models simulate the response of one component, or system, in the landscape to changes in other components within the landscape, whereas in spatially explicit models, "changes in landscape characteristics are calculated separately and enter into the landscape simulation as process-modifying time-series inputs" (Sklar *et al*., 1994). The investigation concluded that the implicit model was simpler to design and required less computer-processing time, but the model could not simulate spatial processes well. The explicit model was more complex but also more realistic since there were feedbacks in the system. Explicit models were deemed better at predicting wetland response and therefore would be more beneficial in assessing alternate management options.

The application of process-based modelling to simulate wetland vegetation response at Long Point would be difficult. Although process-based models simulate physical and biological processes and flow across landscapes, provide feedback, and link causes with effects, the models are based on complex non-linear equations and require extensive computer processing times. Hence, the spatial and temporal scale of the data is often very coarse (Sklar and Costanza, 1991). Furthermore, developing a process-based model for Long Point would require a greater technical understanding of the natural processes and flows in the system and additional datasets for input into the model; datasets that are not currently available for Long Point.

3.4.4 PROBABILITY MODELLING

The use of spatial probability modelling in wetland studies is rare. Brossard and Joly (1994) developed a probability model to predict plant distributions in Svalbard, Norway. The relationships between eight landscape types and the distribution of plant types observed in those landscape types were established through an analysis of correspondence that used Chi-square metrics and then empirical probabilities were derived based on frequency values. The probabilities represented the likelihood of a given plant type existing in a cell of a known landscape type; the probabilities of occurrence were mapped for certain plant species in the study area. Since this technique maps the probability of a species being present or absent in the landscape, it would not be effective for modelling wetland vegetation changes in Long Point. No other studies that employed general probabilities of occurrence to wetland modelling were discovered.

Transition probability models have been frequently utilized to predict changes in vegetation and land use patterns (Sklar and Costanza, 1991). In the literature, several challenges in simulating change were identified. First, transition probabilities are not spatially explicit. Landscapes are not strictly Markovian, i.e. changes in a cell are influenced not only by the initial state of that cell but are also influenced by the state and transitions of neighbouring cells (Turner, 1987). Second, transition rates are not constant through time (Sklar and Costanza, 1991; Turner, 1987). Transition probabilities are derived using two non-contiguous years of data and assume that the transition rates between those two years are constant. In fact, there may be other factors such as climate or human causes that influence the rate of change, and the influence of these factors may change yearly. It would be more realistic to use multi-year sequences of data to estimate transition frequencies; therefore climate dependent transitions can also be developed (Sklar and Costanza, 1991, 265).

Turner (1987) developed three transition probability models that simulated landscape changes in Georgia. The first model was a random simulation model based on empirically derived transition probabilities. The transitions for the second model considered the influence of the four nearest-neighbours and the third model considered the influence of the eight nearest-neighbours. Turner derived transition probabilities for two time periods: 1942 to 1955, and 1955 to 1980. The same years of data used to derive the probabilities were also used to test the model, therefore leading to some questions regarding the validity of the results.

Nevertheless, the simulated landscapes were compared to the actual landscape with spatial pattern statistics. The mean number of patches, size of patches, fractal dimension, and amount of edge between land uses were used to assess the performance of the models. The random simulation produced a highly fragmented landscape that was quite different from the actual landscape. The models that considered the contagion effects of neighbouring cells simulated clustering well, but the spatial arrangements of the patches were not as complex as in the actual landscape. It would be optimal to perform a similar validation technique for the accessing the performance of the wetland vegetation models for Long Point.

Hobbs (1994) proposed that the information from the state and transition model could be combined with simple rule-based modelling to add a temporal and spatial dimension to the transition probabilities. The integration of the standard Markov formation of the transition probability with rules allow (1) the combination of event driven and gradual transitions such as successional changes so that transitions vary over time, (2) the inclusion of stable and unstable elements in the landscape, and (3) neighbourhood effects. The theory is quite complex, but Hobbs believes that the model could be easily derived in a GIS. A model to test the theory was not developed.

3.4.5 RULE-BASED MODELLING

Literature applying the rule-based modelling approach to predict wetland vegetation change in a spatial context is limited. Rule-based models were developed to assess the impacts of abrupt climate change on marshland and foreland in the Weser estuary region in Germany (Osterkamp *et al.*, 2001) and to project the impacts of wetland restoration projects on the quality and quantity of wetland vegetation in the Gulf of Mexico (Ji and Mitchell, 1995). In addition, Poiani and Johnson (1993a; 1993b) developed a rule-based simulation model to simulate the effects of climate change on semi-permanent prairie pothole wetlands. Their model integrates a hydrological subcomponent with a vegetation subcomponent to produce monthly wetland cover and distribution maps for each growing season. The hydrological subcomponent estimates water levels based on three climatic variables: temperature, precipitation, and evaporation. The vegetation subcomponent calculates the amount and distribution of wetland vegetation using a series of if-then statements that consider the existing vegetation type, water depth, duration period at water depth, and the location of the cell within the model. Four permanent communities (upland, meadow and shallow marsh emergent, deep marsh emergent, open water) and three temporary communities (seedlings, mixed young plants, mixed emergent) were included in the vegetation subcomponent.

Although Poiani and Johnson's model was successful in simulating the change in the percentage area of emergent and open water communities, there were several weaknesses of the model. First, the model used two subcomponents. Water level fluctuations were estimated from the hydrological subcomponent and then used to predict the distribution of vegetation communities. Therefore, errors from the hydrological subcomponent may have led to additional errors or compounded the errors in the vegetation subcomponent.

Second, the wetland processes were oversimplified therefore the timing of certain vegetation changes and seedling germination was erroneous. Last, the vegetation communities themselves were oversimplified; hence, the water depth tolerance ranges of individual species were not considered (Poiani and Johnson, 1993b). Nevertheless, the rule-based model is a simple and effective approach to simulating wetland vegetation. The if-then statements in the vegetation subcomponent of the model could be adapted to reflect processes and responses specific to wetland vegetation at Long Point.

3.4.6 SUMMARY

Review of the research related to wetland studies has identified that a variety of techniques have been used to model the dynamic nature of wetland systems in a spatial or non-spatial context. There were also a number of applications to modelling wetland systems discussed, but there were relatively few applications that simulated the spatial distribution of vegetation communities in response to climatic changes or water level fluctuations. Furthermore, the review of the existing models highlighted areas for further work in relation to model development such as improving the spatial and temporal scale of the models and incorporating more variables into the modelling process.

3.5 RESEARCH NICHE

The literature review has identified geographic information systems as an effective and useful tool for assessing historical wetland change and modelling the response of wetland vegetation to change.
A number of studies have successfully applied spatial overlays and modelling techniques to document and model change in wetland systems, but there were a limited number of studies that specifically analyzed historical changes in the structure and spatial patterns of coastal wetland systems over time. Furthermore, there were relatively few studies that integrated a spatial or temporal historical analysis with modelling effects, especially within the context of wetland environments within the Great Lakes. There is limited research quantifying wetland change at Long Point but to date there has been no long-term analysis that examines the implications of climate changes to wetland communities at Long Point and attempts to model those changes in relation to water level fluctuations within a geographic information system. The literature that characterizes the response of wetland vegetation communities to water level fluctuations will help in the development of the wetland model. The geographic information system analysis investigating the implications of climate changes on the Long Point wetland complex will contribute to the area of wetland research by providing an example of the usefulness of geographic information systems in assessing historical change in wetland vegetation systems and relating that change to climatic and water level conditions. The analysis will also provide a comparative example of the applications of a variety of modelling techniques to predict the response of wetland vegetation to climatic changes.

4.0 RESEARCH METHODOLOGY

This chapter discusses the methods that were implemented to assess the effects of water level change as a surrogate for climate change on the Long Point wetland complex. The analysis procedure begins with the collection of wetland classification maps and ancillary data. The pre-processing operations of the data are then outlined. Finally, the statistical and modelling techniques that are employed to quantify wetland vegetation response over time, in relation to water level fluctuations, and simulate vegetation response to climatic changes within the geographic information system ARC/INFO are discussed.

4.1 DATA COLLECTION

The Adaptation and Impacts Research Group (AIRG), Environment Canada provided the mylar wetland vegetation and land use classification maps for the analysis. The classification maps of Long Point were developed as part of the Wetland Vulnerability to Climate Change project initiated by AIRG. The project, funded through Great Lakes 2000, was designed to assess the vulnerability of wetlands along the shoreline of the Great Lakes to climate variability and change - water levels are used as a surrogate for climatic changes. The project builds upon and contributes to a study initiated by the Canadian Wildlife Service (CWS) for the International Joint Commission Water Level Reference (IJCWLR) that includes a database of wetland vegetation for a number of Lake Erie and Lake Ontario wetlands called Wetland Trends Through Time.

Development of the database involved a review of aerial photographs available from the Ontario Ministry of Natural Resources, several Conservation Authorities, and the National Air Photo Library, Natural Resources Canada. Aerial photographs of Long Point representing years of high, medium, and low water levels were selected (Figure 4-1). Black and white and infrared photographs were interpreted for wetland vegetation and land use categories similar to the method developed by the CWS for the IJCWLR (Ecological Services for Planning Limited, 1992). The classification scheme developed for the Wetland Trends Through Time database is provided in Appendix B (Snell and Cecile Environmental Research, 2000). The polygons of the wetland vegetation communities and land use categories were transcribed to mylar overlying Ontario Base Maps (OBMs), which were used as guides (Snell and Cecile Environmental Research, 2001).

Due to the large spatial extent of the Long Point wetland complex, the mylar classification maps for each year were produced as three separate but adjoining sections: a west section containing the mainland, Big Creek marshes and part of the Inner Bay; a middle section, containing most of the Inner Bay marshes; and, an east section, covering the tip of the peninsula that extends out into Lake Erie. The amount of time involved in processing each individual map section was immense; therefore only four of the eight years of data (1945, 1964, 1985, and 1995) were originally included in the analysis. The four years of data were chosen to represent periods of differing water levels. The 1945 data represents a period of medium water levels in which water levels are rising; 1964 reflects one of the lowest water level periods for which data were available; 1985 reflects one of the highest water level periods on record; and, 1995 represents a period of medium water levels in which water levels are declining and, at the time, was the most recent dataset available.

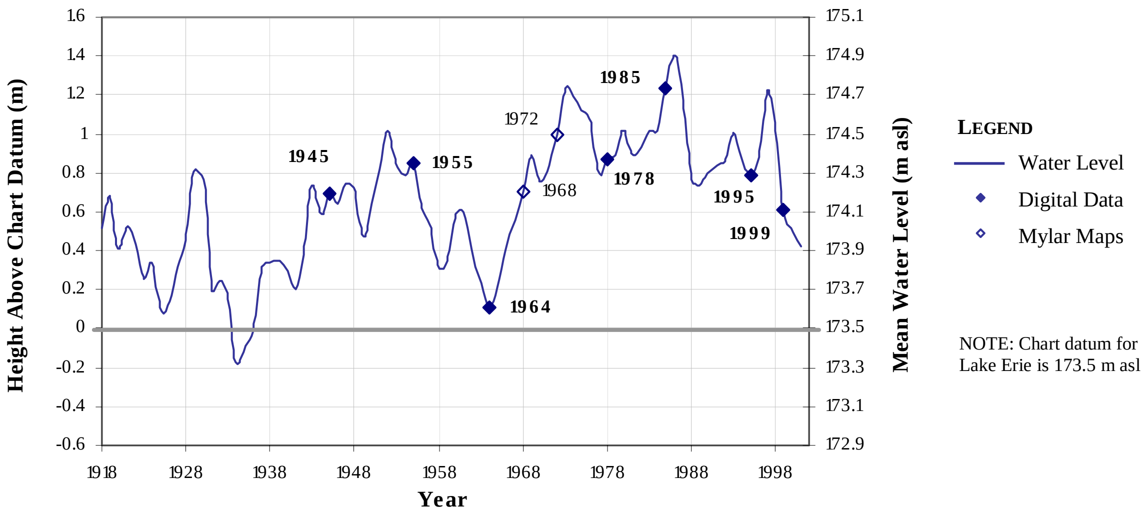


FIGURE 4-1: LAKE ERIE MEAN ANNUAL WATER LEVELS FROM 1918 TO 1999 (IGLD 1985) AND AVAILABILITY OF LONG POINT WETLAND VEGETATION COMMUNITY DATA

Subsequent to processing the abovementioned years of data, three additional years of data (1955, 1978, and 1999) were provided in digital form and included in the analysis. Aerial photographs for 1999 were interpreted, transcribed onto paper, manually digitized by the Long Point Waterfowl and Wetlands Research Fund (LPWWRF), Bird Studies Canada as part of their research examining the recent expansion of *Phragmites* in Long Point, and then provided to AIRG through a data exchange (Wilcox, 2001). The LPWWRF also manually digitized the mylar wetland classification maps for 1955 and 1978 (Wilcox, 2002) that were originally created as part of the Great Lakes 2000 Wetland Vulnerability to Climate Change project. The 1955 data represents a period of medium water levels, where water levels are declining to one of the lowest water levels on record, 1978 represents a period of medium water levels in which water levels are rising to record high levels, and 1999 represents a period in which water levels have been declining from record high levels to medium-low levels.

To accurately assess temporal or spatial trends within wetland vegetation data, it is critical for the date, scale, and quality of the aerial photography to be consistent between the years of data. It is important to obtain aerial photography captured around the same time of the year, thus eliminating any problems associated with differences in the composition and distribution of vegetation that occur during various times of the year. The aerial photographs of Long Point were captured during the summer months for all years except 1964, which was captured in the early spring. The 1964 photographs precede the growing season and therefore exhibit different vegetation growth patterns than the photographs that were collected during the summer months (Snell and Cecile Environmental Research, 2000).

Differences in the scale or resolution of the aerial photography can also have an impact on the results of the trend analysis. Photographs taken prior to 1978 were captured at lower resolutions and, as a result, the classification maps for these years were mapped at a slightly smaller scale. Although some detail was inevitably lost with the lower resolution, the level of detail was still deemed acceptable for analyzing wetland vegetation trends over time (Snell and Cecile Environmental Research, 2000). In addition, the quality of the photographs can have implications on the results of the analysis. The better the photographic quality, the easier it is to distinguish between the various wetland vegetation communities and land use categories. For most years, the quality of the photography was quite good, except for 1945 where some distinction between wetland classes is difficult. Table 4-1 provides a summary of the information pertaining to the collection and interpretation of the aerial photographs.

A number of processing operations were performed on the wetland data in preparation for the temporal and spatial trend analysis and model development. These processes are discussed in detail in the following section.

4.2 DATA PROCESSING

This section outlines the main steps involved in processing the wetland data. The methods for vectorizing the wetland data into the geographic information system ARC/INFO, importing and converting the digital data provided by the LPWWRF, creating a study area boundary and simplifying the wetland classification are discussed. The commands used in ARC/INFO are designated using SMALL CAPITALS. Data flow diagrams outlining the processes used in the analysis are provided in Appendix C.

4.2.1 SCAN VECTORIZING

The mylar wetland classification maps were scanned on a drum-scanner at a resolution of 200 dots per inch (DPI) with the line option and saved as tiff images. The tiff images were converted into grids in the Command Tools module of ARCTOOLS, an interactive button and menu interface in ARC/INFO. The images were converted to grids using nearest-neighbour sampling and square blocking, the optimal method to use when the grids will be vectorized (ESRI, 1999). The grids were then vectorized with the semi-automated tracing features in the Edit Tools module of ARCTOOLS.

In Edit Tools, the grids were displayed in the background environment and used as a reference for the tracing process. New vector coverages were created with arc, node, polygon, and label features. Tic marks with identification numbers were also added to these new coverages at the locations that corresponded with

Table 4-1: Summary of the Long Point Wetland Vegetation Community Data

Year	Date of Photos	Photo Scale	Map Scale	Quality of Photos	Comments/Problems (Notes Provided by Interpreter)
1945	June/July	1:22,000	1:16,000	Fair	• More diversity within classes due to higher water levels than previous years or pre-nutrient pollution • Hard to distinguish emergent from floating or dark emergent from submergent due to scale and quality • *Phragmites* clumped with tall dense emergent • Grass/Sedge appeared different in texture and color than other years • Solid forest cover, therefore hard to distinguish between upland and wetland forest; woodland may be overestimated
1955	July 10	1:15,840	1:16,000	Good	• Higher clumps of vegetation and islands with low white drifts along west side due to sand blown up by Hurricane Hazel, blown seeds or flat dead vegetation • Reduced sharpness of the photographs made interpretation decisions difficult • Hard to distinguish wetland/upland boundary on the point due to thick forests and complex dune shapes
1964	April 9	1:16,000	1:16,000	Very Good	• Taken in early spring, probable shifts to submergent and floating later in the season; extent of emergent, floating and submergent minimal • Growing season signatures different at this time of the year making distinction between some classes difficult • Lowest water levels in past six decades • Road to the west of the Hahn Unit was realigned between 1955 and 1964, eroded away by 1972
1978	Summer	1:10,000	1:10,000	Excellent	• Changes in shoreline between 1972 and 1978 • Storm between 1972 and 1978 piles vegetation mats into striated patterns in the Lee Brown Marsh and creates convoluted and dead end channels • Difficult to distinguish between the wetland and upland boundary on the point (east third section)
1985	July 17	1:8,000	1:10,000	Good	• National Wildlife Area dike and many Big Creek islands built • Difficult to distinguish idle land from bare sand due to bright exposure of photos
1995	July 13	1:10,000	1:10,000	Excellent (but Infrared)	• *Typha* expansion at edge of existing patches • New channels and islands formed; storm shift of some channels • Shadows on vegetation at waters edge show little evidence of vegetation, therefore classified as floating emergent • Signature of *Phragmites* depends on age of the stand, therefore some new patches of *Phragmites* may be included in tall dense emergent or meadow
1999	July 7	1:10,000	1:10,000	Excellent	• No comments or problems noted by interpreter
All Years					• Less diversity over years • Difficult to distinguish between lake, open water and submergent therefore open water is included with submergent • Lake boundary of submergent based on Wilcox/Petrie mapping of submergent vegetation in the Bay • Little bit of shifting in maps prior to 1978 when aligned with the OBMs

(Wilcox, 2001; Snell and Cecile Environmental Research, 2000) the registration marks on the background grids; therefore, the coverages could subsequently be projected into real world coordinates. The interactive tracing tools were then used to vectorize the entire grids. Arcs were automatically added to the coverages as the interactive tracer followed the grid lines. The tracer only stopped for directional input upon arriving at the junctions of two or more grid lines, or when the tracer was unable to decipher or follow the pattern of the grid lines.

The completed coverages were edited for arc and node errors and built to add polygon topology. A code item was added to the attribute tables of the coverages using the ADDITEM command. Unique code values, representing the various wetland vegetation communities and land use categories, were assigned to each polygon in the coverages using a form menu, an interactive button and menu interface used to edit feature attributes within coverages. The code values were based on the Wetland Trends Through Time classification scheme provided in Appendix B (Snell and Cecile Environmental Research, 2000).

The coverages were then registered to real world coordinates. Blank coverages were initially created with the CREATE command, and then PROJECTDEFINE was used to define the following projection information for the coverages:

- PROJECTION: Universal Transverse Mercator (UTM)
- ZONE: 17
- DATUM: NAD27
- UNITS: metres (m)
- SPHEROID: Clarke 1866.

Next, tic marks were generated for the blank coverages. The tic marks received the same identification numbers as the corresponding tics in the vectorized coverages. The vectorized coverages were then projected into the blank coverages with TRANSFORM. The root mean square (RMS) error ranged from 0.001 digitizing units (0.28 m) to 0.020 units (8.22 m), with an average RMS error of 0.009 units (3.04 m). The high RMS errors are due to the quality of original analog maps. The wetland boundaries and registration marks were delineated using a relatively soft lead pencil and as a result some of the registration marks did not translate well during the scanning process and the exact location of the marks had to be estimated during the vectorizing process. Overall, the higher RMS errors will likely have no major impact on the results of the analysis because the wetland data represents natural features with transitional boundaries; therefore, errors were inherently introduced during the aerial photography interpretation and delineation of the wetland vegetation communities and land use boundaries. A verification plot was produced for each coverage year, overlaid with the original mylar map, and examined for missing or mislabelled polygons. Corrections were made to the original unregistered coverages, which were then re-projected into UTM coordinates.

The coverages representing the three map sections for each year were joined together to produce a one continuous coverage of Long Point. The coverages representing the east and west sections were aligned to the middle section with EDGEMATCH, and then the three sections were joined together using the JOIN command. Adjacent polygons along the joined edges were removed with DISSOLVE. Finally, an ARC macro language (AML) program was designed and executed to reclassify the code values within the joined coverages with numeric class values. The AML added a class item to the attribute table then used SELECT and CALCULATE to assign class values to each record based on the original code value.

4.2.2 DATA IMPORT AND CONVERSION

As discussed earlier, the Long Point Waterfowl and Wetlands Research Fund, Bird Studies Canada provided additional digital data for 1955, 1978 and 1999 (Wilcox, 2002; Wilcox, 2001). The 1955 and 1978 data were provided as ARC/INFO export interchange files (e00). The files were converted into coverages with IMPORT; the imported coverages were built to create polygon topology and projected in UTM coordinates. The coverages were then edited in ARCEDIT to resolve any topological errors that were created during the import process. Adjacent polygons with similar code values were dissolved together and the code values were assigned class values with the reclass AML. Although no major problems were observed with the imported 1955 data, the 1978 data appeared to be shifted approximately 60 m to 80 m north in the eastern section. Therefore, new tic marks were added to the 1978 coverage along the outer boundary lines and then these tic marks were used to transform the coverage into a new coverage with proper alignment. The tic marks had the same coordinates and identification numbers as the tic marks used to register the vectorized coverages. The RMS output error was 15.68 m; but no doubt that some of this error originated during the original digitizing process.

The 1999 data was provided as an ArcView (ESRI, 2000) shape file. The SHAPEARC command converted the ArcView shape file into an ARC/INFO coverage. The coverage was cleaned and then the REGIONPOLY command was used to produce a polygon coverage that maintained the original attribute information of the shape file. Information regarding the projection of the coverage was entered using the PROJECTDEFINE command. The reclass AML was executed to reclassify the code values with class values.

Comparison of the shape file with the coverage revealed that many of the polygons within the coverage did not retain the correct attribute information; a problem associated with overlapping polygons created during the digitizing process in ArcView. To overcome this problem, an Avenue Script was written in ArcView to query each polygon in the shape file to determine if more than one polygon intersected with the polygon (i.e. if the polygon spatially overlapped with any other polygons). The smaller polygons overlapping larger polygons tended to retain the attributes of the larger polygons. Therefore, new class, subclass and code items based on the polygon with the smallest area were assigned to the corresponding polygons in the coverage file. Additional editing of the 1999 coverage included removing sliver and spurious polygons with the ELIMINATE command and manually correcting and removing any erroneous or dangling arcs. The coverage was built and unnecessary items were dropped from the attribute table using the DROPITEM command.

4.2.3 BOUNDARY DEFINITION

To study the change in wetland extent within Long Point temporally and spatially, it is important to delineate a definite study area boundary. The exact location of the outer boundary lines of the wetland data were not perfectly aligned between years, therefore a boundary coverage was created to clip all the wetland data to the same spatial extent. The outer boundary lines, presenting the limit of the wetland extent, were extracted from each year of data and merged into a single coverage. All the boundary arcs were deleted from this coverage in ARCEDIT except for the inner most arcs, which represented the minimal spatial extent of the data for the entire study area. The seven years of wetland data were then clipped to the boundary coverage and rebuilt to restore polygon topology.

4.2.4 DATA SIMPLIFICATION

The original classification scheme developed for the Wetland Trends Through Time analysis consisted of over 30 different types of wetland vegetation communities and 20 land use categories. To limit processing time and make the analysis and modelling procedure more manageable, the original code and class values were simplified into ten classes. Wetland vegetation was grouped into communities that had similar responses to water level fluctuations; all non-wetland and non-lake land use categories were grouped together as upland. The simplified classification was based on recommendations from those familiar with Long Point and wetland vegetation response to water level fluctuations. In addition, correlation coefficients were calculated in the statistical software program SPSS (SPSS, 2000) to assess (and/or verify) the relationships between the wetland vegetation communities and water levels. The area values for each wetland vegetation community along with the lake level of each year were used as input in a bivariate correlation analysis to determine the strength and direction of the relationships (Appendix D). The simplified wetland communities are outlined below from wettest to driest.

- LAKE (L)
- OPEN WATER (OW): some evidence or possibility of submergent vegetation
- FLOATING EMERGENT (E1): includes *Lemna*, *Nuphar*, *Nymphaea*, *Zizania*
- EMERGENT (E): includes short emergent, *Sparganium*, *Scirpus*, and trees and shrubs in water
- TALL WET EMERGENT (EW): includes *Typha* and grass/sedge significantly interspersed with water
- TALL DENSE DRY EMERGENT (E2): includes *Typha* and grass/sedge, *Phragmites*
- SHORT WET MEADOW (M1)
- MEADOW (M)
- TREED (T)

- UPLAND (U): non-wetland/non-lake land use classes; includes agriculture, cropland, orchards, industrial, built-up, residential, cottage, marina, park, golf course, river, pond, causeway, bare soil/sand/rock, forested and upland marsh.

An AML was created to reclass the original code and class values with the new simplified code values. The wetland coverages were further simplified as adjacent polygons of similar code values were merged together with DISSOLVE (Figure 4-2a, Appendix E). These simplified coverages were utilized in the spatiotemporal analysis and modelling process.

4.3 SPATIOTEMPORAL TREND ANALYSIS

The temporal trend analysis was conducted in FRAGSTATS*ARC (PMR, 2000), an interface that integrates ARC/INFO with the FRAGSTATS landscape structure and spatial pattern analysis program (McGarigal and Marks, 2001). FRAGSTATS quantifies landscape structure by measuring the areal extent and spatial distribution of patches within a landscape. FRAGSTATS computes a variety of metrics that measure landscape composition and configuration at the patch, class and landscape level. A patch is the basic element or unit of the landscape defined by a particular phenomenon, class refers to a group of patches that share a similar characteristic type, and landscape is simply defined as an area containing a mosaic of patches (McGarigal and Marks, 2001). In this analysis, a patch refers to a single polygon or unit of contiguous cells defined by the simplified wetland vegetation classification, a class refers to a group of patches classified as the same wetland vegetation community, and landscape is the entire study area defined by the study area boundary. Metrics related to area and diversity quantify landscape composition, whereas metrics related to patch, edge, shape, and nearest-neighbour quantify landscape configuration; core area metrics quantify both the composition and configuration of the landscape.

FRAGSTATS utilizes a run wizard that prompts the user to input the run parameters through a series of four steps. The first step prompted for the selection of an input coverage and attribute with which to derive the computations. During each run, a simplified wetland coverage representing a different year was defined as the input coverage and the simplified wetland code (or vegetation community) item was selected as the attribute. The second step defined the data parameters. All ten classes, including lake, were used in the analysis. Lake was included because: (1) the class was deemed an important component of the wetland system at Long Point; (2) there are inter-community transitions between this class and the other wetland communities from year to year; (3) the class is highly influenced by water levels; and, (4) the class is to be included in the modelling process, and therefore necessary for consistency between the analysis and modelling and to allow easier evaluation of the modelling results.

It is acknowledged that the amount of lake area will affect the results of the trend analysis and evaluation of modelling techniques because the lake area comprises a large portion of the total wetland area. To

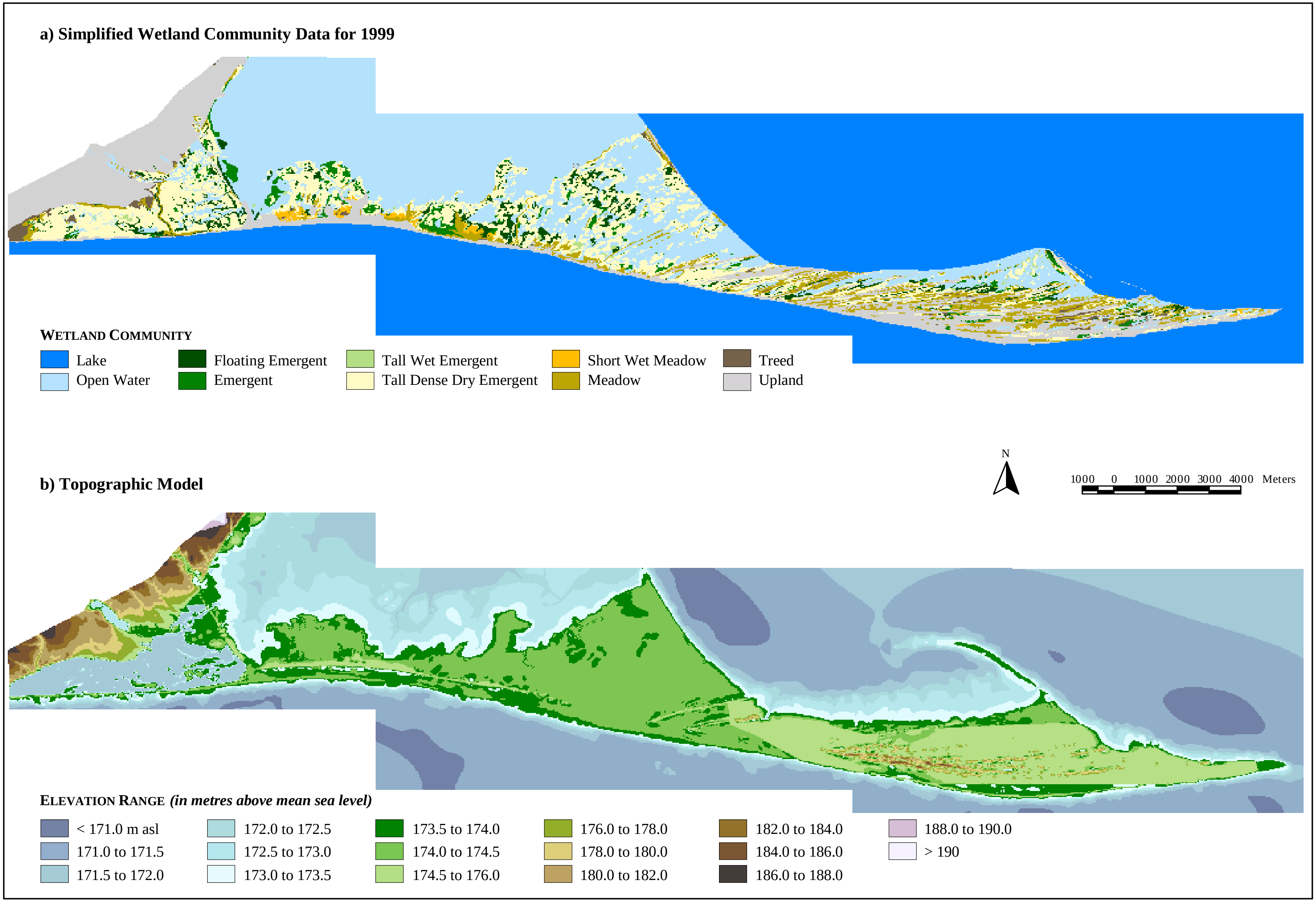


FIGURE 4-2: LONG POINT DATA

determine the effect of the lake area, test runs were completed in FRAGSTATS where the lake class was excluded from the analysis; the general patterns of the results were similar in both cases, though differences did occur in the magnitude and range of some of the landscape metrics. For example, the values of the indices related to density were noticeably larger and wider in range and the values related to landscape shape, diversity, and evenness were slightly larger, but the values related to patch size and variability were smaller and narrower in range. The differences in the density, shape, diversity, and evenness metrics indicate that there are more observed differences in fragmentation, complexity, and diversity within the landscape during periods of declining and rising water level conditions. However, the smaller ranges in the patch size and variability metrics indicates that there are less differences or variability between the patch sizes within the landscape during different water level conditions.

The third step involved the selection of the metrics. Metrics related to area, patch, edge, shape, and diversity were computed in FRAGSTATS for each year of wetland data at the class and landscape level. Metrics at the patch level were of little interpretive value in analyzing the amount and distribution of wetland vegetation communities at Long Point, and thus disregarded. Furthermore, metrics related to core area were not computed in the analysis; exploring these metrics went beyond the scope of this thesis. Descriptions of the individual metrics computed in the analysis are provided in the next chapter. A summary of the input parameters was provided in the final step of the wizard. An output name for the run was entered and then FRAGSTATS was executed. The output INFO tables created for each run (year) were exported into database (dbf) format using the Run Manager. The exported dbf files were then opened up in Microsoft EXCEL (Microsoft Corporation, 1997) for analysis.

Metrics related to nearest-neighbour and contagion and interspersion were calculated with Patch Analyst (Grid) (Rempel, 2000). Patch Analyst (Grid) is an extension within ArcView that includes a user interface for FRAGSTATS that allows grids to be spatially analyzed. The contagion and interspersion metrics could not be computed in the version of FRAGSTATS*ARC that was available, and although several attempts were made to calculate the nearest-neighbour metrics, the function would not run properly in the ARC/INFO interface. The spatial statistics within Patch Analyst (Grid) were derived, exported as dbf tables and opened in EXCEL.

The spatial trend analysis was completed in GRID, a component of ARC/INFO. The wetland community coverages for each year were converted into grids with 12 m resolution using the POLYGRID command; the resolution is consistent with the resolution of the elevation data, which is discussed below. The individual grids were then overlaid together into a single grid with COMBINE. An AML was designed and executed to classify the inter-community changes within the wetland. The value attribute table of the grid was exported as a dbf and opened in EXCEL. The area of change within each wetland community was calculated by multiplying the number of cells representing each unique transition by the grid cell resolution. Percent change to and from each wetland community to other wetland communities was calculated, and overall percent change was noted.

4.4 MODEL DEVELOPMENT

The historical analysis of Long Point culminates with the exploration and development of three simple models for simulating wetland vegetation response to historical water level fluctuations. The input parameters of the models consider either (1) topographic conditions or (2) topographic conditions in relation to the pre-existing wetland vegetation community. This section begins by describing the compilation of the elevation and bathymetry data in order to create a topographical data model that will be used as input for the models. Next, the design processes of the wetland simulation models are outlined. This section concludes by discussing methods for evaluating the performance of the models.

4.4.1 TOPOGRAPHIC DATA

A topographic model was generated from three different sources of data: a bathymetry coverage provided by the National Oceanic and Atmospheric Administration, Great Lakes Environmental Research Laboratory (NOAA, GLERL); a digital land elevation grid provided by the Provincial Geomatics Service Center, Ontario Ministry of Natural Resources (OMNR); and, a point coverage containing spot values derived from OBMs of Long Point. The steps involved in compiling the data into a topographic model of Long Point are discussed.

The bathymetry coverage contained underwater depth measurements that charted the bathymetry of the lake bottom; the depth values ranged from zero to six feet below lake level. The depth values were converted into metres and subsequently subtracted from the chart datum to obtain an elevation value above mean sea level. The bathymetry values were referenced to a chart datum of 173.5 metres above sea level (m asl), the International Great Lakes Datum (IGLD) of 1985 (Lee *et al*, 1996). The IGLD 1985 is a vertical control system for measuring water levels in the Great Lakes Basin. The datum was derived from geodetic surveys completed between 1982 and 1988; the mean year for the surveys was 1985 (USACE, 1996). The bathymetry coverage containing the new elevation values could then be combined with other elevation data to generate a complete topographic model of the land and nearshore at Long Point.

The digital land elevation model of Long Point was produced from contours and digital terrain modelling (DTM) points that were digitized from aerial photographs of the 1970s or from pre-existing hardcopy maps during the OBM mapping process (OMNR, 1997). To accurately predict surface water drainage, the elevation model was hydrologically corrected by the OMNR with drainage information updated by the Long Point Conservation Authority in 2001. The vertical datum for the model was derived from mean sea level established by the Geodetic Survey Division, Energy, Mines and Resources Canada (OMNR, 1997).

Although there is no reference to the actual year of the mean sea level used, it was assumed that the IGLD of 1985 was the vertical reference. This assumption was based on the fact that the lake level on the OBMs, which were used to derive the model, was 173.5 m and this value coincided with the IGLD of 1985 for Lake Erie.

The elevation model was provided in NAD83 format. To be consistent with the other elevation and wetland data, the model had to be projected into NAD27. The PROJECT command was initially used but the results were erroneous. Although the western section of the grid aligned perfectly with the pre-existing wetland data, the middle and eastern sections were shifted approximately 400 m south of their actual locations. It should be noted that on the original model, the grid was shifted approximately 200 m north of the wetland data in the western section, and 200 m south of the data in the middle and eastern sections. In all probability, there was an error with the original projection or alignment of the different OBM sheets during the mapping process.

A number of steps were involved in rectifying the projection of the elevation grid. The projected elevation grid, containing the properly aligned western section, was clipped to the spatial extent of the correctly aligned elevation data. To align the elevation grid correctly in the middle and eastern sections, the projection of the original grid was defined as NAD27 and then the grid was projected into NAD83 format. This new grid was then clipped to represent the area of the elevation data that was correctly aligned with the wetland data. The two clipped and correctly projected elevation grids were used as a backdrop to create a control point coverage. A series of 60 points representing the NAD27 location of prominent features along the shoreline were added to the point coverage. Next, the CONTROLPOINTS command was used to interactively establish links between locations on the original elevation grid and the control point coverage. An affine transformation of the eighth polynomial was completed on the grid within CONTROLPOINTS. The reported input and output RMS errors for the transformation were 18.88 m and 75.15 m respectively. The high RMS error is likely due to the offset between the middle and western section (just offshore of the mainland in the Inner Bay) on the original elevation grid. The greatest amount of error probably occurs in this area, as most other areas of the grid were shifted north or south of their actual location. The entire elevation grid was then adjusted along the links from the original location to their new NAD27 location with the ADJUST command in ARC. The correct NAD27 projection information was defined for the adjusted grid, and then the elevation data was converted into a point coverage with GRIDPOINT.

The final source of input for the topographic model was a point coverage containing the location and value of spot values that were derived from benchmark elevations and contour lines on OBMs. The spot values were referenced to the IGLD 1985. A new point coverage was produced with CREATE to fill voids along the shoreline between the elevation and bathymetry data. A spot item was added to the attribute table of the coverage. Points were added to the coverage in ARCEDIT and spot values were assigned with CALCULATE. The coverage was then built as a point coverage.

The bathymetry point coverage, elevation point coverage, and OBM point coverage were combined into a single coverage with APPEND. A topographic model of Long Point created using TOPOGRID. The topographic model was clipped to the study area boundary then filtered with a low pass filter to remove any anomalies and smooth the surface of the grid (Figure 4-2b). The topographic model could then be combined with the wetland data to develop the wetland simulation models.

4.4.2 SIMULATION MODELS

The wetland vegetation simulation modelling involved the development of three wetland models that were based on two different modelling approaches: decision or rule-based modelling; and, probability and transitional probability modelling. The overall structure of the models is outlined, and then the operational design and processes unique to each of the models are discussed.

4.4.2.1 OVERALL MODEL STRUCTURE

The vegetation simulation models were developed using AML programming in ARC/INFO. The models are initiated at the ARC prompt using the &RUN command. The structures of the three models are identical. The models prompt the user to enter a hypothetical lake level to be used to simulate wetland vegetation response. The lake level must range between 170.79 m asl, the minimum elevation that occurred on the interpolated topographic model, and 180.00 m asl; the maximum level was arbitrarily selected as the upper limit. Depending on the model, the user is prompted to enter a base year and/or water level condition (Figure 4-3). The base year can be selected from any of the seven years of existing wetland

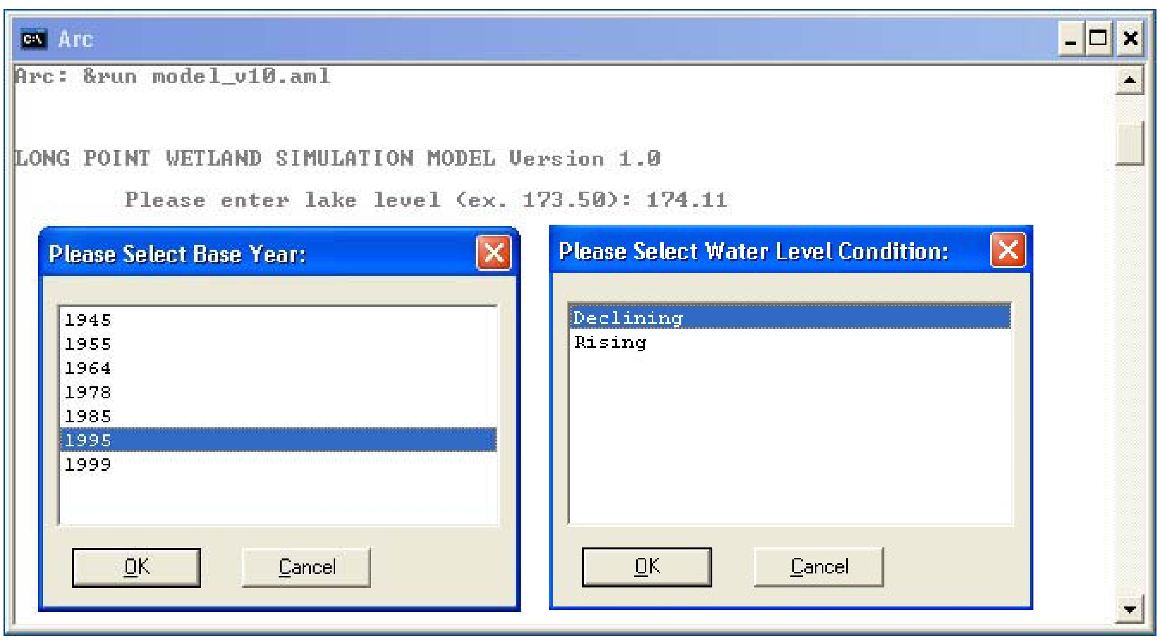


FIGURE 4-3: EXAMPLE OF MODEL INPUT PARAMETERS

vegetation community and land use data for Long Point, but the selection of the water level condition is limited to declining and rising. These two conditions are representative of the general conditions that are occurring between each year of data. Although it would have been optimal to include stable conditions and to have separate declining and rising conditions based on the variation and magnitude of change between periods, it would not have been practical to construct and validate such a model with only seven years of wetland data as a fundamental principle in the validation of predictive models "is that the data against which model predictions are tested must not [be] used on the construction of the model" (Kirchner, 1994, 368).

The data processing and output file generation are then completed in GRID. The lake level entered by the user is used to derive lake level and lake depth grids. A lake level grid is created in which all cells within the grid are assigned a value equal to the hypothetical lake level. This lake level grid is then subtracted from the topographic model to derive a lake depth grid; the value of each cell represents the height in metres above or below the hypothetical lake level. The lake depth grid, along with the raster grid representing the existing wetland vegetation communities for the base years selected, are then used as input to derive a simulated wetland vegetation grid; only the lake depth grid is used as input for the probability model. The input grids are processed on a cell-by-cell basis with DOCELL. The operations completed on the cells vary depending on the model; the processes implemented within each of the models are described in detail below. An output grid representing the simulated wetland vegetation is generated and displayed within the GRID display environment (Figure 4-4). The AML program terminates when the user types QUIT at the GRID prompt. A summary of the model parameters, inputs, and outputs is provided in Appendix G.

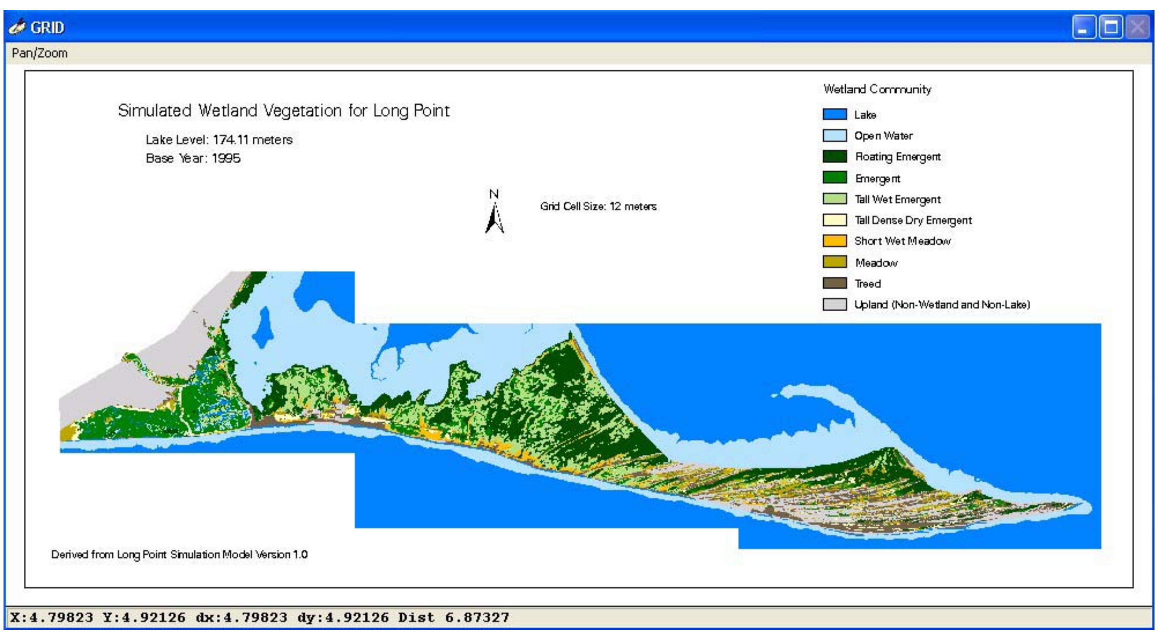


FIGURE 4-4: EXAMPLE OF MODEL OUTPUT

4.4.2.2 RULE-BASED MODEL

Development of the rule-based model was based on the vegetation subcomponent of the wetland simulation model developed by Poiani and Johnson (1993a; 1993b). A series of if-then statements were applied to model changes in wetland vegetation community from the base year to the year represented by the hypothetical lake level according to the pre-existing wetland vegetation, the tolerance ranges of the different wetland vegetation communities, and the adjacency of lake, open water, and upland cells. The tolerance ranges were synthesized from available literature and summarized in Table 2-1 (Chapter 2). Three versions of the rule-based model were developed and the rules associated with each are discussed.

<u>VERSION 1.0</u>

The initial version of the model (1.0) generally adheres to the tolerance ranges of each wetland community and assumes that a wetland vegetation community can only tend to the community immediately above or below that community's specific tolerance range. For example, cells that are lake will remain lake if the depth of the cell is greater than 200 cm. If the cell is less than 200 cm in depth (i.e. the lake level has declined), the lake cell will develop into an open water community since the shallower water promotes the growth of aquatic floating plants. If the cell is less than 200 cm in depth and if the cell is adjacent to upland cells, the cell becomes upland. It is assumed that if the depth of the cell decreases to below 200 cm, lake levels have declined enough to allow sandy shallow areas along the shoreline, which are designated as upland communities, to become exposed. Therefore as water level decline, these lake areas convert to upland.

Cells that are open water change to lake if the depth of the water is greater than 200 cm (i.e. water levels rose). If the depth ranged from 60 to 200 cm, the optimal depth range for open water communities, then the cell remained open water. If the depth was less than 60 cm, then conditions promoted the spread of floating emergent vegetation into the cell. Similar if-then statements were constructed for the fate of floating emergent, emergent, and treed communities based on their water depth tolerances.

Since the simplified wetland classification included wetter and drier tall emergent and meadow communities, the fate of tall emergent and meadow communities included an additional component to address changes to these different communities. Cells that are tall emergent tend to emergent vegetation if the depth of the cell is greater than 30 cm; tall emergent vegetation dies off and emergent vegetation flourish with higher water levels. If the depth of the cell is less than 30 cm and the cell value is less than 30 cm above the lake level, the cell becomes a tall wet emergent. This signifies a cell that is significantly interspersed with water, and rather than tending to emergent vegetation, the tall emergent is classified as wet. If the cell is between 30 and 50 cm above the lake level (the optimal range for tall emergent) then the cell is designated as a tall dense dry emergent. Finally if the cell is greater than 50 cm above the lake level, lake levels have declined, and in response to these drier conditions the tall emergent vegetation tends to meadow. A similar if-then statement was constructed for the short wet and meadow communities.

The fate of upland was addressed differently. Upland cells will most likely stay upland unless the cell experiences prolonged periods of higher water levels, therefore a depth value of zero was used to distinguish change in upland cells rather than using the tolerance range for the community. A value of zero indicates cells with depths that are equal to the lake level. Depth values that are greater than zero represent cells containing water and values less than zero represent cells without water. Therefore, cells that are upland will remain upland if the depth of the cell is above the lake level. If the cell is below lake level, the upland cell changes to treed vegetation. It is assumed that these cells contain upland treed vegetation which are less tolerant of periodic flooding than wetland treed vegetation, thus as water levels rise and these cells become flooded, the vegetation will tend to wetland treed. If the cell is below lake level and adjacent to any lake cells, the cell becomes lake. The if-then statements applied in Version 1.0 are summarized in Table 4-2.

TABLE 4-2: DECISION RULES FOR WETLAND COMMUNITY RESPONSE IN VERSION 1.0

(1) *Fate of lake*:
- IF VEG = L AND WDC = 8 THEN VEG = L
- IF VEG = L AND WDC ≤ 7 THEN VEG = OW
- IF VEG = L AND WDC ≤ 7 AND ADJ = U THEN VEG = U

(2) *Fate of open water*:
- IF VEG = OW AND WDC = 8 THEN VEG = L
- IF VEG = OW AND WDC = 7 THEN VEG = OW
- IF VEG = OW AND WDC ≤ 6 THEN VEG = E1

(3) *Fate of floating emergent*:
- IF VEG = E1 AND WDC ≥ 7 THEN VEG = OW
- IF VEG = E1 AND WDC = 6 THEN VEG = E1
- IF VEG = E1 AND WDC ≤ 5 THEN VEG = E

(4) *Fate of emergent*:
- IF VEG = E AND WDC ≥ 6 THEN VEG = E1
- IF VEG = E AND WDC = 5 THEN VEG = E
- IF VEG = E AND WDC ≤ 4 THEN VEG = E2

(5) *Fate of tall emergent (wet and dense dry emergent)*:
- IF VEG = EW or E2 AND WDC ≥ 6 THEN VEG = E
- IF VEG = EW or E2 AND WDC = 5 THEN VEG = EW
- IF VEG = EW or E2 AND WDC = 4 THEN VEG = E2
- IF VEG = EW or E2 AND WDC ≤ 3 THEN VEG = M

(6) *Fate of meadow (including short wet meadow)*:
- IF VEG = M1 or M AND WDC ≥ 6 THEN VEG = E2
- IF VEG = M1 or M AND WDC = 5 THEN VEG = M1
- IF VEG = M1 or M AND WDC = 3 or 4 THEN VEG = M
- IF VEG = M1 or M AND WDC ≤ 2 THEN VEG = T

(7) *Fate of treed*:
- IF VEG = T AND WDC ≥ 3 THEN VEG = M
- IF VEG = T AND WDC = 2 THEN VEG = T
- IF VEG = T AND WDC = 1 THEN VEG = U

(8) *Fate of upland*:
- IF VEG = U AND WD < 0 THEN VEG = U
- IF VEG = U AND WD ≥ 0 THEN VEG = T
- IF VEG = U AND WD ≥ 0 AND ADJ = L THEN VEG = L

VEG = wetland vegetation community, WD = water depth in centimetres (cm), WDC = water depth category, ADJ = cell adjacency.
Wetland communities: L = lake, OW = open water, E1 = floating emergent, E = emergent, EW = tall wet emergent, E2 = tall dense dry emergent, M1 = short wet meadow, M = meadow, T = treed, U = upland
Water depth categories: (1) ≤ -100 cm, (2) –99 to –80 cm, (3) -79 to –50 cm, (4) –49 to –30 cm, (5) -29 to 30 cm, (6) 31 to 60 cm, (7) 61 to 200 cm, (8) > 200 cm; negative "water depth" values indicate height above lake level

VERSION 1.1

The if-then statements applied in Version 1.1 of the rule-based model were more lenient in terms of the tolerance depth ranges of the different vegetation communities. Generally, the existing vegetation tended to a wetter wetland community if the depth of the water in the cell was greater than 30 cm. If the depth of the water was less than 30 cm and within one drier depth category of the community's optimal range, the vegetation remained the same. If the depth of the water was less than 30 cm and greater than one drier depth category of the community's optimal range, the vegetation tended to the next drier community.

Rather than using mean lake level (i.e. a water depth of 0 cm) to mark the transitions between wetland vegetation communities, a water depth of 30 cm was used as a reference so that vegetation changes only occurred within a cell when there were significant changes in the water depth of the cell. The other notable difference in the set of decision rules is that adjacency of open water to lake was included under the fate of open water to minimize changes to open water communities along the shoreline, particularly along the southern shore of the peninsula. The fates of several wetland communities are discussed below.

For the fate of emergent vegetation, cells that are emergent will change to floating emergent if the depth of the cell is greater than 30 cm in depth (i.e. lake levels have risen). If the depth of the cell is less than 30 cm, and less than 50 cm above lake level, the emergent vegetation will remain; the optimal tolerance range for emergent vegetation is between 30 cm above and below lake level. If the depth of the cell is greater than 50 cm above lake level, the emergent cell will tend to tall dense and dry emergent vegetation. This principle is also applied to the fates of lake, open water, floating emergent, and treed communities.

The fate of tall emergent and meadow communities were addressed slightly differently. The vegetation tended to the wetter wetland community, emergent and tall emergent respectively, if the depth of the cell was greater than 60 cm. The vegetation remained or tended to the wetter form (tall wet emergent and short wet meadow) if the depth of the cell is less than 60 cm and greater than 30 cm. This depth range suggests that the cell has become significantly interspersed with water. Tall emergent vegetation remained or tended to the drier form (tall dense dry emergent) if the depth of the cell is less than 30 cm or if the value of the cell is less than 50 cm above lake level; the optimal range for tall emergent was between 30 and 50 cm above lake level. The optimal depth range for meadow vegetation is between 50 and 80 cm above lake level. Therefore, meadow vegetation remained the same if the depth of the cell ranged from less than 30 cm in depth to 80 cm above lake level. If the depth of the water was greater than 50 cm the tall emergent vegetation tended to meadow, and if the depth was greater than 80 cm meadow tended to treed vegetation.

For the fate of upland, cells that are upland remained upland if the cell was greater than 30 cm above lake level. Upland tended to treed vegetation if the depth of the cell ranged from 30 cm above to 60 cm below lake level. Upland cells where the depth of the cell was greater than 60 cm tended to lake, and if the depth of the cell was greater than zero and adjacent to lake, the cell also changed to lake. A summary of the if-then statements applied to Version 1.1 is provided in Table 4-3.

TABLE 4-3: DECISION RULES FOR WETLAND COMMUNITY RESPONSE IN VERSION 1.1

(1) *Fate of lake*:

IF VEG = L AND WDC ≥ 6 THEN VEG = L
IF VEG = L AND WDC ≤ 5 THEN VEG = OW
IF VEG = L AND WDC ≤ 5 AND ADJ = U THEN VEG = U

(2) *Fate of open water*:

IF VEG = OW AND WDC ≥ 6 AND ADJ = L THEN VEG = L

(5) *Fate of tall emergent (wet and dense dry emergent)*:

IF VEG = EW or E2 AND WDC ≥ 7 THEN VEG = E
IF VEG = EW or E2 AND WDC = 6 THEN VEG = EW
IF VEG = EW or E2 AND WDC = 4 or 5 THEN VEG = E2
IF VEG = EW or E2 AND WDC ≤ 3 THEN VEG = M

(6) *Fate of meadow (including short wet meadow)*:

IF VEG = M1 or M AND WDC ≥ 7 THEN VEG = E2
IF VEG = M1 or M AND WDC = 6 THEN VEG = M1

IF VEG = OW AND WDC ≥ 6 THEN VEG = OW
IF VEG = OW AND WDC ≤ 5 THEN VEG = E1

(3) *Fate of floating emergent*:

IF VEG = E1 AND WDC ≥ 6 THEN VEG = OW
IF VEG = E1 AND WDC = 5 THEN VEG = E1
IF VEG = E1 AND WDC ≤ 4 THEN VEG = E

(4) *Fate of emergent*:

IF VEG = E AND WDC ≥ 6 THEN VEG = E1
IF VEG = E AND WDC = 4 or 5 THEN VEG = E
IF VEG = E AND WDC ≤ 3 THEN VEG = E2

IF VEG = M1 or M AND WDC = 3 or 4 or 5 THEN VEG = M
IF VEG = M1 or M AND WDC ≤ 2 THEN VEG = T

(7) *Fate of treed*:

IF VEG = T AND WDC ≥ 6 THEN VEG = M
IF VEG = T AND WDC = 2 or 3 or 4 or 5 THEN VEG = T
IF VEG = T AND WDC = 1 THEN VEG = U

(8) *Fate of upland*:

IF VEG = U AND WDC ≤ 4 THEN VEG = U
IF VEG = U AND WDC = 5 or 6 THEN VEG = T
IF VEG = U AND WDC ≥ 7 THEN VEG = L
IF VEG = U AND WD ≥ 0 AND ADJ = L THEN VEG = L

VEG = wetland vegetation community, WD = water depth in centimetres (cm), WDC = water depth category, ADJ = cell adjacency.
Wetland communities: L = lake, OW = open water, E1 = floating emergent, E = emergent, EW = tall wet emergent, E2 = tall dense dry emergent, M1 = short wet meadow, M = meadow, T = treed, U = upland
Water depth categories: (1) ≤ -100 cm, (2) –99 to –80 cm, (3) -79 to –50 cm, (4) –49 to –30 cm, (5) -29 to 30 cm, (6) 31 to 60 cm, (7) 61 to 200 cm, (8) > 200 cm; negative "water depth" values indicate height above lake level

4.4.2.3 Probability Model

The probability model was based on the likelihood of certain wetland communities occurring at specific depth ranges. To determine the probabilities, the wetland vegetation grids for 1985 and 1995 were overlaid with lake depth grids, derived from the actual lake level during these years, using the COMBINE command in GRID. The data for 1985 were selected to represent periods of rising water levels, and 1995 was selected to represent periods of declining water levels. These years of data were chosen for three main reasons: (1) the scales of the original data are consistent between the two years, (2) the magnitude of change in mean water levels from the previous period is similar, and (3) enough time had passed from the previous year of data for the wetland vegetation to respond.

The lake depth values on the combined grids were then reclassified into depth categories with AMLs. The initial version of the model (2.0) was based on the wetland community tolerance ranges established from the literature. Two additional versions of the model were developed where the number of depth categories increased and the interval range for the categories decreased to 10 cm. For Version 2.1, 32 depth categories were created. The categories ranged from greater than 200 cm in depth to greater than 100 cm above the lake level; these limits represented the upper and lower tolerance ranges for lake and upland communities defined by the literature. For Version 2.2, the depth categories ranged from greater than 320 cm in depth (where there was a 100 % probability of lake occurring for all years) to over 1200 cm above the lake level (where there was a 100 % probability of upland occurring for all years). A total of 154 depth categories were derived for this version.

Next, FREQUENCY was used to create a summary INFO file of all the unique cell combinations of depth categories and wetland vegetation communities in the combined grids. The summary INFO files were exported as dbf files with INFODBASE and opened in EXCEL for further computation. To derive the probabilities a number of intermediate calculations were completed. First, the total area of each wetland community within each depth category was calculated by multiplying the number of cells of each unique combination by the grid cell resolution. Next, the total area of each depth category was determined by totalling the area of all the wetland communities within that depth category. Last, the probability of a wetland community occurring within the depth category was calculated by dividing the total area of each wetland community that occurs within every depth category by the total area of that depth category.

The probabilities were sorted from greatest to least likelihood of occurring at a particular depth range. Probability ranges were then established for the particular depth range category using the cumulative sum of each probability within that depth range. The probability ranges were then applied to a series of if-then statements within each version of the model. The probabilities for 1985 were applied within an if-then statement for rising water level conditions, and the probabilities for 1995 were applied within an if-then statement for declining water level conditions.

The probabilities, along with a random grid, are then used to derive a simulated wetland vegetation grid. A random grid is generated during each run of the model in GRID; each cell in the grid is randomly assigned a number ranging in value from 0 to 1. Depending on the value of the cell in the random grid, and the probabilities of a certain wetland community occurring, a wetland community value is assigned to that cell in the output grid. For example, in Version 2.2 during declining water level conditions, the probability of lake and open water occurring between a depth range of 280 to 270 cm is 0.988281 and 0.0011719 respectively; no other wetland communities occur at this depth. Therefore, if the random cell value is less than 0.988281, the cell becomes lake. If the random cell value ranges between 0.988281 and 1 (0.988281 + 0.0011719), the cell becomes open water. The if-then operations are complete when all the cells in the random grid have been processed.

4.4.2.4 VEGETATION TRANSITION MODEL

The transition probability model (Version 3.0) was based on the likelihood of wetland vegetation communities changing to another wetland community in relation to declining or rising water levels. The transition probabilities were derived from the overlay of the wetland vegetation grids for 1978, 1985, and 1995 in the spatial trend analysis. The vegetation community transitions between 1978 and 1985 were chosen to represent a period of rising water levels, and the period from 1985 to 1995 was selected to represent a period of declining water levels. These periods were chosen for several reasons: (1) the scale of the original data is consistent between the two years; (2) the magnitude of decline or rise in water levels from the previous period are similar; and, (3) to remain consistent with the years chosen for the probability model.

Using the change matrix analysis tables derived from the spatial analysis, transition probabilities were calculated in EXCEL by dividing the number of cells that represented each unique transition within a specific wetland community by the total number of pixels of that wetland community for the base year.

The transition probabilities were sorted from greatest to least likelihood of occurring for a particular wetland community and applied to a series of if-then statements similar to the probability model. Based on a random grid generation, the existing wetland vegetation community, and the transition probabilities, output grids were created.

4.4.3 MODEL EVALUATION

The accuracy of the models in simulating wetland vegetation growth was evaluated by comparing the area and spatial distribution of the simulated wetland vegetation communities to the historical wetland vegetation community and land use data. Simulations were run on the three models using each of the seven base years of wetland vegetation community data. The actual mean lake level values for these years were used in the simulations. The simulated output grids were overlaid with the actual wetland vegetation grids with COMBINE to provide an indication of the spatial accuracy of the models. An AML was designed and executed to identify correctly predicted cells and to calculate the difference between the actual and predicted wetland community values.

There were several basic measurements used to assess the accuracy of the models. The simulated wetland results produced by each model were used to calculate the percentage of cells correctly predicted for the entire wetland landscape and for each wetland community class. The ability of each model to correctly predict cells within the wetland (at the landscape level) was computed for every simulated year using the following equation in EXCEL:

$$\text{Percent of wetland cells correctly predicted} = \frac{\text{Total number of cells correctly predicted in the wetland}}{\text{Total number of cells in the wetland}} \times 100$$

Similarly, the percentage of correctly predicted cells within each wetland vegetation community (at the class level) was also calculated for every simulated year. The overall accuracy of each model in simulating wetland community response at the landscape and class levels was determined by averaging the percentages of correctly predicted cells within the entire wetland and within the individual wetland communities, respectively. For the models that included water level conditions as an input variable, the accuracy rates in predicting community response for declining and rising water levels were also computed by averaging the simulated results for the years that represented periods of declining or rising water levels. The percentages of cells incorrectly predicted were further analyzed to determine the magnitude of difference between the actual and predicted wetland community values and whether these predicted values were drier or wetter than (above or below) the actual wetland values; the spatial distributions of the errors were also examined to determine the locations of the errors. Furthermore, aggregate areas statistics were generated for the predicted wetland vegetation communities from each simulated year and compared to the actual area to determine the degree in which the areas of the wetland vegetation communities were over or underestimated; the actual and predicted numbers of cells for each simulated year were converted to area values in hectares. Finally, the simulated wetland distribution for each year and model were visually compared to the actual wetland vegetation and land use distribution.

4.5 SUMMARY

This chapter provided a summary of the methods implemented within a geographic information system to assess the effects of climate variability and water level change on wetland communities at Long Point. The methodology outlined the collection and pre-processing of the wetland and ancillary data, the temporal and spatial analysis of the wetland data, and the development of several wetland vegetation simulation models. The results of the spatiotemporal and modelling analyses are discussed in the following chapters. Chapter 5 presents the temporal and spatial response of the wetland vegetation communities to climate change in relation to water level fluctuations. Chapter 6 presents the results of the wetland vegetation simulation modelling.

5.0 SPATIOTEMPORAL TREND ANALYSIS RESULTS AND DISCUSSION

This chapter presents the results of the spatiotemporal analysis of the Long Point wetland complex. In the first section, the temporal trend analysis, changes in the composition and configuration of the wetland complex and vegetation communities are characterized over time and in relation to water level fluctuations. The temporal analysis also summarizes the direction and magnitude of change within each wetland vegetation community. The second section expands on the temporal analysis through a spatial analysis that explores the type and location of change that occurs within the wetland vegetation communities. A synopsis of the wetland changes with respect to declining and rising water levels and projected climate change concludes the chapter.

5.1 TEMPORAL ANALYSIS

The composition and configuration of the Long Point wetland complex can be characterized through a variety of metrics that measure area, patch, edge, nearest-neighbour, shape, diversity, and interspersion at the landscape and class level. A temporal comparison of these metrics from 1945 to 1999 provides an indication of how Long Point has changed over time and how the wetland vegetation has responded to fluctuations in water levels and other factors. First, the metrics used in the analysis are described. Then the metrics are summarized to characterize the response of the wetland vegetation communities from year to year and associated with different water levels at the landscape and class scale. Finally, the class area statistics are further analyzed to summarize the direction and magnitude of change with the wetland vegetation communities.

5.1.1 DESCRIPTION OF METRICS USED TO ASSESS CHANGE

The landscape and class metrics used in the analysis are discussed. These metrics are grouped into seven descriptors of landscape structure and composition: area, patch density, patch size and variability; edge, nearest-neighbour, shape, diversity, and, contagion and interspersion.

5.1.1.1 AREA METRICS

Several metrics have been used to quantify area within Long Point. Total area (TA) is the area of the entire landscape included in the analysis. Since each year of wetland coverage was clipped to the same spatial extent, TA is not a particularly useful landscape descriptor for Long Point; the metric is used, however, to derive other more meaningful indices. Class area (CA) is the total area of a particular class type within the landscape. Here, there are ten classes related to wetland vegetation community such as open water, emergent, meadow, and treed vegetation. Percent of landscape (PLAND) expresses class area as a percentage of the total landscape area. A class type becomes increasingly rare within the landscape as the values of CA and PLAND approach zero and increasingly dominant within the landscape as the values approach the total landscape area or 100 %. Largest patch index (LPI) is the percentage of the total area that is comprised of the largest patch. Patch size becomes increasingly smaller as LPI approaches zero; a LPI of 100 % indicates that the landscape is composed entirely of a single patch (McGarigal and Marks, 2001).

5.1.1.2 Patch Density, Patch Size and Variability Metrics

Patch density, size and variability metrics are useful indicators of fragmentation across the landscape and spatial heterogeneity within particular class types. Number of patches (NP) represents the total number of patches within a class or the entire landscape. Patch density (PD) expresses the number of patches in per unit of area. Smaller values of NP and PD indicate less fragmentation across the landscape or class area; larger values indicate more fragmentation. Mean patch size (MPS) represents the average size of the patches within the total class or landscape area. The larger the value of MPS, the more homogeneous and less fragmented the landscape is; conversely the smaller the value, the more heterogeneous and fragmented the landscape becomes (McGarigal and Marks, 2001; Kienast, 1993).

Patch size standard deviation (PSSD) and coefficient of variation (PSCV) quantify the amount of variability within patch sizes. PSSD measures the absolute variability and PSCV measures the relative variability as a percentage of the mean; both assume a normal distribution in patch size. PSCV is preferred for interpreting landscape structure as PSSD can be misleading if the mean patch size is not considered in conjunction with the interpretation of PSSD (McGarigal and Marks, 2001). PSCV and PSSD values of zero indicate there may only be one patch or all the patches within the class or landscape are the same size. As the values increase, the patches become increasingly fragmented and the landscape or class becomes more spatially heterogeneous.

5.1.1.3 Edge Metrics

Metrics quantifying edge provide further indication of fragmentation and spatial heterogeneity. Total edge length (TE) measures the absolute length of all edges within a particular class or the entire landscape. Edge density (ED) expresses the total edge length per unit of area. The landscape becomes more homogeneous and classes less fragmented as the values of TE and ED approach zero. The landscape becomes spatially heterogeneous and the classes more fragmented as the values increase. Although both statistics are computed in this analysis, the statistics are redundant since the spatial extent of the study area is identical between years.

5.1.1.4 Nearest-Neighbour Metrics

Nearest-neighbour metrics are additional measures of fragmentation. The distance between a patch and the nearest-neighbouring patch of the same class is measured from edge to edge. There are two nearest-neighbour metrics quantified: mean nearest-neighbour distance (MNN) and mean proximity index (MPI). MNN measures patch isolation by averaging the shortest distances between neighbouring patches of the same class type. To derive a MNN for the entire landscape, the individual MNN class values are averaged.

The higher the value, the more isolated the patches are distributed across the class or landscape area; the usefulness of MNN is limited if the distribution of patches within the landscape is complex (McGarigal and Marks, 2001).

MPI incorporates MNN to measure the degree of patch isolation and fragmentation between patches of similar class types within a specified neighbourhood or search radius of the focal patch. The neighbourhood threshold depends on the phenomena under investigation; here the default of 2000 m was used. At the class level, a value of zero indicates that there are no patches of the same class type within a specified neighbourhood. As the value increases, there is less isolation between patches of the same type and less fragmentation in the distribution of the patch types. MPI has not been quantitatively evaluated as a measure of the overall complexity of landscape, so interpretation at the landscape scale may be difficult (McGarigal and Marks, 2001).

5.1.1.5 Shape Metrics

The complexity of patch shapes is quantified by a number of metrics. Landscape shape index (LSI) is the ratio of total edge to area and indicates the complexity of the class or landscape shape compared to a standard shape. Mean shape index (MSI) is an average of all the patch shape indices for a particular class or the entire landscape. An area-weighted mean shape index (AWMSI) is also calculated where weights are applied to patches depending on their size. Heavier weights are applied to larger patches with the assumption that larger patches play a more dominant role within the landscape. Therefore, AWMSI should only be used when the role of the patch is important in the function of the landscape. Shape indices values of one indicate the simplest of shapes, a circle, and as the values increase the shape becomes more complex and irregular. An additional measure of shape complexity is mean patch fractal dimension (MPFD), which indicates the degree of complexity of patch shapes within a particular class type or the landscape by considering the relationship between the perimetre and area of the patches. An area-weighted mean patch fractal dimension (AWMPFD) is also calculated. Fractal values range from one for simple shapes to two for complex and convoluted shapes (McGarigal and Marks, 2001; Kienast, 1993).

5.1.1.6 Diversity Metrics

Patch richness and evenness are important components that influence diversity within the landscape. Patch richness (PR) is the total number of patch types present within the landscape. It is important to note that PR does not consider the relative abundance or the spatial arrangement of patches within the landscape (McGarigal and Marks, 2001). Patch richness density (PRD) expresses the patch richness within the landscape area in per unit of area and relative patch richness (RPR) expresses patch richness as a percentage of the total maximum patch richness within the landscape. The richness metrics are meaningless for the Long Point data because the number of wetland vegetation communities and spatial extent are constant between each year of data. Of importance is patch evenness, which measures the distribution of area among different class types. Evenness is expressed as a ratio of the observed diversity to maximum diversity; maximum diversity occurs when all patch or class types are equally abundant within the landscape (Southwood and Henderson, 2000; Brower *et al.*, 1989). There are three measures of evenness: Shannon's evenness index (SHEI), Simpson's evenness index (SIEI), and a modified Simpson's evenness index (MSIEI). Values of zero indicate that there is no diversity within the landscape; the landscape is dominated by a single patch type or contains only one patch. As values approach zero, the distribution of area among different patches types becomes more uneven across the landscape and as values approach one, this distribution becomes more even. Values of one indicate that the distribution of patch type area is perfectly even across the landscape (McGarigal and Marks, 2001).

There are three metrics that specifically quantify diversity within the landscape: Shannon's diversity index (SHDI), Simpson's diversity index (SIDI), and a modified Simpson's diversity index (MSIDI). The most commonly used diversity metric is SHDI, which measures the uncertainty of different patch types occurring in equal proportions within the landscape (Kienast, 1993). The metric provides a relative index for comparing landscapes over time. SHDI is based on information theory and is sensitive to patch richness (PR). The metric estimates diversity for the landscape using a sample, therefore the metric should only be used when patch richness (number of patches) is greater than 100. Many studies have applied SHDI to landscapes where patch richness was less than ten, and in these cases, the metric should be used as a summary of dominance rather than diversity (Steinhardt *et al.*, 1999). For this analysis, SHDI will be mainly used as a summary of dominance in the landscape. SIDI measures the probability of randomly selecting different patch types within a sample of the landscape. SIDI is more intuitive, less sensitive to richness and the presence of rare class types, and more sensitive to evenness. A modified Simpson's index (MSIDI) transforms the probability values into values that are more similar to SHDI and other general diversity indices (McGarigal and Marks, 2001). As the values of the diversity indices approach zero, the landscape becomes less diverse and contains only a few patches; a value of zero indicates the landscape is composed of a single patch. As the diversity indices increase, the numbers of patch types increase within the landscape or the proportion of area among different patch types become more equal (McGarigal and Marks, 2001; Zaizhi, 2000).

5.1.1.7 Contagion and Interspersion Metrics

Finally, the interspersion of patches within the landscape is quantified with the interspersion and juxtaposition index (IJI). The index measures the relative observed level of interspersion among patch types as a percentage of the maximum level of interspersion given the total number of patch types. Lower percentages indicate that the patch types are poorly interspersed across the landscape, or in other words, the distribution of patch type adjacencies is less proportionate. Higher percentages indicate that patch types are well interspersed across the landscape or patches of similar class types of more equally adjacent to each other (McGarigal and Marks, 2001). IJI measures the interspersion between each patch within the landscape at the landscape level, and measures the relative interspersion of a focal patch of a particular class type with all other patches within that class type at the class level (McGarigal and Marks, 2001; Elkie *et al.*, 1999).

5.1.2 LANDSCAPE METRICS

The landscape metrics computed for Long Point indicate that the composition and configuration of the wetland complex has changed considerably between 1945 and 1999. It is important to note that there are some discrepancies and inconsistencies within the metrics due to differences in the scale and quality of the aerial photographs used to map the wetland vegetation. The aerial photographs taken prior to 1978 were captured at lower resolutions and hence some detail was lost compared to later years. Smaller patches within the landscape were not mapped during the interpretation process and as a result, these years generally had fewer numbers of patches within the landscape. These patches were generally larger in area as well. Further detail was lost in the 1945 photographs due to the poor quality of the data. It was hard to distinguish between similar types of wetland vegetation such as wetland treed and upland, or emergent, floating emergent, and submergent vegetation. The landscape metrics computed for Long Point are presented below. A summary of the metrics is provided in Table 5-1.

5.1.2.1 AREA METRICS

There was little change observed in the LPI over time. The largest patch within the landscape was the lake, which ranged from 55.2 % of the total landscape in 1964, the lowest water level period on record, to 56.4 % in 1985, the highest water level period on record. There was a small positive trend in the LPI in response to water level fluctuations as the metric increased with increasing water level.

5.1.2.2 PATCH DENSITY, PATCH SIZE AND VARIABILITY METRICS

There were noticeable changes in the NP and PD over time. In 1945, there were 977 patches in the landscape. Between 1945 and 1955, a period of rising water levels, the number of patches increased by a total of 617 patches. NP experienced a decrease of 148 patches from 1955 to 1964, as water levels declined to record lows. Between 1964 and 1985, the number of patches in the landscape increased by 2092 patches as water levels rose to record high levels; approximately 69.9 % of this increase occurred between 1964 and 1978. Between 1985 and 1995, as water levels declined, NP decreased by 1375 patches. NP decreased by another 54 patches from 1995 to 1999, as water levels continued to decline. The most notable changes within PD occurred between 1964 and 1999. In 1964, PD measured 4.6 patches per 100 ha; the PD increased by 6.7 to 11.3 patches per 100 ha in 1985 as water levels rose to record high levels, then decreased to 6.7 patches per 100 ha in 1999 as water levels declined. The changes in NP and PD related to water level fluctuations. Both metrics increased during periods of rising water levels and decreased during periods of declining water levels.

It is interesting to note that the number of patches had more than doubled between 1945 and 1999. The lake levels for these two years were almost identical; lake levels were 174.19 m asl in 1945 and 174.11 m

Table 5-1: Landscape Metrics for Long Point, 1945-1999

Landscape Metric	Year / *Mean Water Level*	1945 *174.19*	1955 *174.35*	1964 *173.61*	1978 *174.37*	1985 *174.73*	1995 *174.29*	1999 *174.11*	Trend *w/ Declining Levels*
Area Metrics	**Acronym**								
Total Area (ha)	TA	31136.850	31136.870	31136.992	31136.847	31136.862	31137.018	31136.020	-
Largest Patch Index (%)	LPI	55.371	55.601	55.172	56.027	56.396	56.317	56.138	Decreases
Patch Density, Patch Size and Variability Metrics									
Number of Patches (#)	NP	977	1584	1436	2898	3528	2153	2099	Decreases
Patch Density (#/100 ha)	PD	3.138	5.087	4.612	9.307	11.331	6.915	6.741	Decreases
Mean Patch Size (ha)	MPS	31.870	19.657	21.683	10.744	8.826	14.462	14.834	Increases
Patch Size Standard Deviation (ha)	PSSD	582.912	456.046	479.872	345.272	316.006	398.335	401.963	Increases
Patch Size Coefficient of Variation (%)	PSCV	1829.030	2320.018	2213.125	3213.626	3580.399	2754.356	2709.741	Decreases
Edge Metrics (No Weights)									
Total Edge Length (m)	TE	1049725.804	1283725.702	1283705.281	1593849.028	1604015.959	1492615.096	1385531.266	Decreases
Edge Density (m/ha)	ED	33.713	41.228	41.228	51.188	51.515	47.937	44.499	Decreases
Nearest-Neighbour Metrics *									
Mean Nearest-Neighbour Distance (m)	MNN	136.70	101.60	100.30	66.00	61.40	69.80	90.30	Increases
Mean Proximity Index	MPI	8349.13	3437.11	6550.26	4777.30	5129.95	5124.58	5860.57	Decreases
Shape Metrics									
Landscape Shape Index	LSI	16.782	20.522	20.522	25.480	25.643	23.862	22.150	Decreases
Mean Shape Index	MSI	2.052	1.947	1.991	1.869	1.839	2.015	1.868	Slightly Increases
Area-Weighted Mean Shape Index	AWMSI	5.299	4.840	5.694	6.039	6.087	5.540	5.456	Decreases
Mean Patch Fractal Dimension	MPFD	1.398	1.396	1.402	1.424	1.443	1.427	1.412	Decreases
Area-Weighted Mean Patch Fractal Dimension	AWMPFD	1.259	1.222	1.246	1.209	1.225	1.225	1.223	-
Diversity Metrics									
Shannon's Diversity Index	SHDI	1.372	1.419	1.369	1.348	1.290	1.335	1.339	Decreases from 1945 and 1964; Increases from 1964 to 1999
Simpson's Diversity Index	SIDI	0.641	0.642	0.640	0.624	0.613	0.626	0.628	
Modified Simpson's Diversity Index	MSIDI	1.024	1.026	1.021	0.979	0.949	0.983	0.990	
Shannon's Evenness Index	SHEI	0.596	0.616	0.594	0.586	0.560	0.580	0.582	
Simpson's Evenness Index	SIEI	0.712	0.713	0.711	0.694	0.681	0.695	0.698	
Modified Simpson's Evenness Index	MSIEI	0.445	0.446	0.444	0.425	0.412	0.427	0.430	
Contagion and Interspersion Metrics *									
Interspersion and Juxtaposition Index (%)	IJI	71.24	77.96	71.05	79.37	80.01	72.86	71.59	Decreases

* Derived from Patch Analyst in ArcView using the raster data (output was computed to two decimals); all other metrics were derived in FRAGSTATS*ARC using the vector data

asl in 1999. Some of the difference in NP is due to the difference in spatial resolution of the original aerial photography. The 1945 photographs were captured and interpreted at a smaller scale than the 1999 photos. The remaining differences are likely due to a combination of natural and anthropogenic factors including the pattern of water level fluctuations leading up to these years and land use changes and disturbances since 1945.

MPS changed over time and in relation to water level fluctuations. Between 1945 and 1955, as water levels rose, MPS decreased from 31.9 ha to 19.7 ha, a change of 12.2 ha. MPS increased slightly by 2.0 ha from 1955 to 1964, as water levels declined to record low levels. Between 1964 and 1978, MPS decreased by 10.9 ha to 10.7 ha as water levels rose. MPS experienced a slight decrease of 1.9 ha from 1978 to 1985 as water levels continued to rise to record high levels. Between 1985 and 1995, as water levels declined, MPS increased by 5.6 ha to 14.5 ha. MPS increased slightly to 14.8 ha by 1999 as water levels continued to decline. MPS negatively responded to changing water levels. MPS decreased during periods of rising water levels and increased during periods of declining water levels.

There were notable changes in PSCV over time and in relation to water level fluctuations. From 1945 to 1955 as water levels rose, PSCV increased from 1829.0 % to 2320.0 %; a difference of 491.0 %. Between 1955 and 1964, a period of declining water levels, PSCV decreased by 106.9 %. PSCV increased substantially by 1000.5 % between 1964 and 1978 as water levels rose and increased another 366.8 % from 1978 to 1985 as water levels continued to rise. From 1985 to 1995 and 1995 to 1999, periods of declining water levels, PSCV declined by 826.0 % and 44.6 % respectively. PSCV increased during periods of rising water levels and decreased during periods of declining water levels, thus indicating a positive relationship to fluctuating water levels.

5.1.2.3 Edge Metrics

Changes in TE and ED were evident over time. Between 1945 and 1955 as water levels rose, TE and ED increased by 234 km and 7.5 m/ha, respectively. Between 1955 and 1964 as water levels declined, TE decreased by 0.2 km but ED did not experience a change. ED was calculated using TE and TA of the landscape; since these values are nearly identical for 1955 and 1964 and the ED is only reported to three decimals, no change was reported. TE and ED increased noticeably between 1964 and 1985 (320 km, 10.3 m/ha) as water level rose, but most of the change occurred between 1964 and 1978, the period which experienced a more substantial rise in water levels. From 1985 to 1999, as water levels declined, TE and ED decreased by a total of 218 km and 7.0 m/ha, respectively. Overall, TE and ED increased during periods of rising water levels and decreased during periods of declining water levels.

5.1.2.4 Nearest-Neighbour Metrics

There were evident changes in nearest-neighbour metrics over time. In 1945, MNN and MPI were 136.7 m and 8349.1, respectively. Between 1945 and 1955, a period of rising water levels, MNN decreased by 35.1

m and the value of the MPI decreased by 4912.0. From 1955 to 1964, as water levels declined, there was a marginal decrease in MNN, but the MPI value increased substantially by 3113.1. Between 1964 and 1978, as water levels rose, MNN and MPI experienced decreases of 34.3 m and 1773.0, respectively. MNN decreased another 4.6 m between 1978 and 1985 as water levels continued to rise, but MPI increased in value by 352.6 during the same period. From 1985 to 19995, as water levels declined, MNN increased by 8.4 m and MPI decreased marginally by 5.4. Between 1995 and 1999, as water levels continued to decline, MNN increased another 20.5 m and MPI increased 736.0. It is interesting to note that MNN decreased over time from 1945 to 1985, then increased in value between 1985 and 1999. Some of this change over time can be attributed to the increasing spatial scale of the aerial photography and wetland maps from 1945 to 1978.

In response to water level conditions, MNN generally increased during periods of declining water levels and decreased during periods of rising water levels. There was a discrepancy between 1955 and 1964 when the value actually decreased with declining water levels, but the amount of decrease was marginal compared to the changes observed between the other years. There is also a general negative relationship between MPI and rising water levels. The value of MPI increases as water levels decline and decreases as water levels rise. The opposite trend occurs between 1978-1985 and 1985-1995. As noted earlier, however, the validity of MPI in measuring structural complexity at the landscape scale is questionable.

5.1.2.5 Shape Metrics

There were small changes in the LSI over time. Between 1945 and 1955, a period of rising water levels, LSI increased by 3.7. No change was reported between 1955 and 1964; LSI is based on total edge and area of the landscape, as noted earlier these values were almost identical for these years, and thus the LSI values are identical. Between 1964 and 1978, a period of declining water levels, LSI value increased by 5.0. From 1978 to 1985 water levels continued to rise and LSI experienced a marginal increase in value of 0.2. Then between 1985-1995 and 1995-1999 as water levels declined, LSI decreased by 1.8 and 1.7 respectively. Generally, the changes in LSI over time correspond positively to fluctuating water levels. LSI decreases as water levels decline and increased as water levels rise.

There was little change observed in MSI over time. The value of MSI ranged from 1.8 in 1985, the year with the highest water level, to a value of 2.1 in 1945, the year with the poorest quality and smallest scale of the original aerial photography. There is a general trend in MSI with fluctuating water levels, the value of MSI slightly increased during periods of declining water levels and decreased during periods of rising water levels except between 1995 and 1999 where the value actually decreased with declining water levels. The change in MSI over time was only 0.2 and when considering that the upper limit of MSI is infinite, this amount of change may be meaningless. AWMSI cannot be appropriately applied to this dataset, but the index was examined to provide further insight into the results of the MSI. Although AWMSI did experience small changes over time, there were no clear relationships with water level fluctuations.

Marginal changes in MPFD were also observed over time. Between 1945 and 1955 as water levels rose, MPFD decreased slightly. Between 1955 and 1964 as water levels declined, MPFD increased. From 1964 to 1985, as water levels rose, MPFD increased slightly and then as water levels declined from 1985 to 1999, MPFD decreased. Subsequent to 1964, MPFD increased during periods of rising water levels and decreased during periods of declining water levels. Prior to 1964, this relationship was reversed. The poorer quality and smaller scale of the earlier data inherently had some effect on MPFD. Minor changes in AWMPFD were evident over time, but no discernable trends were apparent in relation to water level fluctuations.

5.1.2.6 Diversity Metrics

There were small changes in all three diversity indices (SHDI, SIDI and MSIDI) and evenness indices (SHEI, SIEI, MSIEI) over time. Between 1945 and 1955 as water levels rose, the indices values slightly increased. Between 1955 and 1964 as water levels declined, the indices values slightly decreased. From 1964 to 1978 and 1978 to 1985, periods of rising water levels, the diversity indices experienced marginal decreases in value. Then from 1985 to 1995 and 1995 to 1999, periods of declining water levels, the diversity indices experienced marginal increases in value. There were no clear relationship between the diversity and evenness indices and water level fluctuations over the entire period of study, but two distinct trends were evident. Between 1945 and 1964, the indices related positively with increasing water levels. The diversity and evenness indices increased as water levels rose and decreased as water levels declined. Conversely, the indices responded negatively to increasing water levels from 1964 to 1999. The change in response was most likely caused by differences in the quality and scale of the data between the two time periods. Little can be concluded from the results over the entire period of study, but some meaning can be inferred from the results post 1964, where the diversity and evenness indices negatively responded to fluctuations in water levels.

5.1.2.7 Contagion and Interspersion Metrics

There were noticeable changes in IJI over time. Between 1945 and 1955, a period of rising water levels, IJI increased by 6.7 %. Between 1955 and 1964, a period of declining water levels, IJI decreased by 6.9 %. From 1964 to 1978 as water levels rose, IJI increased 8.3 %; IJI increased a further 0.6 % as water levels continued to rise between 1978 and 1985. Between 1985 and 1999 as water levels declined, IJI decreased a total of 8.4 %; most of this decrease (7.1 %) occurred between 1985 and 1995, the period that experienced a more substantial decline in water levels. IJI responded positively to fluctuating water levels; as water levels rose, the value of IJI increased, and as water levels declined, the value of IJI decreased.

5.1.2.8 Summary

The landscape analysis of Long Point indicates that the wetland has experienced tremendous change between 1945 and 1999. Many of the changes in the composition and structure of the wetland relate to periods of declining and rising water level conditions. A summary of the key findings is provided in Table 5-2.

Table 5-2: Key Findings of the Landscape Analysis

Response of Landscape		
To Declining Water Levels	**To Rising Water Levels**	**Metric Indicated By**
• Lake surface decreases as wetland vegetation and upland areas along the shoreline expand into newly exposed areas of the lake	• Lake surface increases and inundates low lying wetland and upland areas along the shoreline	• LPI
• Less fragmentation in the wetland as drier wetland vegetation communities expand forming solid and continuous stands of vegetation	• Greater fragmentation within the wetland as wetland vegetation communities become interspersed with water	• NP, PD, MPS, TE, ED
• Less variability among patches sizes within the wetland • Larger continuous patches of wetland vegetation communities overall	• Greater variability among patch sizes within the wetland • Drier vegetation interspersed and fragmented with water • Lake and open water actually increase in area and upland may experience no change	• PSCV
• Distribution of patches of similar types becomes more isolated • Less fragmentation as vegetation communities form continuous patches, thus there are fewer larger patches in the wetland that are further away from each other	• Distribution of patches becomes less isolated • Greater fragmentation within the wetland as larger patches are interspersed with water creating more smaller patches closer to each other	• MNN, MPI
• Shape of the wetland becomes less complex • Patch shapes are simpler due to less fragmentation	• Shape of the wetland becomes more complex • Patch shapes are more complex and convoluted due to greater fragmentation	• LSI, MPFD
• Proportion, distribution, and abundance of area between different patch types becomes more even • Greater diversity as there is less dominance of a single wetland community and a greater number of patch types within the wetland	• Proportion, distribution, and abundance of area between different patch types becomes more uneven • Less diversity as the wetland is dominated by a single wetland community (lake or open water)	• SHDI, SIDI, MSDI • SHEI, SIEI, MSEI • Trend evident from 1964 to 1999; opposite trend occurs prior to 1964
• Interspersion of patch types in the wetland decreases	• Greater interspersion between patch types in the wetland • Distribution of the patch type adjacency becomes more proportionate	• IJI

5.1.3 Class Metrics

Many of the metrics computed at the class level responded similarly to the landscape metrics over time; describing these metrics in detail would be somewhat redundant. However, there were several important observations in the class metrics over time and in response to fluctuating water levels depending on the type of wetland vegetation community. These trends are described below. The class metrics for the individual wetland vegetation communities are provided in Appendix F.

5.1.3.1 Area Metrics

There were notable trends in the areas of the individual wetland communities over time (Figure 5-1). CA of the open water community increased from 1945 to 1985, then the community area declined. The area of tall dense dry emergent vegetation in the wetland decreased from 1945 to 1978, and then the community

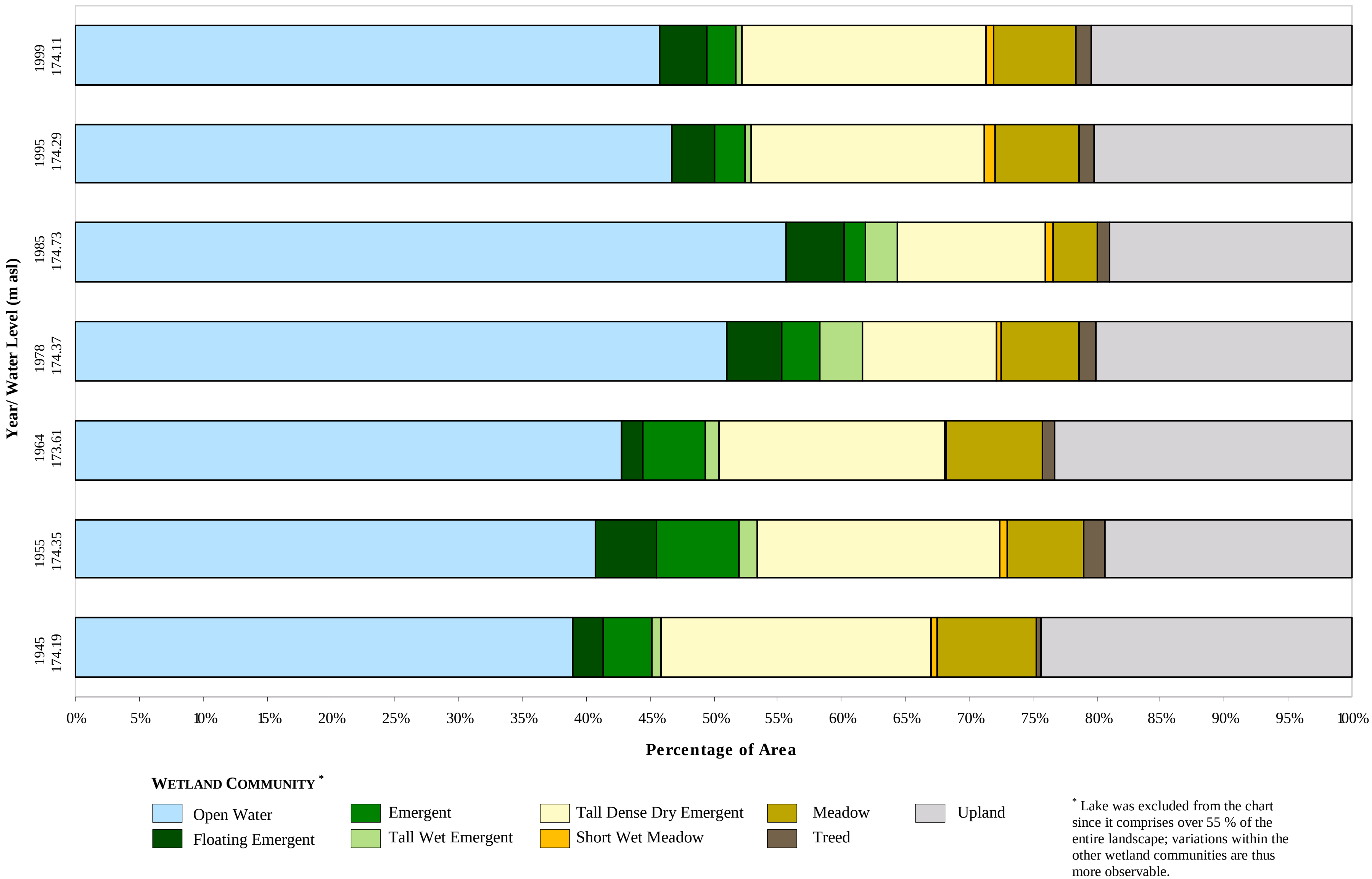


FIGURE 5-1: DISTRIBUTION OF WETLAND VEGETATION COMMUNITIES IN LONG POINT, 1945-1999

experienced an increase in area from 1978 to 1999. The treed wetland community varied over time with little relation to water level fluctuations. The area of treed vegetation was particularly high in 1955 and 1978 (during periods of rising water levels) and in 1995 (as water levels declined). The change is treed vegetation is likely a result of land use patterns in the wetland. There could have been more tree-planting programs before these years, or there could have been less clear cutting for farmland and as a result the vegetation community had the opportunity to expand and grow.

In addition to the trends noted above, there were observed changes in the areas of several other wetland communities in relation to water level fluctuations (Figure 5-2). CA for the lake, floating emergent, tall wet emergent, and short wet meadow vegetation communities generally increased between periods of rising water levels from 1945 to 1955 and from 1964 to 1985. Conversely, CA for these communities decreased between periods of declining water levels from 1955 to 1964 and 1985 to 1999. A few discrepancies in these trends were noted. The CA for floating emergent and short wet meadow vegetation increased from 1995 to 1999 as water levels declined and tall wet emergent declined from 1978 to 1985 as water levels continued to rise. Changes in the CA meadow vegetation and upland communities clearly related to water level fluctuations. CA of these communities increased during periods of declining water levels, and decreased during periods of rising water levels.

The PLAND statistic revealed the dominant wetland communities within Long Point. The lake community was most dominant community in the landscape throughout the period of study. The lake community comprised from 55.2 % of the total landscape in 1964 (the lowest water level period) to 56.4 % in 1985 (the highest water level period). Open water is the second most dominant community, the percentage of open water in the landscape ranged from 17.4 % in 1945 to 24.3 % in 1985. Upland is the third most dominant community in the landscape comprising of 8.3 % on the landscape in 1985 to 10.9 % in 1945. The most dominant vegetation community in the wetland each year was tall dense dry emergent vegetation, which ranged from 4.6 % in 1978 to 9.4 % in 1945. Meadow was second most dominant vegetation community ranging from 1.5 % in 1985 to 3.5 % in 1945. PLAND for the wetland communities responded similarly to the CA for the communities over time and in relation to water level fluctuations.

The LPI of the wetland vegetation communities varied over time, but there were two clear trends in the LPI of several communities in relation to water level fluctuations. The LPI in the lake community increased during periods of rising water levels from 1945 to 1955 and from 1964 to 1985 and decreased during periods of declining water levels from 1955 to 1964 and 1985 to 1999. An inverse relationship occurred in the LPI of the meadow vegetation and upland communities to water level fluctuations as the LPI decreased during periods of rising water levels, and increased during periods of declining water levels.

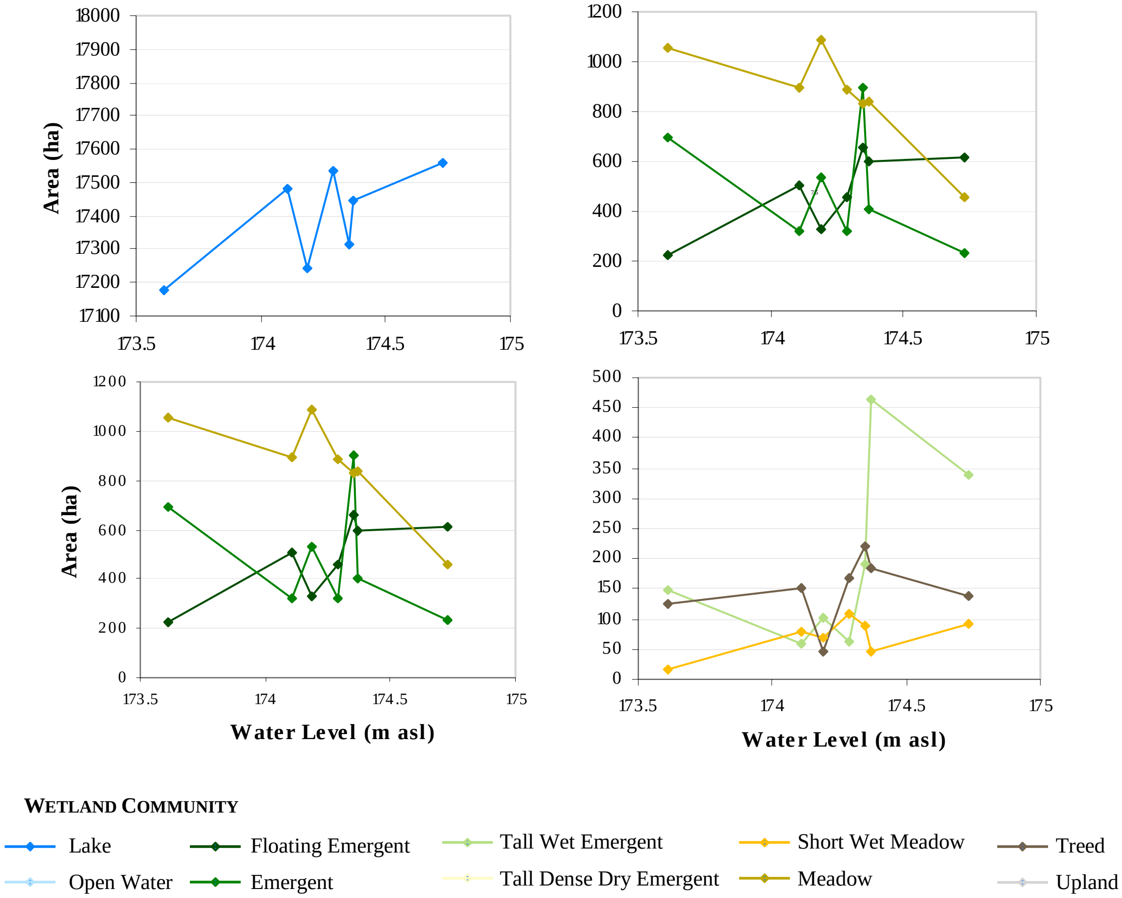


FIGURE 5-2: RELATIONSHIP BETWEEN CLASS AREA AND WATER LEVELS

5.1.3.2 PATCH DENSITY, PATCH SIZE AND VARIABILITY METRICS

Changes in the patch density, patch size, and variability metrics for the individual wetland vegetation communities were evident over time and in relation to water levels. NP, PD, and PSCV for the open water community were higher during periods of extreme low (1964) and high (1985) water levels periods. Conversely, the MPS of the open water community was notably smaller during these low and high water level years than periods of medium water level conditions. It is also interesting to note, that there was quite a large range in MPS for the emergent vegetation community during periods of rising water levels. From 1945 to 1955, the MPS for this community was 9.9 ha; from 1964 to 1978 and 1978 to 1985 the MPS decreased to 2.8 and 1.1 ha respectively. During periods of medium or declining water level conditions, the MPS ranged from 4.1 ha (1999) to 6.0 ha (1964).

NP and PD for the floating emergent, emergent, tall dense dry emergent, short wet meadow, treed, and upland communities generally increased from 1945 to 1955 and 1964 to 1985 as water levels rose, and decreased from 1955 to 1964 and 1985 to 1999 as water levels declined. There were several small discrepancies in the trend between 1995 and 1999 that could be explained by the differences in interpretation of the original aerial photographs. First, the NP and PD for the emergent and tall dense dry emergent vegetation communities increased from 1995 to 1999 as water levels declined. Second, the NP and PD of treed vegetation reached a maximum in 1978 then declined as water levels continued to rise. The other notable trend in NP and PD occurred over time for the tall wet emergent and meadow vegetation communities, where the values increased from 1945 to 1985 then decreased from 1985 to 1999.

MPS for the floating emergent, meadow, treed vegetation, and upland communities generally decreased during periods of rising water levels (1945-1955, 1964-1985), and increased during periods of declining water levels (1955-1964, 1985-1999). MPS of tall wet emergent vegetation decreased from 1945 until 1985; between 1985 and 1995 MPS increased and then the value decreased from 1995 to 1999. The MPS of the lake community increased during periods of rising water levels and decreased during periods of declining water levels.

PSCV for the floating emergent, tall wet emergent, tall dense dry emergent, and short wet meadow vegetation communities generally increased as water levels rose (1945-1955, 1964-1985) and decreased as water levels declined (1955-1964, 1985-1999). But there was a peak in the PSCV for tall dense dry emergent vegetation in 1978. PSCV for emergent vegetation generally decreased over time, though the community did experience an increase in PSCV from 1945 to 1955.

5.1.3.3 Edge Metrics

There were notable changes in the edge metrics of the wetland vegetation communities over time. The amount of TE and ED for open water increased until 1985 when both metrics began to decrease in value. The metrics also increased for the meadow community until 1978 before the values started to decrease in value. For the emergent vegetation community, TE increased between 1945 and 1955 then continually declined from 1955 to 1999 while ED decreased from 1945 to 1978 and then increased from 1978 to 1999. It is uncertain why these relationships occur over time, but land use changes and other processes in the wetland must influence the relationships in some manner.

There were several trends observed in the edge metrics of the wetland vegetation communities in relation to water level fluctuations. TE and ED for the floating emergent and short wet meadow vegetation communities increased with rising water levels (1945-1955, 1964-1985) and decreased with declining water levels (1955-1964, 1985-1995), but there was a small increase between 1995 and 1999 as water levels declined for both metrics. This general relationship also applied to the tall wet emergent and treed vegetation communities, although these communities did experience an earlier peak in 1978, before TE and ED decreased between 1978 and 1985 as water levels continued to rise. This trend is likely a result of the smaller amount of water level rise compared to the rise between 1964 and 1978, and the fact that the communities also had time to adapt to the flooding conditions, hence the values of TE and ED decreased.

Although the edge metrics for lake, open water, and tall dense dry emergent vegetation varied over time, there were no evident trends in these communities in relation to water level fluctuations.

5.1.3.4 Nearest-Neighbour Metrics

Changes in the nearest-neighbour metrics were evident at the class level over time, but few trends were observed in relation to water level fluctuations. For floating emergent and treed vegetation, MNN decreased during periods of rising water levels from 1945 to 1955 and 1964 to 1985, and increased during periods of declining water levels from 1955 to 1964 and 1985 to 1999. This general relationship also applied to emergent and tall wet emergent vegetation though there were periods of discrepancies. Between 1945 and 1955, the MNN for the emergent community increased substantially as water levels rose and between 1978 and 1985, the tall wet emergent community experienced a slight increase in MNN as water levels continued to rise. There were no clear trends in the MPI values at the class level.

5.1.3.5 Shape Metrics

The shape metrics for the wetland communities vary over time and in response to water level conditions. Similar to the LSI of the entire wetland complex, the LSI values of all the wetland communities, except for lake, positively responded to fluctuating water levels. Generally, the LSI values increased between 1945 and 1955 and from 1964 to 1985, periods of rising water levels, and decreased between 1955 and 1964 and from 1985 to 1999, periods of declining water levels. There were noted discrepancies in the LSI for open water, emergent vegetation, and upland in 1964, where the values increased as water levels declined. LSI for tall wet emergent and meadow vegetation also increased between 1955 and 1964 as water levels declined and decreased between 1978 and 1985 as water levels continued to rise. In addition, the LSI for tall dense dry emergent vegetation increased between 1985 and 1995 as water levels declined. The discrepancies of these communities likely relate to the magnitude of water level fluctuation and how quickly these vegetation communities respond and adapt to the change.

Similarly to the MSI values at the landscape level, no clear relationship with water level fluctuations was evident at the class level for all wetland communities. However, there was a general response of the MSI to water level fluctuations for the tall wet emergent, tall dense dry emergent, and meadow vegetation communities. The MSI values of these communities generally increased during periods of declining water levels (1955-1964, 1985-1995) and decreased during periods of rising water levels (1945-1955, 1964-1978, 1978-1985). However, the MSI values between 1995 and 1999 decreased as water levels decline. This discrepancy may be due in part to the differences in the interpretation of the aerial photography for 1999 compared to the earlier years. The AWMSI responded identically to the MSI values over time.

There were observed changes in the values of MPFD over time for several wetland communities. The MPFD values for open water, tall wet emergent, and treed vegetation communities increased from 1945 to 1985, then decreased from 1985 to 1999. The same trend was evident in emergent vegetation over time, although there was a slight decrease in 1955. MPFD for the floating emergent vegetation community generally responded with water level fluctuations. The values decreased as water levels declined (1955-1964, 1985-1995, and 1995-1999) and increased as water levels rose (1964-1978, 1978-1985). However, there was a small decline in value of 0.1 from 1945 to 1955 as water levels rose. The MPFD for the meadow community responded similarly to the landscape value. Between 1945 and 1955, MPFD decreased as water levels rose, and increased between 1955 and 1964 as water levels declined. After 1964, this relationship reverses, and the MPFD increased as water levels rose between 1964 and 1985 then decreased as water levels declined between 1985 and 1999. It is difficult to draw any conclusions from these relationships.

5.1.3.6 Contagion and Interspersion Metrics

The majority of the interspersion and juxtaposition index values for the wetland communities responded similarly to the landscape over time and in response to water level fluctuations. During periods of rising water levels, the IJI of the lake, open water, tall dense dry emergent vegetation, meadow, treed vegetation, and upland communities increased. During periods of declining water levels, the IJI of these communities decreased. IJI values for the emergent and floating emergent vegetation communities generally responded the same, but there were periods of discrepancy.

Between 1955 and 1964, as water levels declined, the IJI value of emergent vegetation increased. The greater interspersion during this period could be an indication of how quickly the vegetation responds to declining water level conditions. Emergent vegetation likely germinated from seed banks that were exposed on mudflats in the wetland as water levels receded. Therefore, there was greater interspersion within this community as the adjacency of the patches of the community become more proportionate. Between 1978 and 1985, as water levels continued to rise, the IJI value of floating emergent vegetation decreased. The floating emergent vegetation community reached maximum interspersion in 1978 due to the substantial rise in water levels between 1964 and 1978. The adjacency of the patches in the community were more proportionate, but then as water levels rose slightly from 1978 to 1985, the community was able to adapt and expand in area to the higher water level conditions, and as a result the interspersion of the community decreased and the adjacency of the patches became more disproportionate.

The final observation with the IJI metric relates to the short wet meadow vegetation community. IJI of the short wet meadow vegetation community varied over time, but there did not appear to be a clear trend with fluctuating water levels. The IJI value increased between 1945 and 1955 as water levels rose, then the value decreased between 1955 and 1964 as water levels declined. The IJI value increased from 1964 to 1978 as water levels rose and then decreased as water levels continued to rise further between 1978 and 1985. From 1985 to 1999, the IJI value increased, as water levels declined. Variations in the response of the IJI value for short wet meadow vegetation are clearly related to the total area of the community in the wetland. There were substantial changes in the area of this community over the period of study; for years that had less interspersion (1955, 1985, and 1995) the total area of the community was noticeably larger compared to other years. Like tall wet emergent vegetation, some of this variation can also be explained by the response time of the vegetation to water level fluctuations.

5.1.3.7 Summary

The structure of the wetland communities at Long Point has considerably changed throughout the period of study. A summary of the key trends in relation to declining and rising water level conditions are provided in Table 5-3.

Table 5-3: Key Findings of Class Analysis

Response of Classes		
To Declining Water Levels	**To Rising Water Levels**	**Metric Indicated By**
• Increased areas of meadow and upland communities • Meadow expands lakeward during drier conditions • Upland areas along the shoreline become exposed as the lake recedes	• Increased areas of lake and wetter emergent and meadow communities • Lake water inundates low lying areas along the shore and drier vegetation communities in the wetland, thus the areas of tall wet emergent and short wet meadow increases • Floating emergent vegetation expands	• CA, PLAND
• Lake surface decreases as meadow vegetation and upland areas along the shoreline expand into newly exposed areas of the lake	• Lake surface increases and inundates low lying wetland and upland areas along the shoreline	• LPI
• Less fragmentation within the vegetated communities as the communities expand forming solid and continuous stands of floating emergent, emergent, tall dense dry emergent, short wet meadow, and treed vegetation • Less fragmentation in the upland community • Greater fragmentation of open water in extreme low water level conditions	• Greater fragmentation within vegetated communities as the communities become interspersed with water • Greater fragmentation of open water in extreme high water level conditions	• NP, PD, MPS, TE, ED
• Less variability among patches sizes within the wetter emergent and meadow communities • Larger continuous patches of floating emergent, tall wet emergent, tall dense dry emergent, and short wet meadow	• Greater variability among patch sizes within the wetland • Wetter emergent and meadow vegetation communities are interspersed and fragmented with water	• PSCV
• Distribution of patches becomes more isolated for the floating emergent, emergent, tall wet emergent, and treed communities • Less fragmentation in these communities as the vegetation forms continuous patches thus there are fewer larger patches in the wetland that are further away from each other	• Distribution of patches within becomes less isolated • Greater fragmentation within the wetland as larger patches are interspersed with water creating more smaller patches closer to each other	• MNN
• Complexity of the communities decrease • Patch shapes within each community are generally simpler due to less fragmentation	• Complexity of the communities increase • Patch shapes within the communities are more complex and convoluted due to greater fragmentation and interspersion	• LSI, MPFD
• Mean complexity of the patch shapes for tall wet emergent increase as the vegetation becomes more fragmented as water levels decline; community is taken over by drier vegetation • Complexity of tall dense dry emergent and meadow communities increase as these communities colonize areas exposed by receding water levels • Complexity of floating emergent decreases	• Mean complexity of the patch shapes for the floating emergent community increases • Complexity in the tall wet emergent as the vegetation community expands • Complexity in the tall dense dry emergent and meadow communities decrease as the community range contracts	• MSI

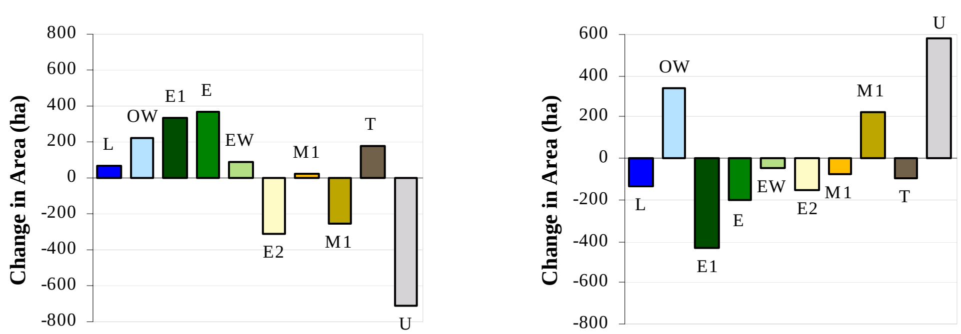
Response of Classes		
To Declining Water Levels	To Rising Water Levels	Metric Indicated By
• Interspersion of patch types decreases for most communities	• Greater interspersion between patch types and the distribution of the patch type adjacency becomes more proportionate for most communities • Patches of tall wet emergent and short wet meadow may actually increase as the drier emergent and meadow communities are flooded	• IJI

5.1.4 Direction and Magnitude of Change

The class areas of the wetland vegetation communities were further analyzed to determine the direction and magnitude of change. The area analysis indicated that substantial changes have occurred within the wetland communities between the periods of study. Notable changes are highlighted for each period. Refer to Tables 5-4 through 5-6 for the change in area and percent change statistics for each wetland community for the different time periods.

5.1.4.1 1945 to 1955: Rising Water Levels

Between 1945 and 1955, water levels in Lake Erie increased by 0.16 m. Although this increase was small, there were noticeable changes in the wetland communities to wetter and more water tolerant vegetation in the wetland continuum (Figure 5-3a). By 1955, the area of floating emergent vegetation increased by 330.0 ha (100.1 %), tall wet emergent vegetation by 90.9 ha (90.2 %), emergent vegetation by 366.5 ha (68.8 %), and short wet meadow by 20.3 ha (29.4 %). The greatest decrease in area occurred within the meadow (253.2 ha, 23.3 %) and upland communities (708.9 ha, 20.9 %). Tall dense dry emergent vegetation also experienced a decline in area by 308.5 ha, but because this community is one of the dominant vegetation communities in the wetland, the loss only represented 10.5 % of the community's total area.



a) 1945-1955: Rising **b) 1955-1964: Declining**

Figure 5-3: Change in Wetland Community Area, 1945-1964

5.1.4.2 1955 to 1964: Declining Water Levels

From 1955 to 1964, water levels declined by 0.74 m to one of the lowest water level periods on record for Lake Erie. Consequently, the wetland vegetation tended to drier communities (Figure 5-3b). The areas of the meadow and upland communities increased by 219.1 ha (26.3 %) and 578.0 ha (21.6 %) respectively. By 1964, the areas of wetter vegetation communities had been severely reduced. The area of short wet

TABLE 5-4: WETLAND COMMUNITY AREA IN LONG POINT, 1945-1999*

WETLAND COMMUNITY	AREA (ha)						
	1945	1955	1964	1978	1985	1995	1999
Lake	17240.770	17312.443	17178.920	17445.130	17560.090	17535.280	17479.000
Open Water	5415.527	5632.808	5973.606	6977.494	7558.565	6356.960	6241.567
Floating Emergent	329.588	659.635	226.268	599.609	614.517	456.379	507.117
Emergent	532.718	899.181	694.627	405.126	231.935	319.192	319.997
Tall Wet Emergent	100.792	191.709	148.552	463.796	337.704	63.578	58.071
Tall Dense Dry Emergent	2927.794	2619.266	2461.872	1431.805	1568.040	2492.748	2612.112
Short Wet Meadow	68.968	89.221	17.260	46.960	91.573	109.592	77.717
Meadow	1086.254	833.095	1052.151	840.401	458.934	887.075	893.418
Treed	45.374	219.298	125.568	184.286	137.638	167.929	151.079
Upland	3389.065	2680.214	3258.168	2742.240	2577.866	2748.285	2795.942

* Areas were derived through the FRAGSTATS analysis

TABLE 5-5: WETLAND VEGETATION CHANGE IN LONG POINT, 1945-1999

WETLAND COMMUNITY	CHANGE IN AREA (ha)					
	1945-1955	1955-1964	1964-1978	1978-1985	1985-1995	1995-1999
Lake	71.673	-133.523	266.210	114.960	-24.810	-56.280
Open Water	217.281	340.798	1003.888	581.071	-1201.605	-115.393
Floating Emergent	330.047	-433.367	373.341	14.908	-158.138	50.738
Emergent	366.463	-204.554	-289.501	-173.191	87.257	0.805
Tall Wet Emergent	90.917	-43.157	315.244	-126.092	-274.126	-5.507
Tall Dense Dry Emergent	-308.528	-157.394	-1030.067	136.235	924.708	119.364
Short Wet Meadow	20.253	-71.961	29.700	44.613	18.019	-31.875
Meadow	-253.159	219.056	-211.750	-381.467	428.141	6.343
Treed	173.924	-93.730	58.718	-46.648	30.291	-16.850
Upland	-708.851	577.954	-515.928	-164.374	170.419	47.657

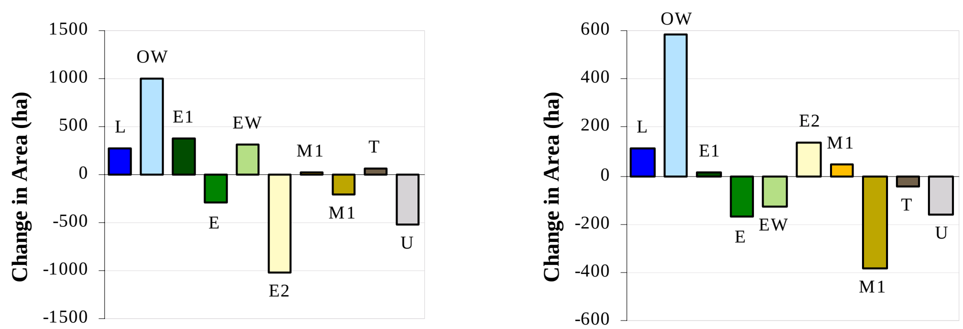
TABLE 5-6: PERCENT CHANGE IN WETLAND VEGETATION AT LONG POINT, 1945-1999

WETLAND COMMUNITY	PERCENT CHANGE (%)					
	1945-1955	1955-1964	1964-1978	1978-1985	1985-1995	1995-1999
Lake	0.42	-0.77	1.55	0.66	-0.14	-0.32
Open Water	4.01	6.05	16.81	8.33	-15.90	-1.82
Floating Emergent	100.14	-65.70	165.00	2.49	-25.73	11.12
Emergent	68.79	-22.75	-41.68	-42.75	37.62	0.25
Tall Wet Emergent	90.20	-22.51	212.21	-27.19	-81.17	-8.66
Tall Dense Dry Emergent	-10.54	-6.01	-41.84	9.51	58.97	4.79
Short Wet Meadow	29.37	-80.65	172.07	95.00	19.68	-29.09
Meadow	-23.31	26.29	-20.13	-45.39	93.29	0.72
Treed	383.31	-42.74	46.76	-25.31	22.01	-10.03
Upland	-20.92	21.56	-15.83	-5.99	6.61	1.73

meadow decreased by 72.0 ha (80.7 %), floating emergent by 433.4 ha (65.7 %), emergent by 204.6 ha (22.8 %), and tall wet emergent by 45.2 ha (22.5 %). Wetland treed vegetation also experienced a decrease in area of 93.7 ha (42.7 %).

5.1.4.3 1964 TO 1978: RISING WATER LEVELS

There were significant changes in the wetland between 1964 and 1978, as water levels rose by 0.76 m (Figure 5-4a). There were substantial increases in the areas of tall wet emergent (315.2 ha, 212.2 %), short wet meadow (29.7 ha, 172.1 %), and floating emergent vegetation (373.3 ha, 165.0 %), as the corresponding drier wetland vegetation communities were flooded. The area of emergent vegetation decreased by 289.5 ha (41.7 %), tall dense dry emergent vegetation by 1030.1 ha (41.8 %), and meadow by 211.8 ha (20.1 %). During this period, the area of treed vegetation in the wetland increased by 58.7 ha (46.8%).



a) 1964-1978: Rising **b) 1978-1985: Rising**

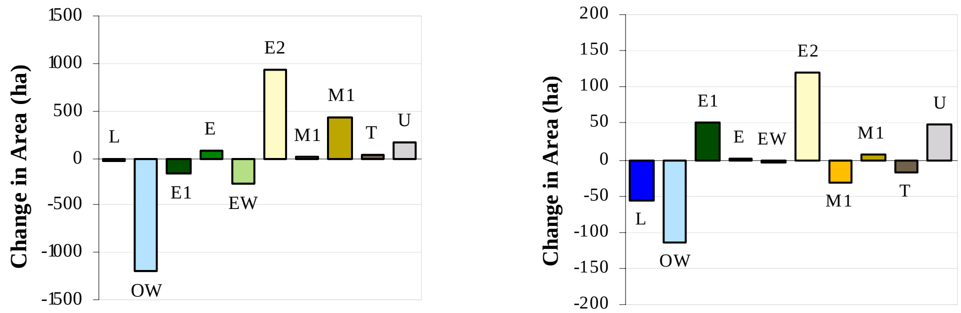
FIGURE 5-4: CHANGE IN WETLAND COMMUNITY AREA, 1964-1985

5.1.4.4 1978 TO 1985: RISING WATER LEVELS

Water levels continued to rise by 0.36 m between 1978 and 1985, on the way to the highest water level on record for Lake Erie. During this period, there were notable changes in several wetland vegetation communities (Figure 5-4b). The only community to experience a significant increase in area was short wet meadow, which increased by 44.6 ha (95.0 %). The amount of the drier vegetation communities in the wetland continued to decrease in response to the rising lake level. The area of meadow decreased by 381.5 ha (45.4 %) and emergent vegetation decreased by 173.2 ha (42.8 %). There were also decreases in the amount of tall wet emergent of 126.1 ha (27.2 %) and treed vegetation of 46.6 ha (25.3 %).

5.1.4.5 1985 TO 1995: DECLINING WATER LEVELS

After the record high water levels of 1986 (174.90 m asl), water levels within Lake Erie declined. Between 1987 and 1995 as lake levels dropped 0.61 m from 174.90 m asl to 174.29 m asl, the wetland vegetation progressed towards drier vegetation community classes (Figure 5-5a). The area of meadow increased by 428.1 ha (93.3 %), tall dense dry emergent by 924.7 ha (59.0 %), emergent by 87.3 ha (37.6 %), and treed by 30.3 ha (22.0 %). There were notable losses in the area of wetter wetland communities. The area of tall wet emergent vegetation decreased by 274.1 ha (81.2 %), floating emergent vegetation by 158.1 ha (25.7 %), and open water by 1201.6 ha (15.9 %).



a) 1985-1995: Declining **b) 1995-1999: Declining**

FIGURE 5-5: CHANGE IN WETLAND COMMUNITY AREA, 1985-1999

5.1.4.6 1995 TO 1999: DECLINING WATER LEVELS

Between 1995 and 1999 as water levels declined a further 0.18 m, the wetland vegetation continued to shift towards the drier vegetation communities but the amount of change was not as dramatic compared to the amount of change between 1985 and 1995 (Figure 5-5b). There are two main reasons for the smaller observed change. First, there was a smaller decline in lake levels during this period. The lake levels dropped 18 cm during this period compared to 44 cm during the previous period. Second, the period of change only spanned a total of five years, whereas the change from 1985 to 1995 occurred over 11 years and the vegetation communities had a longer time to respond and adapt to changes in the water level. The greatest change occurred within the short wet meadow community, which decreased in area by 31.9 ha (29.1 %). There were also small decreases in the area of treed (16.9 ha, 10.0 %) and tall wet emergent vegetation (5.5 ha, 8.7 %). The most notable increases in area occurred within the floating emergent (50.7 ha, 11.1 %) and tall dense dry emergent (119.4 ha, 4.8 %) communities.

5.1.4.7 SUMMARY

There were significant changes within the areas of the wetland vegetation communities at Long Point between 1945 and 1999. Several trends in the direction and magnitude of change were observed in relation to fluctuating water levels (Table 5-7).

TABLE 5-7: KEY FINDINGS OF AREA ANALYSIS

RESPONSE OF CLASSES	
TO DECLINING WATER LEVELS	**TO RISING WATER LEVELS**
• Vegetation tends towards drier communities	• Vegetation tends toward wetter communities
• Significant increases in tall dense dry emergent, and meadow vegetation as these communities expand lakeward and develop on newly exposed areas	• Increases in floating emergent vegetation as the community expands with the higher water
• Decreases in floating emergent, tall wet emergent and short wet meadow vegetation as moisture conditions in these communities decline	• Increases in tall wet emergent and short wet meadow as tall dense dry emergent and meadow communities become interspersed by lake water due to higher water levels

Response of Classes	
To Declining Water Levels	**To Rising Water Levels**
• Lake area declines and upland areas along the shore increase • Upland increases as newly exposed sandy shores along the shoreline remain undeveloped of vegetation	• Lake area expands • Extent of upland decreases as lake water floods areas along the shoreline • Wetter vegetation communities compress the extent of upland communities inlands as the wetter vegetation migrates landward

5.2 Spatial Analysis

Changes within the wetland communities at Long Point were further explored by a spatial analysis that involved overlaying the wetland grids together to determine the type and location of change that occurred between contiguous years of data. The first section of the spatial analysis describes the major inter-community changes that occurred within the wetland vegetation between the periods of study. The second section highlights the location of significant vegetation community change within the wetland complex.

5.2.1 Type of Change

The spatial overlay of the wetland grids provided an indication of the inter-community changes or migration within the wetland vegetation communities between contiguous years of data. The changes in the wetland vegetation between each time period are discussed; change matrices for each period are provided in Tables 5-8 and 5-9. There is a slight discrepancy between the total area of each wetland community in these Tables compared to the total class area previously reported. The discrepancy is due to the different calculation methods used in each analysis; the areas reported in the temporal analysis were calculated from the vector data in FRAGSTATS and the areas reported in the change matrices were derived from the raster data in GRID. There is no consistent trend in the differences between the areas and the maximum difference is around 4 ha, which is less than one percent of the total landscape area.

5.2.1.1 1945 to 1955: Rising Water Levels

Overall, 4061.8 ha (13.0 %) of wetland experienced a change in vegetation community between 1945 and 1955 (Table 5-8a). The most notable change occurred within the tall dense dry emergent vegetation class, which accounted for 30.4 % of the total change. Of the total area of tall dense dry emergent to change, 555.4 ha (44.9 %) converted to emergent, 227.0 ha (18.4 %) to open water, 212.9 ha (17.2 %) to floating emergent, and 137.6 ha (11.1 %) to tall wet emergent vegetation. Upland accounted for 20.9 % of the total change, of which 429.4 ha (50.5 %) converted to meadow, 155.1 ha (18.2 %) to treed, 104.2 ha (12.3 %) to tall dense dry emergent, and 86.3 ha (10.1 %) to lake. The meadow community accounted for 19.4 % of the total change with 562.4 ha (71.2 %) changing to tall dense dry emergent vegetation. Changes in floating emergent and emergent vegetation were also noteworthy accounting for 14.4 % and 13.9 % of the total change respectively. Approximately, 138 ha (32.7 %) of emergent vegetation changed to floating emergent, and 109.1 ha (46.7 %) of floating emergent and 166.6 ha (39.4 %) of emergent vegetation changed to open water. Upland experienced the greatest net loss between 1945 and 1955 as the area

TABLE 5-8: WETLAND VEGETATION CHANGE MATRIX, 1945-1978 (HA)*

a)

	WETLAND COMMUNITY	CHANGE TO 1955: L	OW	E1	E	EW	E2	M1	M	T	U	TOTAL (F)	% OF TOTAL CHANGE
CHANGE FROM 1945	L	-	1.25	0.00	0.17	0.00	0.10	0.00	1.01	0.00	20.81	23.34	0.6
	OW	4.80	-	170.40	74.23	8.81	81.86	3.77	9.65	0.82	11.72	366.06	9.0
	E1	0.42	109.12	-	59.13	1.86	46.60	1.53	5.82	2.16	7.20	233.83	5.8
	E	0.12	166.54	137.97	-	13.03	91.05	1.21	8.08	0.71	3.67	422.37	10.4
	EW	0.00	16.98	6.67	29.76	-	30.43	0.00	0.00	0.00	0.00	83.84	2.1
	E2	0.33	227.04	212.86	555.38	137.58	-	2.06	60.19	2.33	38.23	1236.01	30.4
	M1	0.03	16.73	0.63	1.14	0.20	6.83	-	19.90	0.30	4.36	50.13	1.2
	M	3.43	21.10	25.65	55.56	13.12	562.36	35.84	-	18.91	53.74	789.70	19.4
	T	0.00	0.04	0.01	0.26	0.00	2.12	0.00	2.66	-	0.84	5.93	0.1
	U	86.33	25.78	8.78	14.66	0.43	104.23	25.89	429.35	155.10	-	850.55	20.9
TOTAL (T)		95.44	584.58	562.97	790.29	175.03	925.57	70.30	536.66	180.33	140.57	4061.75	
NET CHANGE (T-F)		72.10	218.52	329.14	367.92	91.20	-310.44	20.17	-253.04	174.40	-709.98		
PERCENT (%) OF TOTAL CHANGE		2.35	14.39	13.86	19.46	4.31	22.79	1.73	13.21	4.44	3.46		
NET GAIN/LOSS %		1.78	5.38	8.10	9.06	2.25	-7.64	0.50	-6.23	4.29	-17.48		

b)

	WETLAND COMMUNITY	CHANGE TO 1964: L	OW	E1	E	EW	E2	M1	M	T	U	TOTAL (F)	% OF TOTAL CHANGE
CHANGE FROM 1955	L	-	3.07	0.00	0.00	0.00	0.46	0.00	0.85	0.00	155.53	159.91	4.4
	OW	0.89	-	90.24	92.58	6.75	97.93	6.98	53.18	4.41	41.43	394.40	10.8
	E1	0.00	355.02	-	87.54	16.70	103.22	0.00	32.34	1.47	8.11	604.40	16.5
	E	0.00	201.70	58.32	-	15.71	294.44	1.05	51.83	0.98	22.02	646.04	17.6
	EW	0.00	17.19	2.56	5.89	-	143.42	0.00	11.56	0.00	0.71	181.34	5.0
	E2	0.06	113.89	18.50	215.74	95.44	-	0.58	355.88	4.19	85.06	889.34	24.3
	M1	0.00	4.90	0.60	0.00	0.00	6.18	-	54.45	0.56	20.56	87.25	2.4
	M	0.09	20.17	1.64	31.22	1.71	64.45	6.25	-	24.88	279.07	429.49	11.7
	T	0.14	4.98	0.27	2.52	0.68	8.12	0.00	30.69	-	92.61	140.01	3.8
	U	25.65	14.24	0.50	5.31	0.75	14.80	0.69	56.62	9.94	-	128.51	3.5
TOTAL (T)		26.83	735.16	172.66	440.80	137.75	733.03	15.55	647.40	46.43	705.10	3660.70	
NET CHANGE (T-F)		-133.08	340.76	-431.74	-205.24	-43.59	-156.31	-71.70	217.90	-93.59	576.59		
PERCENT (%) OF TOTAL CHANGE		0.73	20.08	4.72	12.04	3.76	20.02	0.42	17.69	1.27	19.26		
NET GAIN/LOSS %		-3.64	9.31	-11.79	-5.61	-1.19	-4.27	-1.96	5.95	-2.56	15.75		

c)

	WETLAND COMMUNITY	CHANGE TO 1978: L	OW	E1	E	EW	E2	M1	M	T	U	TOTAL (F)	% OF TOTAL CHANGE
CHANGE FROM 1964	L	-	8.64	0.00	0.00	0.00	0.01	0.00	0.65	0.00	13.41	22.71	0.5
	OW	5.20	-	177.81	41.79	40.38	80.04	0.63	28.34	12.41	46.08	432.68	9.4
	E1	0.00	157.28	-	6.41	8.61	5.93	0.00	1.56	0.03	21.24	201.05	4.4
	E	0.78	291.95	99.58	-	15.13	83.61	0.23	57.83	1.11	51.34	601.55	13.1
	EW	0.00	61.27	13.78	7.07	-	25.08	0.00	3.77	0.04	6.62	117.65	2.6
	E2	7.29	663.97	211.03	149.54	337.74	-	0.43	72.86	5.33	73.22	1521.42	33.1
	M1	0.04	3.99	2.02	2.36	0.01	0.58	-	4.05	0.06	2.65	15.75	0.3
	M	20.16	160.13	56.17	65.91	25.93	214.19	27.26	-	27.79	130.82	728.37	15.9
	T	0.13	10.70	1.35	3.37	0.58	6.90	1.76	29.36	-	19.87	74.02	1.6
	U	255.57	81.78	13.28	33.91	3.44	71.83	15.42	318.36	84.96	-	878.54	19.1
TOTAL (T)		289.17	1439.70	575.02	310.36	431.83	488.16	45.73	516.77	131.73	365.26	4593.73	
NET CHANGE (T-F)		266.46	1007.02	373.97	-291.18	314.18	-1033.26	29.98	-211.59	57.72	-513.29		
PERCENT (%) OF TOTAL CHANGE		0.73	6.29	31.34	12.52	6.76	9.40	10.63	1.00	11.25	2.87	7.95	
NET GAIN/LOSS %		-3.64	5.80	21.92	8.14	-6.34	6.84	-22.49	0.65	-4.61	1.26	-11.17	

* Areas were calculated using the total number of cells corresponding to each type of change
F = decrease in wetland community due to change; T = increase in wetland community due to change
L = Lake; OW = Open Water; E1 = Floating Emergent; E = Emergent; EW = Tall Wet Emergent; E2 = Tall Dense Dry Emergent; M1= Short Wet Meadow; M = Meadow; T = Treed; U = Upland

TABLE 5-9: WETLAND VEGETATION CHANGE MATRIX, 1978-1999 (HA)*

a)

Wetland Community	Change to 1985										Total (F)	% of Total Change
	L	OW	E1	E	EW	E2	M1	M	T	U		
Change from 1978												
L	-	0.76	0.01	0.01	0.00	0.00	0.00	0.14	0.06	25.14	26.14	0.8
OW	22.20	-	151.86	24.03	47.49	112.23	6.15	31.68	7.79	29.72	433.17	13.1
E1	0.19	305.34	-	25.60	24.65	36.35	1.47	6.67	0.92	11.61	412.79	12.4
E	1.86	167.26	77.57	-	16.73	56.22	2.07	16.95	1.01	18.89	358.56	10.8
EW	0.10	129.53	74.76	11.30	-	115.55	0.65	1.96	1.18	3.76	338.79	10.2
E2	0.65	154.66	67.00	49.56	107.15	-	4.36	40.82	10.28	46.37	480.86	14.5
M1	0.03	15.64	0.62	1.08	0.00	5.00	-	2.66	1.41	8.29	34.73	1.0
M	7.40	108.49	35.06	43.53	11.75	212.76	36.17	-	24.98	147.57	627.72	18.9
T	0.49	14.07	4.26	4.31	0.03	16.08	4.72	49.39	-	28.04	121.39	3.7
U	107.84	121.16	15.84	25.42	4.46	64.30	23.17	95.11	26.67	-	483.97	14.6
Total (T)	140.76	1016.90	427.00	184.85	212.27	618.48	78.77	245.39	74.30	319.39	3318.12	
Net Change (T-F)	114.62	583.73	14.21	-173.71	-126.52	137.62	44.04	-382.33	-47.09	-164.58		
Percent (%) of Total Change	4.24	30.65	12.87	5.57	6.40	18.64	2.37	7.40	2.24	9.63		
Net Gain/Loss %	3.45	17.59	0.43	-5.24	-3.81	4.15	1.33	-11.52	-1.42	-4.96		

b)

Wetland Community	Change to 1995										Total (F)	% of Total Change
	L	OW	E1	E	EW	E2	M1	M	T	U		
Change from 1985												
L	-	4.16	0.00	0.00	0.00	0.32	0.00	1.47	0.00	60.26	66.21	2.0
OW	5.92	-	239.46	210.96	34.39	648.94	26.37	128.59	5.44	49.52	1349.58	40.2
E1	0.00	75.43	-	62.09	3.34	282.25	4.80	27.78	1.84	10.32	467.86	13.9
E	0.00	11.85	8.22	-	3.15	113.49	1.27	53.08	2.62	14.93	208.61	6.2
EW	0.00	9.55	2.39	3.47	-	277.14	11.00	22.44	0.30	4.08	330.36	9.8
E2	0.59	30.76	50.36	12.25	15.48	-	15.22	280.21	9.58	49.92	464.37	13.8
M1	0.00	0.71	0.43	2.02	0.00	1.68	-	36.03	2.25	10.02	53.14	1.6
M	1.48	6.49	3.80	3.31	1.15	41.08	6.52	-	62.64	83.81	210.30	6.3
T	0.37	1.28	1.11	0.22	0.00	5.03	0.55	23.76	-	31.56	63.88	1.9
U	33.39	3.50	3.44	2.51	0.13	19.34	5.72	67.68	10.54	-	146.25	4.4
Total (T)	41.76	143.73	309.21	296.83	57.64	1389.27	71.44	641.03	95.21	314.44	3360.56	
Net Change (T-F)	-24.45	-1205.86	-158.64	88.21	-272.72	924.90	18.30	430.73	31.33	168.19		
Percent (%) of Total Change	1.24	4.28	9.20	8.83	1.72	41.34	2.13	19.08	2.83	9.36		
Net Gain/Loss %	-0.73	-35.88	-4.72	2.62	-8.12	27.52	0.54	12.82	0.93	5.00		

c)

Wetland Community	Change to 1999										Total (F)	% of Total Change
	L	OW	E1	E	EW	E2	M1	M	T	U		
Change from 1995												
L	-	1.24	0.06	0.00	0.00	1.63	0.00	1.45	0.09	74.87	79.33	3.8
OW	0.03	-	62.32	56.76	9.26	182.13	3.50	29.78	3.79	33.11	380.68	18.4
E1	0.00	35.84	-	15.25	1.92	89.63	0.91	19.28	1.89	11.66	176.37	8.5
E	0.00	24.57	42.81	-	0.91	36.66	0.46	10.67	1.02	11.68	128.78	6.2
EW	0.00	6.68	2.55	2.16	-	17.90	0.00	2.43	0.04	1.11	32.88	1.6
E2	0.84	131.93	91.35	30.72	12.04	-	1.94	100.76	8.48	75.08	453.14	21.9
M1	0.04	2.25	0.69	0.27	0.33	15.84	-	13.05	1.30	17.11	50.88	2.5
M	3.10	22.44	16.04	11.81	1.53	122.98	6.31	-	16.32	143.47	343.97	16.6
T	0.00	5.52	2.10	0.82	0.20	9.60	0.23	18.62	-	35.15	72.24	3.5
U	18.65	34.60	9.95	11.48	1.09	95.40	6.06	153.26	22.23	-	352.73	17.0
Total (T)	22.65	265.06	227.88	129.27	27.27	571.77	19.41	349.30	55.15	403.23	2070.99	
Net Change (T-F)	-56.68	-115.62	51.51	0.49	-5.60	118.63	-31.46	5.33	-17.09	50.50		
Percent (%) of Total Change	1.09	12.80	11.00	6.24	1.32	27.61	0.94	16.87	2.66	19.47		
Net Gain/Loss %	-2.74	-5.58	2.49	0.02	-0.27	5.73	-1.52	0.26	-0.83	2.44		

* Areas were calculated using the total number of cells corresponding to each type of change
F = decrease in wetland community due to change; T = increase in wetland community due to change
L = Lake; OW = Open Water; E1 = Floating Emergent; E = Emergent; EW = Tall Wet Emergent; E2 = Tall Dense Dry Emergent; M1= Short Wet Meadow; M = Meadow; T = Treed; U = Upland

decreased by 710.0 ha (17.5 %). The area of tall dense dry emergent and meadow community also experienced notable net losses, while floating emergent and emergent vegetation experienced notable net gains.

5.2.1.2 1955 TO 1964: DECLINING WATER LEVELS

Between 1955 and 1964, 3660.7 ha (11.8 %) of the wetland experienced a change in vegetation community (Table 5-8b). Changes from tall dense dry emergent vegetation accounted for 24.3 % of the total change, of which 355.9 ha (40.0 %) changed to meadow, 215.7 ha (24.3 %) to emergent, and 113.9 ha (12.8 %) to open water. Emergent vegetation accounted for 17.6 % of the total change, of which a total of 294.4 ha (45.6 %) changed to tall dense dry emergent, and 201.7 ha (31.2 %) to open water. Changes within floating emergent represented 16.5 % of the total change. A total of 355.0 ha (58.7 %) of floating emergent changed to open water, 103.2 ha (17.7 %) to tall dense dry emergent vegetation, and 87.5 ha (14.5 %) changed to emergent. Meadow and open water also accounted for 11.7 % and 10.8 % of the total change respectively. Within meadow, approximately 2879 ha (65.0 %) changed to upland. Within open water, 97.9 ha (24.8 %) changed to tall dense dry emergent vegetation, 92.6 ha (23.5 %) to emergent, and 90.2 ha (22.9 %) floating emergent. Overall, upland experienced the greatest net gain of 15.8 %. Open water had a net gain of 9.3 % and floating emergent a net loss of 11.8 %.

5.2.1.3 1964 TO 1978: RISING WATER LEVELS

A total of 4593.7 ha (14.8 %) of wetland changed vegetation communities from 1964 to 1978 (Table 5-8c). Tall dense dry emergent vegetation accounted for 33.1 % of the total change and incurred the greatest net change, a loss in area of 22.5 %. Approximately 90 % of tall dense dry emergent vegetation tended towards wetter wetland communities. A total of 664.0 ha (43.6 %) changed to open water, 337.8 (22.2 %) to tall wet emergent vegetation, 211.0 ha (13.9%) to floating emergent, and 149.6 ha (9.8 %) to emergent. Upland accounted for 19.1 % of the total change, of which 318.4 ha (36.2 %) changed to meadow and 255.6 ha (29.1 %) changed to lake. Changes within the meadow and emergent vegetation represented 15.9 % and 13.1 % of the total change. These communities generally shifted towards wetter vegetation as 214.2 ha (29.4 %) of meadow changed to tall dense dry emergent vegetation, and 160.1 ha (22.0 %) changed to open water but 130.8 ha (18.0 %) did change to upland. In addition, 292.0 ha (48.5 %) of emergent vegetation changed to open water and 99.6 ha (16.6 %) changed to floating emergent. Overall, open water experienced a significant net gain of 21.9 %, and upland area incurred a net loss of 11.2 %.

5.2.1.4 1978 TO 1985: RISING WATER LEVELS

Between 1978 and 1985, 3318.1 ha (10.7 %) of land experienced a change in wetland vegetation community (Table 5-9a). Change within meadow accounted for 18.9 % of the total change, of which 212.8 (33.9 %) shifted to tall dense dry emergent vegetation, 147.6 ha (23.5 %) to upland, and 108.5 ha (17.3 %) to open water. Upland and tall dense dry emergent accounted for 14.6 % and 14.5 % of the total change respectively. Of the upland area to change, 121.2 ha (25.0 %) changed to open water, 107.8 ha (22.3 %) to lake, and 95.1 ha (19.7 %) to meadow. Open water and tall wet emergent vegetation accounted for 154.7 ha (32.2 %) and for 107.2 ha (22.3 %) of the change from tall dense dry emergent vegetation. Open water, floating emergent, emergent, and tall wet emergent vegetation combined to account for 46.5 % of the total change. Although nearly 95 % of open water shifted towards drier vegetation communities, a high percentage of floating emergent, emergent, and tall wet emergent vegetation shifted towards a wetter wetland community. Open water incurred another substantial net gain of 17.6 % in area as water levels continued to rise, while meadow experienced a net loss of 11.5 %.

5.2.1.4 1985 TO 1995: DECLINING WATER LEVELS

A total area of 3360.6 ha (10.8 %) of wetland experienced a change in vegetation community between 1985 and 1995 (Table 5-9b). Noticeable changes occurred within open water, floating emergent, and tall dense dry emergent vegetation. Open water accounted for 40.2 % of the total change, of which 649.0 ha (48.1 %) changed to tall dense dry emergent, 239.5 ha (17.7 %) to floating emergent, 211.0 ha (15.6 %) to emergent, and 128.6 ha (9.5 %) to meadow. Floating emergent represented 13.9 % of the total change, of which 282.3 ha (60.3 %) converted to tall dense dry emergent, 75.4 ha (16.1 %) to open water, and 62.1 ha (13.3 %) to emergent. Tall dense dry emergent vegetation accounted for 13.8 % of the total change, of which 280.2 ha (60.3 %) changed to meadow. Overall, the wetland communities shifted towards drier vegetation. The area of open water experienced a net loss of nearly 36 %, whereas tall dense dry emergent and meadow experienced gains of 27.5 % and 12.8 % respectively.

5.2.1.5 1995 TO 1999: DECLINING WATER LEVELS

From 1995 to 1999, approximately 2071.0 (6.7 %) ha of the wetland incurred a change in vegetation community (Table 5-9c). Tall dense dry emergent vegetation accounted for the greatest percentage of change. A total of 131.9 ha (29.1 %) shifted to open water, 100.8 ha (22.2 %) to meadow, 91.4 ha (20.2 %) to floating emergent, and 75.1 ha (16.6 %) to upland. Open water represented 18.4 % of the total change, of which 182.1 ha (47.8 %) changed to tall dense dry emergent vegetation. Changes within upland accounted for 17.0 % of the total change, of which 153.3 (43.4 %) shifted to meadow, and 95.4 ha (27.0 %) shifted to tall dense dry emergent vegetation. Meadow represented 16.6 % of the total change. A total of 143.5 (41.7 %) ha of meadow changed to upland and 123.0 ha (35.8 %) changed to tall dense dry emergent vegetation. Net changes between 1995 and 1999 are not as substantial as previously noted, but open water continued to suffer with a net loss of 5.6 %, and tall dense dry emergent expanded by 5.7 %.

5.2.1.6 SUMMARY

The results of the spatial analysis provided an indication of the types of changes that occur within the wetland vegetation communities over time and in part due to water level fluctuations. A summary of the key findings is presented in Table 5-10.

TABLE 5-10: KEY FINDINGS RELATED TO TYPE OF CHANGE

RESPONSE OF WETLAND	
TO DECLINING WATER LEVELS	**TO RISING WATER LEVELS**
• Wetter vegetation communities shift to drier vegetation	• Drier vegetation communities shift to wetter communities
• Losses in lake to upland	• Gains in lake from upland
• Significant losses in open water as the community changes to floating emergent, emergent, and tall dense dry emergent vegetation	• Significant gains in open water from floating emergent and emergent vegetation communities • Changes in emergent vegetation to floating emergent vegetation communities
• Significant gains in tall dense dry emergent vegetation from open water, floating emergent, emergent, tall wet emergent vegetation communities	• Significant losses in tall dense dry emergent vegetation to open water, floating emergent, emergent, and tall wet emergent vegetation
• Losses in short wet meadow to meadow	• Gains in short wet meadow from meadow and upland
• Gains in meadow from tall dense dry emergent	• Changes in meadow to open water, tall dense dry emergent vegetation, and upland communities
• Gains in upland from meadow and lake	• Significant losses in upland as community changes to meadow and lake, also some change to open water

5.2.2 LOCATION OF CHANGE

The second component of the spatial overlay analysis determined the location of changes that occurred within the wetland between the periods of study. The spatial overlay of the grids containing the wetland data for contiguous years identifies areas in the wetland where changes have and have not occurred. To and from changes in the wetland communities were identified from the overlay analysis. Changes to wetland communities between the periods of study are summarized.

5.2.2.1 1945 TO 1955: RISING WATER LEVELS

Between 1945 and 1955, there were observable changes to wetter vegetation communities particularly within the Inner Bay (Figure 5-6a). Areas in the Crown and Company Marshes changed to open water, floating emergent, emergent, and tall wet emergent vegetation. Areas along the northern section of the outer peninsula between Squires Ridge and Bluff Point also tended to these wetter communities. There were notable changes to short wet meadow at the base of the Crown Marsh and along the southern shore of the outer peninsula. The ridges in the outer peninsula generally tended towards tall dense dry emergent, meadow, and treed vegetation. Lake area along the southern shoreline of the entire peninsula and around the eastern tip increased. Areas within Big Creek generally tended to tall dense dry emergent and meadow vegetation. However, open water and floating emergent vegetation communities did develop in water channels in Big Creek.

5.2.2.2 1955 TO 1964: DECLINING WATER LEVELS

Between 1955 and 1964, as water levels declined to one of the lowest periods on record, the vegetation generally shifted towards drier vegetation communities and upland areas (Figure 5-6b). Tall dense dry emergent and meadow vegetation appeared within the marshes of the Inner Bay. Areas of the Company Marsh tended to tall dense dry emergent vegetation and open water communities. In addition, smaller patches of floating emergent vegetation developed in the marsh. Vegetation within the Crown Marsh mainly tended to tall dense dry emergent and meadow vegetation, but patches of floating emergent,

emergent, and short wet meadow vegetation appeared. Areas in the outer peninsula tended to meadow and upland communities, although open water and emergent vegetation developed in the northern sections open to the Outer Bay. Large areas within Big Creek remained unchanged, but areas of meadow and upland developed in the river channel and areas of open water developed in the NWA.

5.2.2.3 1964 TO 1978: RISING WATER LEVELS

As water levels rose from 1964 to 1978, there were observable changes to wetter vegetation communities (Figure 5-7a). Open water, floating emergent, emergent, and tall wet emergent vegetation increased across Long Point. A large area of vegetation tended to short wet meadow in the Crown Marsh. There were also noticeable increases to lake along the shoreline and northern extent of the wetland in the Outer Bay. A significant area of wetland changed to upland along the southern shoreline of the Company Marshes. Tall dense dry emergent and meadow vegetation appeared inland along ridges in the outer peninsula and tall dense dry emergent occurred within the Long Point NWA, just to the east of Squires Bay. Areas of Big Creek tended to tall dense dry emergent and meadow vegetation, but these increases mainly occurred along the marsh-upland boundary. Tall wet emergent vegetation developed in the area that later became diked, and floating emergent and emergent vegetation developed in the Big Creek River delta.

5.2.2.4 1978 TO 1985: RISING WATER LEVELS

From 1978 and 1985, water levels continued to rise but the increase in water levels was not as substantial as during the increase from 1964 to 1978, therefore the amount of visible change is not as apparent (Figure 5-7b). There are however, some notable changes. First, the amount of lake increased along the shore of the peninsula, especially along the northern shore of the outer tip. There were increases to open water along the outer peninsula between Squires Bay and Bluff Point and in the outermost tip. Smaller increases in open water occurred in the Inner Bay and Big Creek Marshes. Patches within the Long Point NWAs and Company Marsh changed to open water, floating emergent, and tall wet emergent, but these patches were notable smaller in area and more fragmented in appearance. Changes to short wet meadow along the southern shore of the outer peninsula and Crown Marsh also occurred. Drier vegetation communities, such as meadow and treed, tended towards tall dense dry emergent vegetation along the peninsula, but in the Crown and Big Creek Marshes, tall wet emergent vegetation shifted to tall dense dry emergent vegetation. There was also a large patch of vegetation that changed to meadow in the Hahn Unit at the western most edge of the map.

5.2.2.5 1985 TO 1995: DECLINING WATER LEVELS

Between 1985 and 1995, water levels declined from record high levels to medium levels, and as a result, the vegetation showed visible changes to drier communities (Figure 5-8a). Most noticeable were changes to tall dense dry emergent and meadow vegetation along the entire peninsula. There were minor increases in floating and emergent vegetation in the Big Creek Marsh Unit, the Crown Marsh, Long Point NWA, and

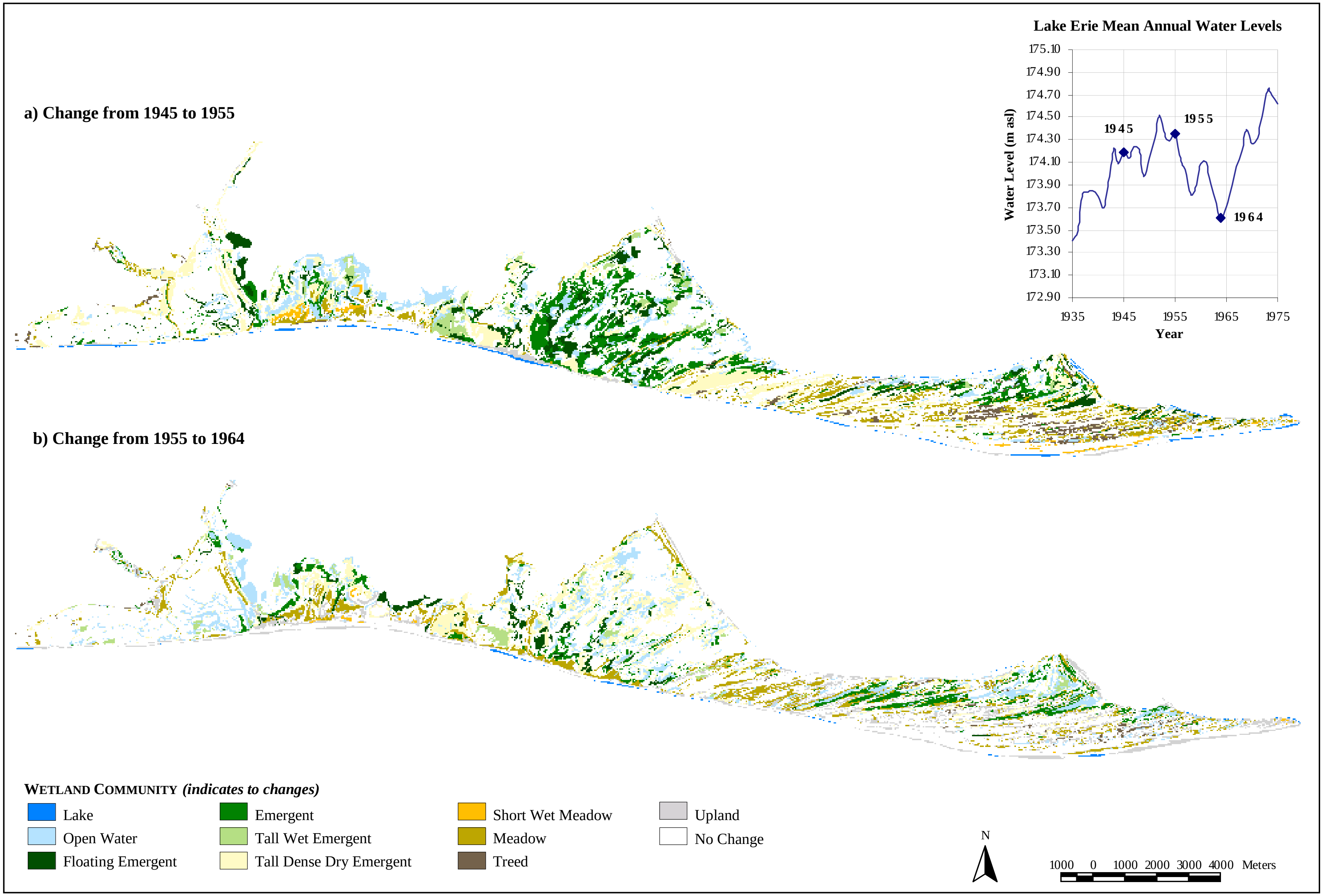


FIGURE 5-6: WETLAND VEGETATION COMMUNITY CHANGE IN LONG POINT, 1945-1964

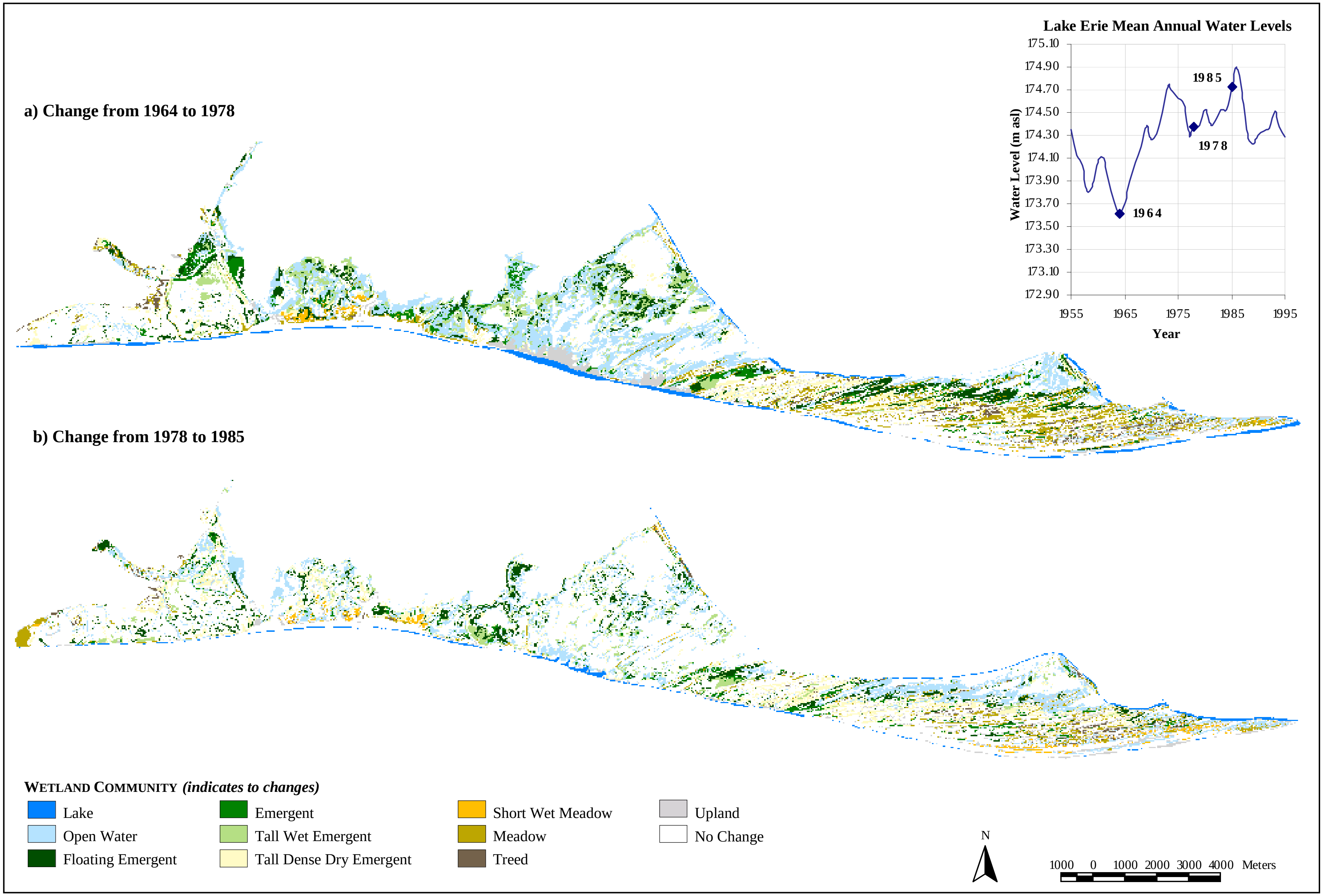


FIGURE 5-7: WETLAND VEGETATION COMMUNITY CHANGE IN LONG POINT, 1964-1985

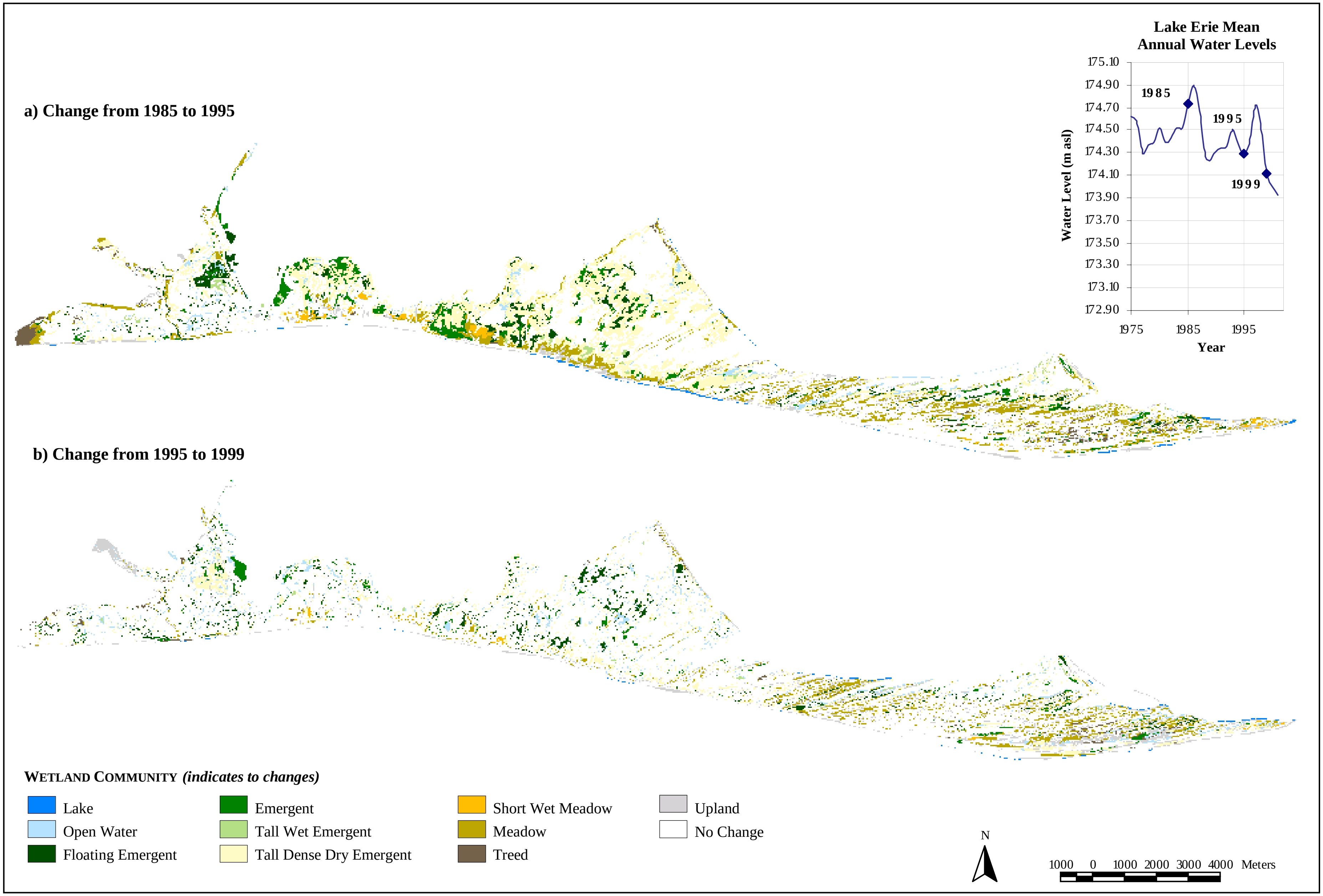


FIGURE 5-8: WETLAND VEGETATION COMMUNITY CHANGE IN LONG POINT, 1985-1999

Company Marsh. Most of these changes coincided with areas that were previously open water. In addition, short wet meadow increased along the southern shore of the peninsula around the Crown and Company Marshes, Long Point NWAs, and along the outer most portion of the tip. Lake areas along the shoreline mainly changed to upland, but there were some minor increases in lake.

5.2.2.6 1995 TO 1999: DECLINING WATER LEVELS

Water levels continued to decline between 1995 and 1999. During this period, there were marginal changes within the wetland (Figure 5-8b). Increases in meadow and upland were observed in the outer peninsula. A large area within the Big Creek River channel converted to upland, but this change is likely the result of differences in the interpretation of the aerial photography. Other areas of significant change occurred within the Big Creek dike where tall dense dry emergent vegetation increased, along the Inner Bay just to the east of the Big Creek NWA where emergent vegetation increased, and within the Crown Marsh where floating emergent vegetation increased. The rest of the wetland complex experienced minor changes to open water and tall dense dry emergent vegetation communities.

5.2.2.7 SUMMARY

Areas within the Long Point wetland complex have experienced similar changes to wetland vegetation communities in relation to periods of different water level conditions, especially during periods of increasing water levels. The types of changes, and the locations of these changes are dependent on the geomorphic landform. Key patterns in the location of to changes are summarized (Table 5-11). There were also several areas in the wetland that remained unchanged over time. No changes were observed in lake areas further offshore and upland areas along the mainland. Little changes were also observed in large portions of Big Creek, at the base of the Crown Marsh, portion of the outer peninsula just east of Squires Bay, and deeper and larger areas within the Inner Bay.

TABLE 5-11: KEY FINDINGS RELATED TO LOCATION OF CHANGE

RESPONSE OF WETLAND	
TO DECLINING WATER LEVELS	**TO RISING WATER LEVELS**
• Changes to tall dense dry emergent and meadow vegetation in the Inner Bay and northern portion of the outer peninsula • Smaller patches of open water and floating emergent vegetation communities develop	• Changes to open water, floating emergent, emergent and tall wet emergent vegetation in the Inner Bay and northern portion of the outer peninsula • Significant increases in short wet meadow at the base of the Crown Marsh and along the southern shore of the peninsula
• Shifts to meadow and upland in the outer peninsula	• Changes to tall dense dry emergent, meadow and treed vegetation in the outer peninsula • Changes to open water and wetter emergent communities in swales in the outer peninsula
• Few changes in the Big Creek, but some observable changes to meadow and tall dense dry emergent vegetation • Meadow and upland in Big Creek River channel	• Open water and floating emergent in Big Creek water channels • Tall wet emergent, floating emergent and emergent in delta and diked areas of Big Creek
• Increases in upland along the shoreline and outer peninsula	• Increases in lake along the shoreline of the peninsula and north of the peninsula in the Inner Bay

5.3 DISCUSSION

Spatiotemporal trend analyses provide a key to how wetland vegetation responds over time to changes in water level conditions that can be used to infer how the wetland may respond to alterations in water levels due to projected climate change. Historical wetland vegetation response at Long Point is characterized for declining and rising water level conditions from the results of the analysis. Other factors affecting wetland change are also addressed. The implications of enhanced global warming and projected water level declines to the composition and structure of the Long Point wetland complex are discussed. Finally, the results of the spatiotemporal analysis are briefly compared to similar studies.

5.3.1 DECLINING WATER LEVELS

A number of changes occurred within the Long Point wetland complex as water levels declined. First and foremost, the vegetation within the wetland tended towards drier wetland vegetation communities. Generally, there were significant increases in the area of tall dense dry emergent and meadow vegetation, especially in the Inner Bay and Big Creek Marshes and along ridges in the outer peninsula. The increases in these drier communities were generally offset by decreases in the area of wetter wetland communities. There were notable decreases in the lake community along the southern shore of the peninsula as water levels declined. Consequently, upland areas increased as the receding lake levels exposed sandy areas along the shore; these areas remained void of wetland vegetation. Meadow vegetation also expanded lakeward developing on newly exposed areas of the lake substrate. Areas of open water in the Inner and Outer Bay developed into wetter and drier forms of emergent and meadow vegetation. Furthermore, there were decreases in the area of floating emergent, tall wet emergent and short wet meadow vegetation as these communities tended towards their drier counterparts and areas of tall dense dry emergent vegetation tended to meadow.

Generally, there was less fragmentation and complexity within the wetland during periods of declining water levels. The shapes of the patches in the wetland and vegetation communities are simpler as drier wetland vegetation communities expand and form larger, solid, and continuous patches of vegetation. During extreme low water level conditions, however, fragmentation within open water community increased as floating emergent and emergent vegetation developed in shallower open water areas. Furthermore, the complexity of the patch shapes for the tall wet emergent, tall dense dry emergent, and meadow vegetation increased as water levels declined. Tall wet emergent vegetation became more fragmented as water levels declined, as the vegetation tended to tall dense dry vegetation that thrive in less saturated areas of the community. Tall dense dry emergent and meadow vegetation expanded and colonized newly exposed areas by the receding water levels. The areas of tall dense dry emergent and meadow vegetation were patchier and irregular in shape, thus the complexity of their patch shapes increased.

Less fragmentation in the wetland also affected the distribution of patch types and the variability in patch sizes within the wetland. The wetland was composed of fewer, but larger, patches and these patches were located further away from each other, thus patches within the wetland or patches of similar vegetation communities became more isolated from each other. In addition, there is less variability among patch sizes within the wetland since larger continuous patches of wetland vegetation developed. Interspersion of patch types decreased as the distribution of the patches within the wetland and vegetation communities became less proportionate.

Finally, as water levels declined there was greater diversity within the wetland; or in other words, dominance of a single wetland community in the landscape decreased. During higher water level periods, lake and open water were the dominant communities in the wetland. As water levels declined these communities became less dominant in the wetland as other drier vegetation communities expanded. The proportion, distribution, and abundance of the area between different community types became more equal, and as a result, the evenness of the landscape increased.

5.3.2 RISING WATER LEVELS

During periods of rising water levels, several trends in wetland community response were evident in Long Point. Drier wetland vegetation shifted towards wetter wetland communities. Generally, the area of floating emergent vegetation increased, in addition to the areas of tall wet emergent and short wet meadow that developed as drier emergent and meadow communities became flooded and interspersed by higher lake levels. The amount of lake surface increased as lake water inundated low-lying vegetated wetland and upland areas along the southern shore and around the tip of the peninsula. The extent of upland area decreased, as lake water flooded upland areas along the shore of the peninsula and wetter vegetation communities compress the extent of upland communities inland as the wetter vegetation migrated landward. There were also shifts in the upland communities to meadow vegetation during periods of rising water levels.

The communities that experienced the greatest amount of change were the tall dense dry emergent and meadow vegetation communities. Tall dense dry emergent and meadow vegetation communities in the wetland were flooded and interspersed by the high lake water and as a result the vegetation became significantly interspersed with water enough so that tall wet emergent and short wet meadow communities develop, or patches within the community are completely obliterated by open water and floating emergent communities. Generally, tall dense dry emergent vegetation tended to open water, floating emergent, emergent, and tall wet emergent vegetation. There were noticeable increases in the amount of open water, floating emergent, emergent, and tall wet emergent vegetation communities in the Inner Bay and Big Creek Marshes. Along the outer peninsula, meadow vegetation tended to open water and tall dense dry emergent with minor shifts to floating emergent and emergent vegetation. Finally, changes in upland areas along the shore were common during periods of increasing lake levels; upland generally tended to meadow as the soil moisture of the soil increased or changed to lake if the vegetation was completely flooded by the high lake levels.

As Lake Erie water levels rose, there was greater fragmentation and complexity within Long Point. Wetland vegetation communities became significantly interspersed and fragmented by higher water. The shape of patches within the wetland became more complex and convoluted due to the greater fragmentation and interspersion of the wetland vegetation communities. Greater fragmentation also resulted in greater variability among patches sizes within the wetland and wetland communities. The distribution of the patches became less isolated. Larger patches within the wetland were interspersed with water, creating smaller patches that were closer in distribution to each other. Therefore, interspersion between patch types increased in the wetland and the distribution of patch type adjacencies becomes more proportionate. Furthermore, the wetland becomes dominated by a fewer number of wetland communities, mainly lake and open water are prominent within the landscape, and the diversity decreases. Therefore, the proportion, distribution and abundance of area between different patch types in the wetland and vegetation communities become more uneven.

5.3.3 OTHER FACTORS AFFECTING CHANGE

Although water level fluctuations have been instrumental in changing the structure and composition of the wetland communities within Long Point, there have been other processes and factors that have affected change across the wetland during the period of study. Change in the scale and quality of the aerial photography and differences in their interpretation had some influence on the change analysis. Prior to 1978, the scale of the photographs and interpreted maps were smaller scale than the later years, and quality of the photographs were often poorer. This undoubtedly resulted in some discrepancies or differences between the landscape and class metrics over time. Furthermore, some discrepancies between the wetland vegetation classifications were produced from different interpretations of the original aerial photography; this would of resulted in some differences in the trends observed between 1995 and 1999.

Land use changes are another process within the landscape that would greatly impact the results of the analysis. Fortunately, the natural environment of Long Point was been protected through numerous conservation efforts, and thus development has been limited to the mainland and certain portions of the peninsula. There has been some development in the wetland, mainly the creation of the Provincial Park in 1956, the newer Provincial Park in 1961 and the Big Creek NWA dike in 1985. Natural hazards, such as fire and storms, have had an impact on the wetland vegetation communities and structure of the wetland. In 1962 and 1963, fire destroyed portions of the wetland in the Long Point Company Marsh and Provincial Park. Depending on the length of recovery of the wetland vegetation to fire, drier vegetation communities in the wetland may decrease and the burnt areas could be classified as upland areas. In addition, Hurricane

Hazel and other intense storms have resulted in changes along the lake-upland boundary of the peninsula. Storm events would have a greater impact on the physical environment rather than on the wetland vegetation as the configuration of the shoreline along the upland-lake boundary would alter, but during severe storms, wetland vegetation could be completely destroyed.

Finally, the diversity of landforms across the wetland complex also affected change within the wetland vegetation communities. Long Point contains a number of geomorphic forms that influence how the wetland vegetation responds to water level fluctuations. Some areas, such as the Big Creek Marshes and NWA Dike, are less influenced by fluctuations in lake levels than other portions of the wetland. There may be notable differences in the changes within the structure and composition of wetland vegetation communities in these different regions. Despite these additional factors of change, water level fluctuations remain a primary influence of wetland vegetation change within Long Point and thus can be used to assess the response of the wetland vegetation to projected climate change.

5.3.4 PROJECTED CLIMATE CHANGE AND WATER LEVEL DECLINES

Climate change scenarios are projecting Lake Erie water levels to decline significantly under enhanced global warming, two such scenarios suggest a water level decline in the range from 1.36 m to 1.48 m (Lee *et al*, 1996; Staple, 1993). How will projected declines of this magnitude affect the structure and composition of the wetland complex at Long Point? The wetland vegetation response to historical water level fluctuations provides an indication of how the wetland vegetation may respond to future climate change. The spatiotemporal trend analysis presented here suggests a pattern of wetland community response to declining water level conditions that is characterized by:

- Significant increases in tall dense dry emergent and meadow vegetation, especially within the Inner Bay and Big Creek Marshes (excluding the diked area);
- Vegetation shifts to meadow, treed, and upland communities in the outer peninsula;
- Decreases in the spatial extent of open water and deeper water communities;
- Decreases in the complexity of the patches and communities within the wetland;
- Less fragmentation and heterogeneity within the wetland and wetland communities, except for open water which may become highly fragmented;
- Less interspersion in the wetland and between the wetland communities;
- Less variability in patch sizes in the wetland;
- More isolation between patches within the wetland vegetation communities; and,
- Patches that are more evenly distributed across the wetland and more proportionate in area.

These changes in the structure and composition of the Long Point wetland complex may have negative implications to the natural integrity of the wetland. The overall quality and the productivity may decrease as denser and drier vegetation communities expand and non-wetland plants and trees invade. The loss of submergent and floating-leaved vegetation may significantly reduce the utility of the wetland by many wildlife species, especially for shorebirds and migratory waterfowl that use the wetland during their annual migrations. The availability of food and shelter may decrease resulting in greater competition between wildlife and the spread of disease. The overall diversity of the wetland may decrease, as fewer plant and wildlife species inhabit Long Point. Furthermore, there may be more occurrences of fire in the wetland, due to the drier vegetation matter in the wetland. Although fire has not had a significant impact on the wetland, historically, the frequency and intensity of the fires may increase. The implications of projected water levels declines due to enhanced global warming may be detrimental to the natural integrity and diversity of the Long Point wetland complex.

5.3.5 COMPARISON TO SIMILAR STUDIES

The results of the spatiotemporal trend analysis are comparable with other studies that have examined the role of fluctuating water levels on wetland vegetation in coastal wetlands of the Great Lakes. The results of this analysis indicate that during periods of declining water levels conditions, drier emergent and meadow vegetation communities expand while aquatic communities including submergent and floating emergent vegetation decline. Conversely, during periods of rising water levels, submergent and floating emergent vegetation communities increase and drier emergent and meadow vegetation decrease. Similar patterns of wetland vegetation response to water level fluctuations were observed in other wetlands within the Great Lakes (Gottgens *et al*., 1998; Quinlan, 1985; Whillans, 1985; Farney and Bookhout, 1982; ILERB, 1981; Jaworski *et al*., 1979; Harris and Marshall, 1963).

Furthermore, the response of the Long Point wetland complex to projected water level declines presented in this discussion are consistent with two studies that have examined the impacts of future climate change at Long Point. Staple (1993) concluded that climate change and associated water level declines would result in the expansion of drier marsh meadow and terrestrial vegetation in the Long Point wetland complex. Similarly, Bayly (1979a) noted that projected water level declines would lead to an increase in tall emergent vegetation in the Inner Bay. Few comparisons can be made with other studies with regards to analysing the changes in wetland structure and pattern over time and in relation to water level fluctuations, as literature related to this type of analysis in wetland systems is scarce. However, Quinlan (1985) did observe greater interspersion between open water and plant communities during periods of rising water levels and Gottgens *et al*. (1998) observed that the amount of edge and patchiness of emergent vegetation did decrease with declining water level conditions. The landscape analysis for Long Point indicated greater interspersion among wetland vegetation communities, and increased amounts of edge and patchiness in the landscape during periods of rising water levels.

5.4 SUMMARY

The spatiotemporal trend analysis of Long Point characterized how the wetland and vegetation communities have responded over time in relation to water levels fluctuations. The historical response of the wetland to climate change, as indicated by the water level fluctuations, provided a key to how the wetland may respond to future climate change. The analysis presented here suggests that the Lake Erie water level declines, due to enhanced global warming, may have serious repercussions on the composition and configuration of the Long Point wetland complex and in turn impact the natural integrity of the wetland. The responses of the wetland vegetation to historical water level conditions at Long Point were simulated with several spatial models. The results of the modelling efforts are reported in Chapter 6.

6.0 MODELLING ANALYSIS RESULTS AND DISCUSSION

This chapter discusses the results of the wetland vegetation community simulation modelling for Long Point. Simulated wetland vegetation maps for the rule-based model, probability model, and vegetation transition model are presented. The accuracy of each model is evaluated according to the model's ability to correctly predict the area and spatial distribution of wetland vegetation communities within the study area. The three models are discussed separately below. A synopsis of the wetland modelling techniques concludes the chapter. The results of the simulated wetland communities for 1999 are presented in this chapter; the results for the other years are provided in Appendices H though J. Tables summarizing the accuracy results for each simulation run for every year and version of the models were compiled and are provided in Appendix K.

6.1 RULE-BASED MODEL

The rule-based model simulated wetland vegetation communities according to pre-existing wetland vegetation during a base year, a hypothetical lake level, the tolerance depth ranges of the different wetland vegetation communities established from the literature, and adjacency of lake, open water, and upland cells to each other. The results from each version of this model are presented; overall accuracy was determined using an average of the number of correctly predicted cells from all the simulated years of data, as the model was derived from wetland community tolerance depth ranges established from the literature. A discussion assessing the overall results from each version of the model is presented.

6.1.1 VERSION 1.0

Overall, Version 1.0 (v1.0) was 65.9 percent (%) accurate in predicting wetland community response at Long Point. The percentage of correctly predicted cells ranged from 64.7 % for 1999 to 68.9 % for 1985. The model was successful at correctly predicting lake and upland communities. Approximately 84.7 % of lake cells and 68.7 % of upland cells were correctly predicted. Moderate success was achieved in simulating the open water (47.8 %), floating emergent (29.4 %), and emergent (29.2 %) vegetation communities. Tall dense dry emergent vegetation was the least accurate community; only 13.7 % of the cells were correctly predicted.

Of the 34.1 % of cells that were incorrectly predicted, a total of 68.6 % of the cells were within one community of the actual wetland value; on average 44.5 % of the incorrectly predicted cells were within one community above (i.e. drier) the actual value and the remaining 24.1 % were below (i.e. wetter) the actual value. For most years, the majority of incorrectly predicted cells were above (drier than) the actual values; however, the majority of incorrectly predicted cells for 1985, the highest water level periods on record, were below (wetter than) the actual values. An additional 18.7 % of the cells differed by two communities. The total percentage of incorrectly predicted cells within one or two communities from the actual values totalled 87.3 %.

There were noticeable differences between the actual and predicted areas of the wetland communities (Figure 6-1). Generally, there were larger differences between the predicted and actual areas for the deeper or wetter wetland communities along the continuum than for drier communities. Tall dense dry emergent vegetation and upland area were consistently underestimated for all simulated years, and floating emergent and emergent vegetation were overestimated. Also interesting to note, is the percentage differences in area for communities that account for smaller areas within the wetland. The areas of floating emergent, tall wet emergent, short wet meadow, and treed vegetation are much smaller in extent compared to the other communities, as a result, the percentage difference between the predicted and actual areas are greatly overestimated.

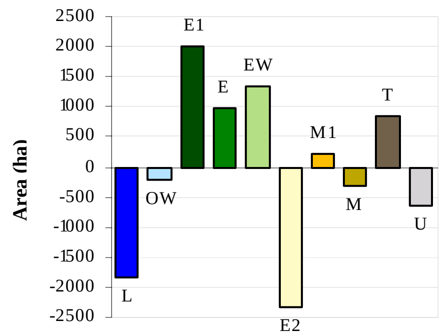


FIGURE 6-1: DIFFERENCE BETWEEN PREDICTED AND ACTUAL AREAS V1.0, 1999

Spatially, the results from the simulation modelling were satisfactory (Figure 6-3a). The version was able to depict prominent features within the wetland complex, such as linear channels of water in the Big Creek Marshes, quite well but the model was unable to simulate the spatial distribution and extent of certain vegetation communities correctly. Generally, the model was unable to decipher some lake and upland areas along the shore. Areas along the peninsula were classified as treed or drier wetland vegetation rather than upland, and areas of lake around the outer peninsula were misclassified as open water. In addition, open water areas in deeper sections of the Inner Bay were misclassified as lake.

There were also classification errors within the vegetated areas of the Inner Bay and Big Creek Marshes. Open water areas in the Long Point Company Marsh were classified as shallower (and drier) wetland communities such as floating emergent or emergent vegetation. Areas in the Big Creek, Crown, and Company Marshes were misclassified as wetter wetland communities compared to the actual values. For example, areas of tall dense dry emergent vegetation in the Company Marsh were generally classified as wetter forms of emergent vegetation except for 1964, the lowest water level period on record, when the model accurately simulated tall dense dry emergent vegetation in the Crown Marsh. In addition, there are horizontal edges in the northern portion of the outer peninsula marking the transitions between vegetation communities for 1955, 1978, and 1985, the three years in which water levels had been rising. These horizontal edges coincided with bands of similar elevation in the topographic model. It is uncertain whether this is a true representation of the real topography, as there may be a shelf or shallower platform off the shore of the peninsula, or due to an error in the elevation model provided by the OMNR.

The spatial distribution and extent of errors between the actual and predicted wetland community values were generally consistent between each simulated year. For all years, there were large areas of correctly predicted cells of upland along the mainland and in the outer peninsula, open water in the Inner Bay, and lake in the Outer Bay and along the shore of Lake Erie. There were larges areas north of the peninsula in the Outer Bay and areas in the Inner Bay that were incorrectly classified within one wetland vegetation community from the actual value. Areas in Big Creek were generally two communities from the actual value. Smaller areas within the Long Point Company Marsh were more than two communities from the actual values. Greater classification errors also occurred along the lake-upland boundary on the southern shore of the entire peninsula and along ridges in the outer peninsula (Figure 6-3b).

6.1.2 VERSION 1.1

The simulated results for Version 1.1 (v1.1) of the model were better compared to v1.0. Version 1.1 was 81.8 % accurate in simulating wetland vegetation response within the wetland complex. The percentage of correctly predicted cells ranged from 79.1 % in 1964 to 84.5 % in 1999. The model was successful at predicting lake, open water, and upland communities; the percentages of cells to be correctly predicted within these communities were 98.5 %, 84.0 % and 63.8 % respectively. The model achieved moderate success at predicting meadow (29.6 %), tall dense dry emergent (25.3 %), emergent (22.3 %), floating emergent (22.2 %), and tall wet emergent vegetation (20.3 %). The model was least accurate at predicting short wet meadow and treed vegetation, as 11.2 % and 11.3 % of cells were correctly predicted respectively. The success rates for the lake, open water, meadow, tall dense dry emergent, and tall wet emergent vegetation communities improved over the results from v1.0, but the percentages of cells correctly predicted for the treed, short wet meadow, emergent, floating emergent vegetation, and upland communities did decrease.

Version 1.1 incorrectly predicted 18.2 % of the cells. Of these cells, only 39.9 % of the cells were within one wetland community from the actual wetland value and another 31.5 % of the cells were two communities from the actual wetland community value. A total of 71.4 % of the incorrect cells differed by one or two community values. There were no evident trends in the differences between the actual and predicted wetland community values as to whether the predicted community values were wetter or drier than the actual values.

Comparison of the actual and predicted aggregate areas for the communities revealed that the model significantly underestimated areas of the tall dense dry emergent vegetation and upland communities, and overestimated the areas of the emergent and treed vegetation communities for all years (Figure 6-2). Similar to v1.0, the percentage differences in areas indicate that the tall wet emergent, short wet meadow, and treed vegetation communities were greatly overestimated in v1.1 of the model.

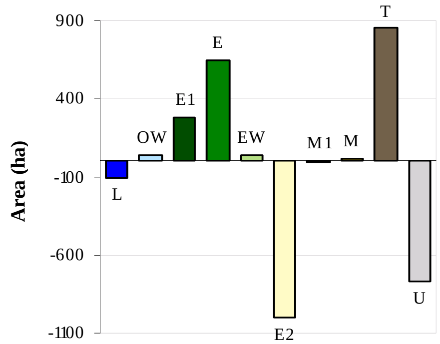


FIGURE 6-2: DIFFERENCE BETWEEN PREDICTED AND ACTUAL AREAS V1.1, 1999

Visually, the output results of the simulated vegetation communities are quite good (Figure 6-4a). Areas along the shore of the peninsula were correctly classified as lake, and deeper sections of the Inner Bay were correctly classified as open water in this version. There were notable errors with the classification of other wetland vegetation communities though. Like v1.0, areas in the Big Creek Marshes were classified as wetter vegetation communities compared to the actual vegetation. Areas in the Crown and Company Marshes remained misclassified, but did tend to drier communities compared to those communities predicted in v1.0. Overall the model simulated a greater amount of tall wet emergent and tall dense dry emergent vegetation during periods of declining water levels. During periods of rising water levels, particularly for 1985, upland areas along the shore of the peninsula were washed out and classified as lake or open water. Similarly to v1.0, there are horizontal edges between wetland communities in the northern section of the outer peninsula, but these edges appear in the simulated wetlands for six years, except 1964 the year that marked the lowest water level period on record.

The spatial distribution of the errors indicates that most variation between the actual and predicted wetland community values occurs in the outer peninsula and along the southern shore of the peninsula. Smaller differences between the actual and predicted values (i.e. the predicted vegetation is one or two communities different from the actual community class) are evenly distributed across the entire wetland. Although for 1978, 1995 and 1999 greater differences in wetland community values were observed in the Inner Bay (Figure 6-4b).

6.1.3 COMPARISON OF VERSIONS

A comparison of the results of the simulated wetland vegetation response between the two versions of the rule-based model is complex. Depending on the measurements of accuracy examined, both versions of the model outperformed the other. Version 1.0 was more successful in predicting a higher percentage of floating emergent, emergent, and upland cells correctly. The classification accuracies in predicting floating emergent and emergent vegetation were 7.3 % and 6.8 % higher in v1.0 than in v1.1. The accuracy in

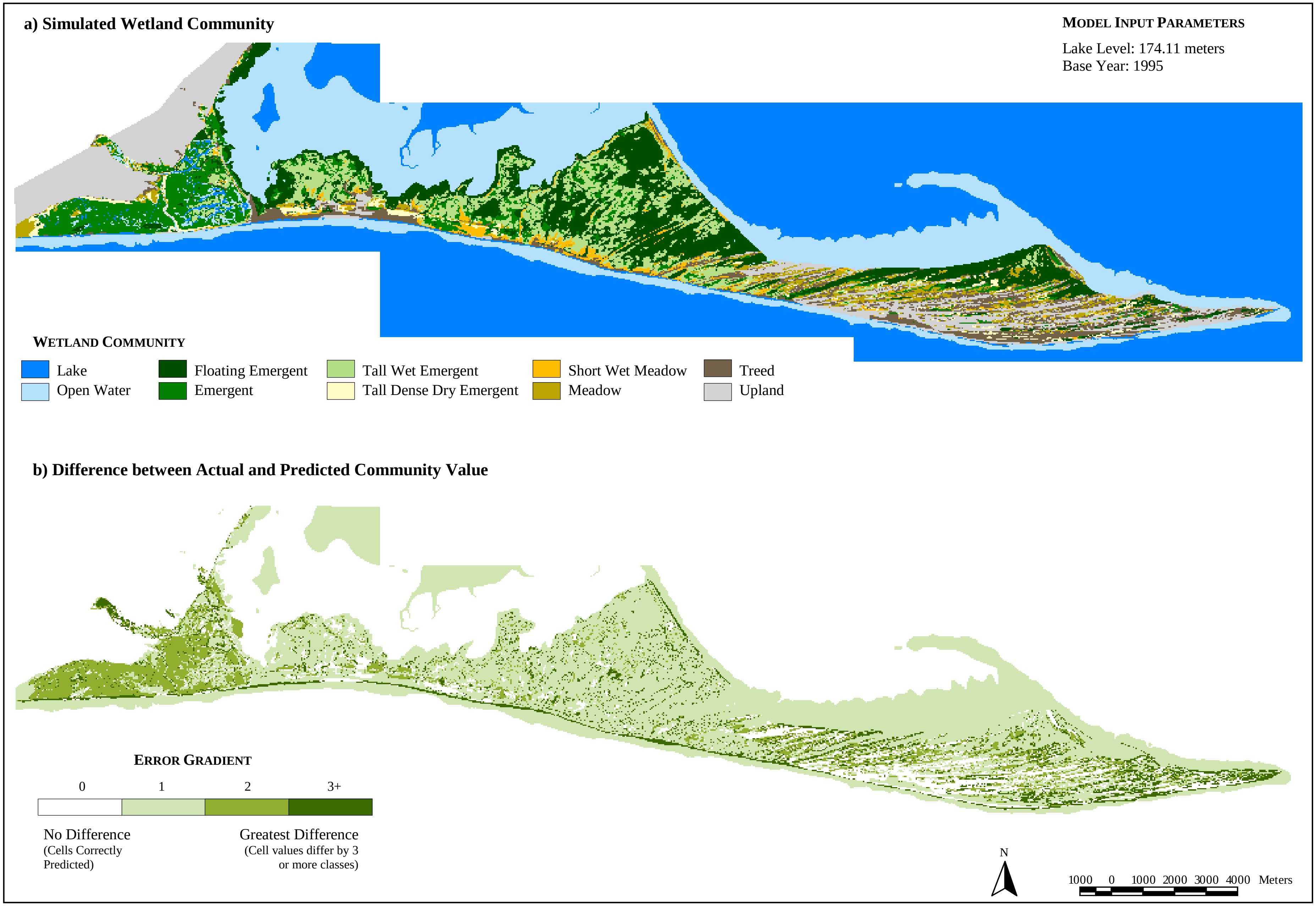


FIGURE 6-3: RESULTS OF THE RULE-BASED MODEL V1.0, 1999

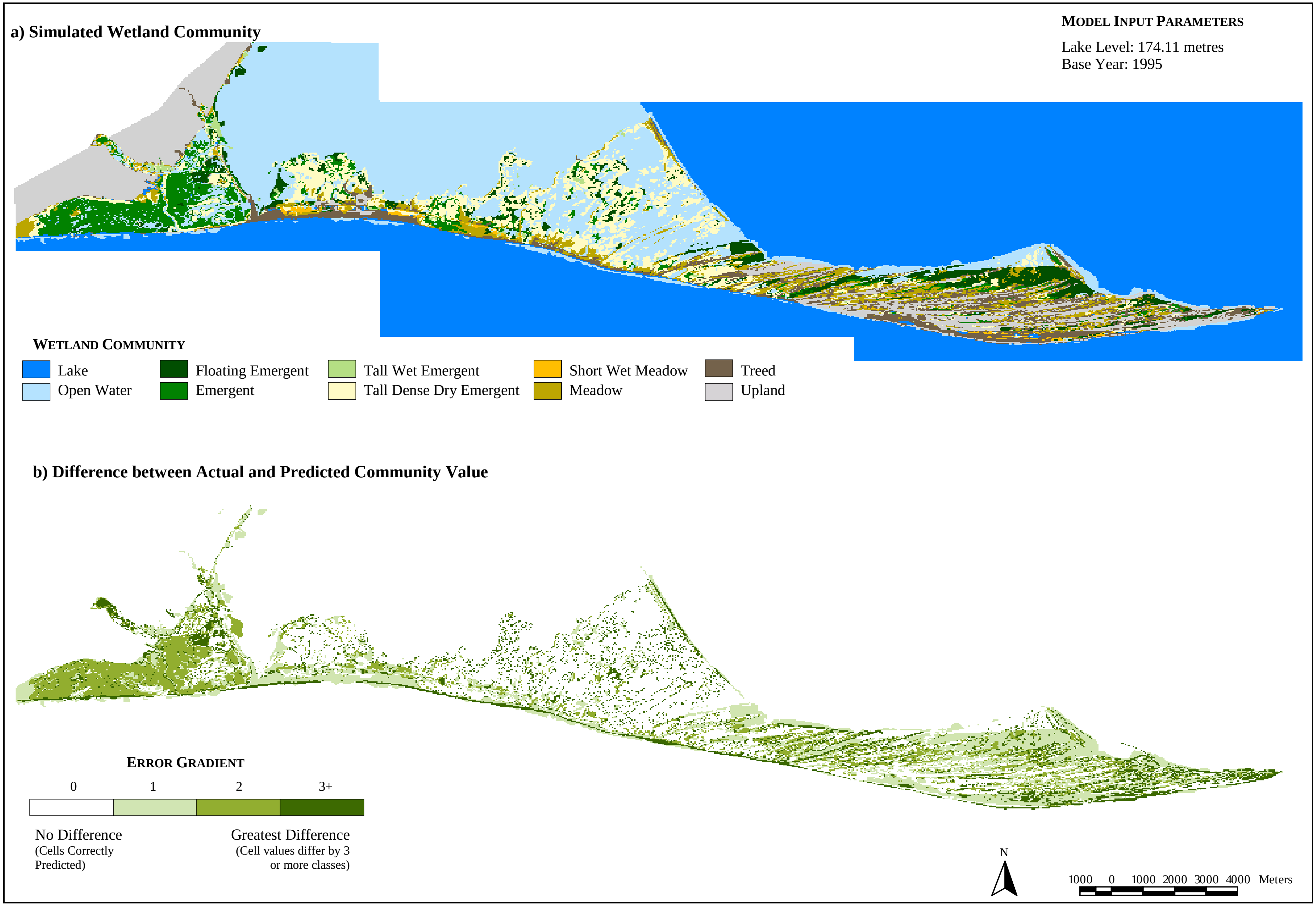


FIGURE 6-4: RESULTS OF THE RULE-BASED MODEL v1.1, 1999

predicting upland area was also higher than v1.1 by 5.0 %. The percentages of short wet meadow and treed cells correctly predicted were marginally higher in v1.0. Furthermore, a greater number of incorrectly predicted cells were within one or two communities of the actual wetland community value. The percentage of incorrect cells within one wetland community of the actual value totalled 68.6 % in v1.0 compared to 40.0 % in v1.1. A total of 87.3 % of the cells were within two communities in v1.0 compared to 71.5 % in v1.1. The amount of error can be somewhat misleading however, when considering that a significant number of open water and lake cells misclassified in v1.0 were correctly classified in v1.1 due to modifications made to the decision rules applied in v1.1. These cells may account for some of the differences in community value reported for the two versions.

Version 1.1 was more successful in terms of the overall accuracy of the simulated wetland vegetation response. The overall accuracy of the model increased from 65.9 % in v1.0 to 81.7 % in v1.1, an improvement of 15.2 %. In addition, v1.1 was more successful at predicting a higher percentage of cells in the lake, open water, tall wet emergent, tall dense dry emergent, and meadow vegetation communities correctly compared to the previous version. The classification of open water improved the most as the accuracy of this community increased 36.2 %; the accuracy of lake improved by 13.8 %, tall dense dry emergent vegetation by 11.6 %, and meadow by 10.0 %. Tall wet emergent vegetation experienced a marginal improvement in classification accuracy. The spatial distribution and extent of lake and open water were more accurately simulated in v1.1; shoreline areas along Lake Erie were correctly classified as lake, and deeper water areas in the Inner Bay, including the Company Marsh, were more likely classified as open water in the second version. Both versions of the model overestimated the area of emergent and treed vegetation, and underestimated the areas of tall dense dry emergent vegetation and upland community. Furthermore, both versions were sensitive to the horizontal contour bands of elevation in the northern section of the outer peninsula, however, in v1.0 this was only evident in periods where water levels were rising. Errors in the topographic model may have contributed to this problem.

6.1.4 RULE-BASED MODEL ASSESSMENT

The decision rule-based modelling technique proved to be fairly successful. The wetland vegetation communities simulated by the model produced areas of contiguous cells of smaller and larger patches within the landscape. The model was successful in simulating the theoretical response of wetland vegetation communities to rising and declining water level conditions. During periods of declining water levels, the wetland vegetation communities tended to drier communities along the wetland continuum, and during periods of rising water levels, the wetland communities tended to wetter vegetation communities. The model was also able to delineate features within the landscape, such as water channels in the Big Creek Marshes and ridges and islands along the peninsula.

For the most part, the model was able to simulate the spatial distribution and extent of patches, but failed to classify these patches with the correct wetland vegetation community class. The model also significantly overestimated the areas of smaller wetland communities in the landscape. The percent difference between the actual and predicted areas of floating emergent, tall wet emergent, short wet meadow, and treed were commonly overestimated by more than double the actual area of these communities. Finally, the model was sensitive to irregularities in the topographic model. As noted earlier, it is unclear whether these irregularities are due to errors in the data or consist due to natural processes in the study area. Existing OBM map sheets of the study area were examined and the elevation and contour marks on the maps were compared to the information provided on the topographic model and there were discrepancies between these values. Therefore, the problem could likely be due to errors in the topographical data. Alternatively, natural processes, mainly sediment transport, could have also been a source of error. The topographic model and OBM maps contain data representing one specific period in time. The topography of the region is not static, but constantly evolving, due to long-shore drift and sediment erosion and deposition. Thus, sediment transport may have contributed to some of these irregularities in the data.

6.2 PROBABILITY MODEL

The probability model simulated the wetland vegetation based on the likelihood of certain wetland communities occurring at specific depth ranges. The accuracy of the model was determined by averaging the simulated results for five years; two years were averaged to determine the accuracy of the model in predicting vegetation response for declining water level conditions (1964, 1999) and three years were averaged for rising water level conditions (1945, 1955, 1978). The results for 1985 and 1995 were excluded since these years were used to derive the probabilities for the rising and declining water level periods. The results of the three versions of the model are presented. Differences between the performances of the versions and the overall impressions of the modelling technique are discussed.

6.2.1 VERSION 2.0

Version 2.0 (v2.0) accurately predicted 55.7 % of the cells correctly. The percentage of correctly predicted cells ranged from 53.6 % for 1955 to 59.2 % for 1999. The results for years with declining water level conditions (57.7 %) were slightly more accurate than the results for rising conditions (54.4 %). The model was able to successfully predict lake and upland areas; the success rates for these communities were 75.1 % and 56.9 % respectively. The model achieved moderate success at predicting open water communities (37.8 %). The success rates for the other communities were poor; short wet meadow was the least accurate community modelled as only 1.0 % of the cells were correctly predicted. Generally, the average percentage of cells correctly predicted for each community were higher for periods of declining water levels than for rising water levels, except for floating emergent, emergent, and tall wet emergent vegetation. But the differences in correctly predicting these communities for declining and rising water level conditions ranged from 0.4 to 1.4 %.

Approximately 44.3 % of the cells in the model were incorrectly predicted. Of those cells, 38.4 % were within one wetland vegetation community from the actual value. Another 10.0 % were two communities from the actual value. Therefore a total of 48.5 % were within one or two communities from the actual wetland value. There were a greater portion or percentage of incorrectly predicted cells above the actual values, i.e. the predicted community of the cells were drier than the actual community values.

There were several notable differences between the predicted and actual areas of the wetland communities. The aggregate areas of open water and upland community were consistently overestimated. The model also underestimated the amount of lake area each year. The absolute magnitude of the differences between the actual and predicted areas were larger the simulations for years prior to 1985. There were notable increases in the percentage differences between the actual and predicted areas in 1945 for tall wet emergent and treed vegetation, and in 1964 for short wet meadow. The predicted areas of these communities were overestimated by more than 200 % of their actual areas. The area statistics for 1999 are reported in Figure 6-5.

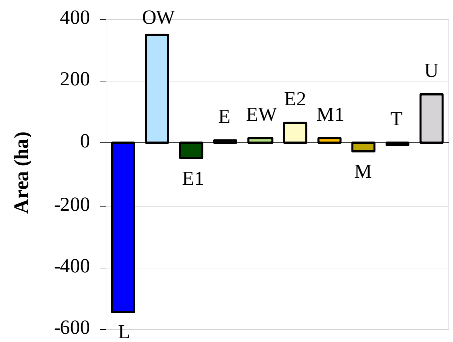


FIGURE 6-5: DIFFERENCE BETWEEN PREDICTED AND ACTUAL AREAS v2.0, 1999

The spatial results of the simulated vegetation for the model were disappointing (Figure 6-8a). There were no real contiguous patches or areas of wetland communities defined, except for a few upland areas on the mainland and in the outer peninsula. The lake area predominantly was simulated as lake, but speckled with cells of open water. Also, within the 1964 results, there is a band of cells classified as various wetland vegetation types in the northeast corner of the lake. Classification errors were noted in Big Creek where vegetated areas were classified as lake and open water, and along the peninsula and outer tip where few meadow and treed communities were predicted. The one positive outcome of the results was the marginal, but observable, differences between the simulated results for rising and declining water level conditions. In 1964 and 1995, the Inner Bay appears to be dominated by drier vegetation communities and for 1978 the area is dominated by open water and wetter emergent vegetation. The results depicting the spatial distribution of error for each year are similar. Lake and open water areas in the Inner and Outer Bay were misclassified within one community. Larger differences were evenly distributed across the entire mainland and peninsula (Figure 6-8b).

6.2.2 VERSION 2.1

The overall accuracy of Version 2.1 (v2.1) was 55.7 %. The accuracy ranged from 54.2 % for 1955 to 58.4 % for 1999. The simulation results for periods of declining water levels (57.0 %) were marginally better than the average results for periods of rising conditions (55.0 %). The model was successful in predicting

76.1 % of the lake cells and 58.0 % of the upland correctly. Moderate success was achieved in simulating cells of open water (35.2 %). The success rates for the other communities were poor. The least accurately simulated community remained short wet meadow, but the accuracy did increase to 2.0 % over v2.0. There were marginal differences in the accuracy of the models for declining and rising water level conditions; the maximum difference between the accuracies was only 4.4 %.

Of the 44.3 % of cells that were incorrectly predicted, 39.9 % were within one community from the actual value. A total of 49.6 % were within one or two communities. Similarly to v2.0, a greater proportion of cells were predicted above the actual value or simulated as a drier wetland community. Though the proportion of cells predicted above and below the actual values was more even (similar) than in v2.0.

There were several observations between the actual and predicted areas for the wetland communities. Lake was consistently underestimated each year; lake area was notably underestimated in 1964, the lowest water level period on record. The model also underestimated the area of tall dense dry emergent vegetation, except for 1978 as water levels rose drastically. Upland area was overestimated each year by the model (Figure 6-6). The percentage difference in area revealed that the amount of short wet meadow was overestimated by 140.2 ha (813.2%) in 1964 and treed was overestimated by 139.2 ha (308.4 %) in 1945. These percentages were notably larger compared to other years.

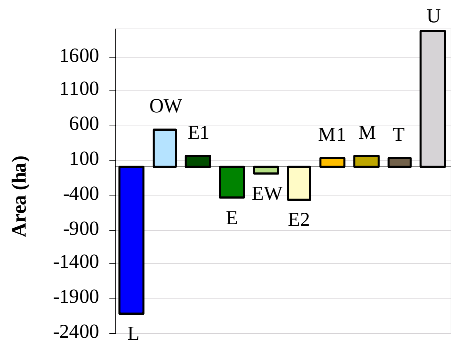


FIGURE 6-6: DIFFERENCE BETWEEN PREDICTED AND ACTUAL AREAS V2.1, 1999

The output maps of the simulated wetland vegetation showed little improvement compared to v2.0 (Figure 6-9a). The results did exhibit less noise particularly in the lake and upland areas, and contiguous areas of lake and upland cells were more apparent. The spatial distributions of the classification errors for v2.1 were similar to the previous version; greater differences between the actual and predicted values were observed in Big Creek and along the peninsula, and a higher concentration of cells in the lake and open water areas were generally one community different (Figure 6-9b).

6.2.3 VERSION 2.2

Version 2.2 (v2.2) was 49.0 % accurate in simulating wetland vegetation response. The accuracy ranged from 37.9 % for 1964 to 58.7 % for 1999. The accuracy results for years of rising water levels (49.5 %) were marginally better than the simulated results for years of declining conditions (48.3 %). The model was successful in predicting a higher percentage of lake and upland cells correctly. Approximately 61.3 % of lake cells and 57.8 % of upland were correctly predicted. The version was moderately successful in predicting open water cells (42.7 %). The model poorly simulated the other wetland communities. The least accurate communities were tall wet emergent and short wet meadow vegetation as 1.1 % of cells in both of these communities were correctly predicted. Generally, the probabilities derived from the 1985 data did a better job at correctly predicting cells for communities that were wetter than the tall dense dry emergent vegetation community for rising water levels conditions. The 1995 probabilities that were used for declining water level conditions were better at predicting drier communities ranging from tall dense dry emergent vegetation to upland.

Of the 51.0 % of cells that were incorrectly predicted, 43.2 % were within one wetland community from the actual value. Greater portions of these cells were within one community predicted below (drier than) the actual value. On average, 8.8 % of the cells differed by two communities, therefore a total of 52.0 % of the cells were predicted within one or two communities from the actual values. A greater proportion of the cells were predicted drier than (above) their actual wetland value.

Examination of the actual and predicted areas of the communities indicated that v2.2 substantially underestimated the amount of lake area, and overestimated the amount of open water and upland communities (Figure 6-7). Generally, the percentage differences between the actual and predicted areas were small each year. Although the model overestimated by the areas of smaller wetland communities in 1964, the lowest water level period on record. Floating emergent vegetation was overestimated by 317.6 ha (140.5 %), short wet meadow by 141.7 ha (822.1 %) and treed vegetation by 203.6 ha (161.9 %).

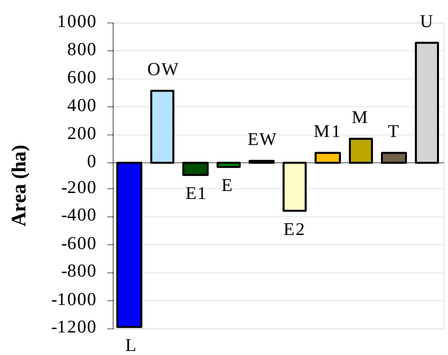


FIGURE 6-7: DIFFERENCE BETWEEN PREDICTED AND ACTUAL AREAS V2.2, 1999

The simulated results of the model were quite interesting (Figure 6-10a). There are definite improvements in the model in simulating contiguous patches of lake and upland areas, but contoured bands of drier wetland vegetation appeared in the lake area. The model was also successful in simulating vegetation drier vegetation communities for periods of declining water level conditions, and simulating wetter communities during periods of rising water levels. The spatial distribution maps of the amount of classification errors are similar between years. Larger classification errors are distributed across the mainland and peninsula but also occur in the lake, coinciding with the contoured bands of drier wetland vegetation simulated by the model. There is also a higher concentration of cells within one community from the actual value in lake and open water communities (Figure 6-10b).

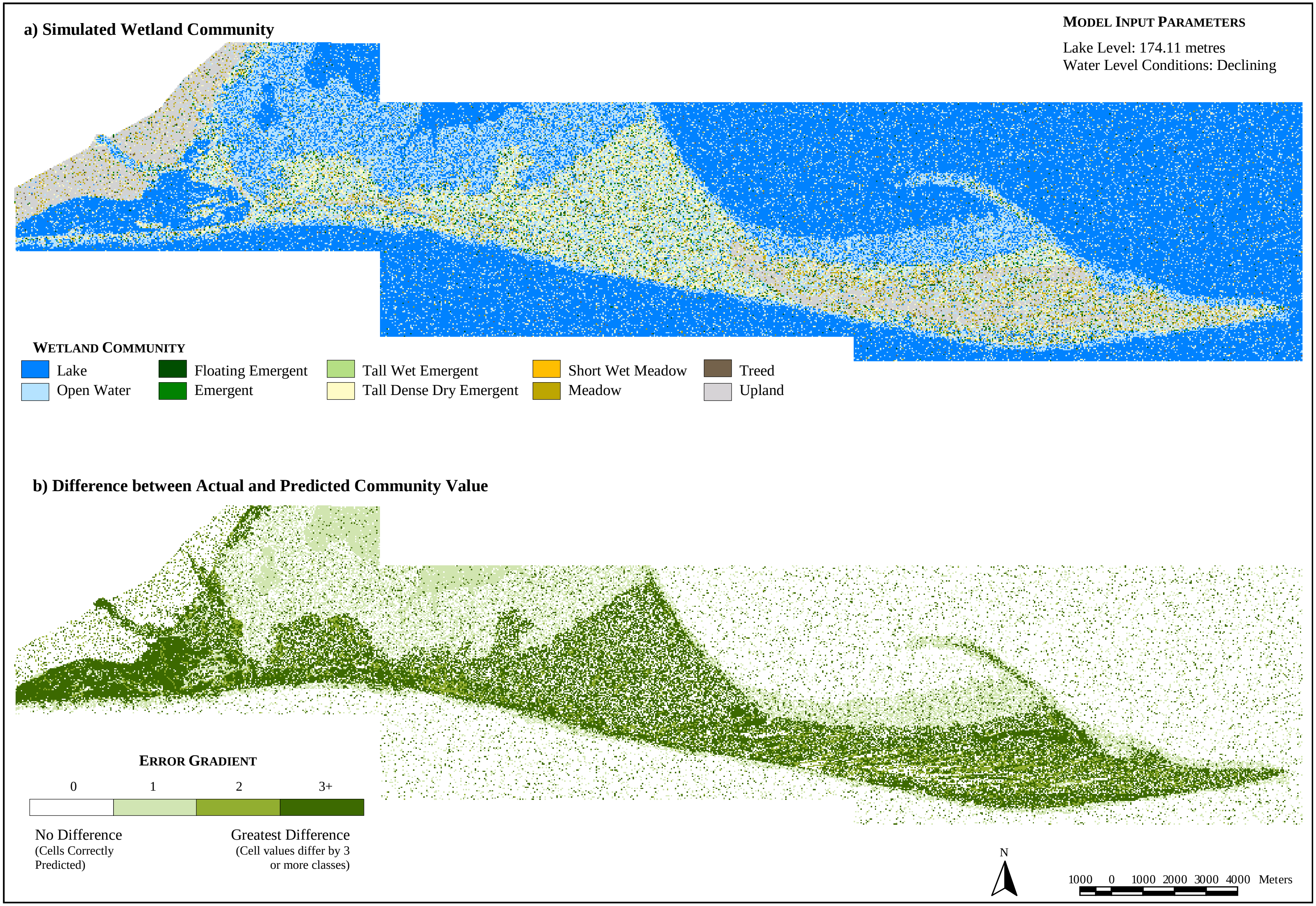


FIGURE 6-8: RESULTS OF THE PROBABILITY MODEL v2.0, 1999

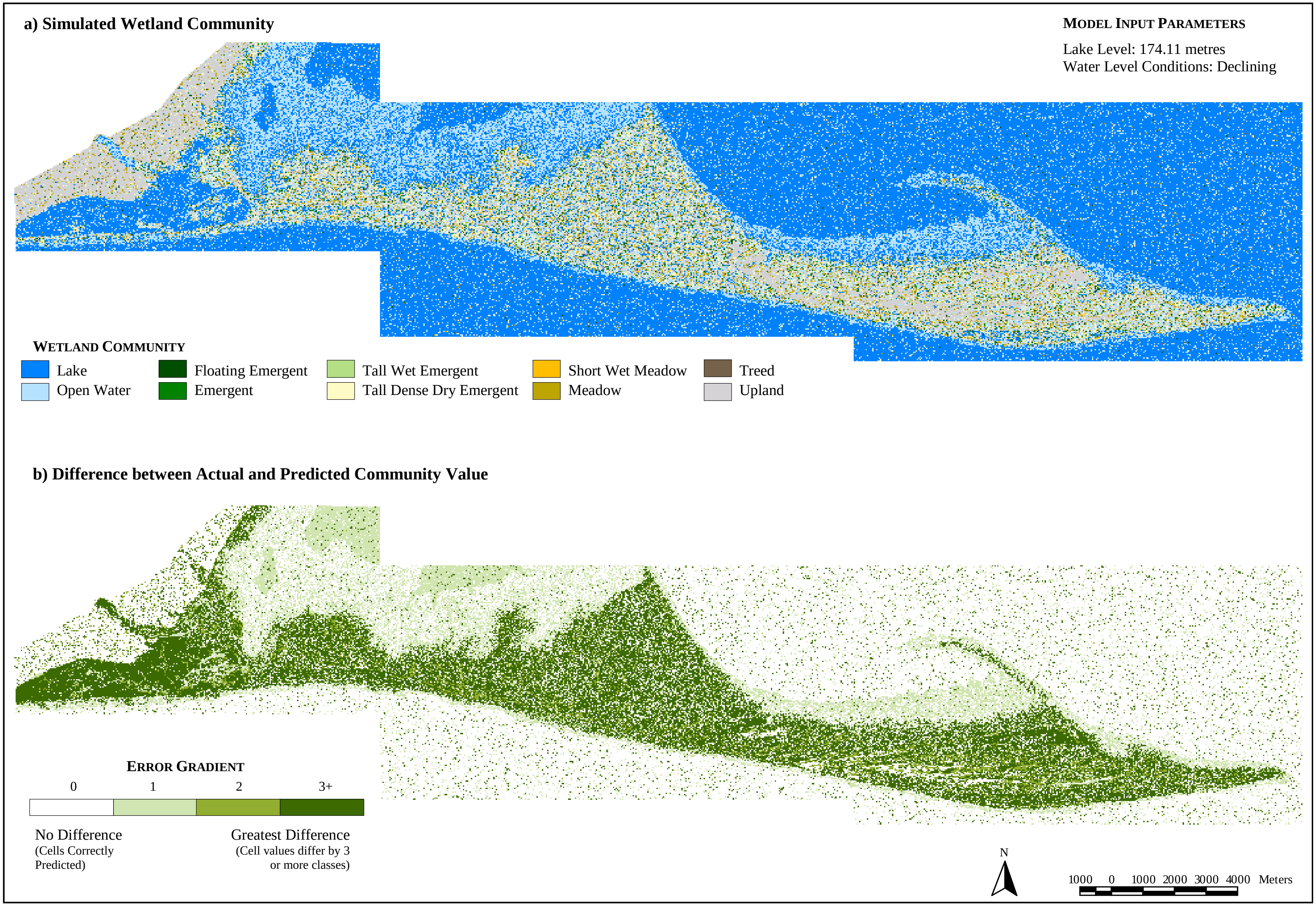


FIGURE 6-9: RESULTS OF THE PROBABILITY MODEL v2.1, 1999

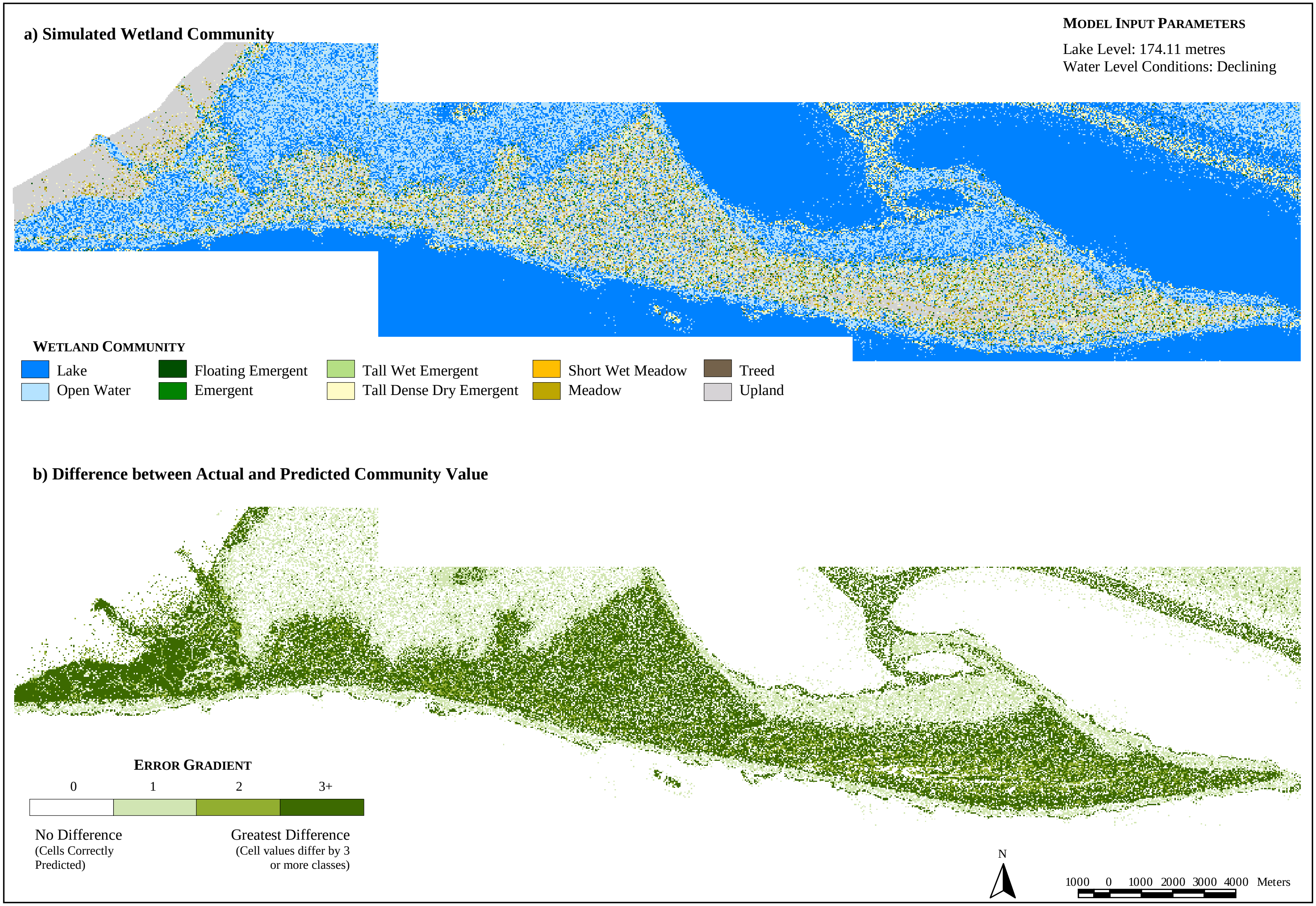


FIGURE 6-10: RESULTS OF THE PROBABILITY MODEL V2.2, 1999

6.2.4 COMPARISON OF VERSIONS

There were few differences between the three versions of the probability model. The overall accuracy of v2.0 and v2.1 were identical; 55.7 % of the cells were correctly predicted within each version. The accuracy of v2.2 decreased to 49.0 %. All three versions simulated lake and upland areas successfully and open water moderately successful. The percentage of correctly predicted cells of lake and upland were marginally higher in v2.1 than in v2.0. The success rate of lake noticeable decreased in v2.2. The percentage of open water cells correctly predicted was greatest in v2.2 (42.7 %), and smallest in v2.1 (35.8 %). The models poorly predicted the distribution of the various emergent and meadow vegetation communities within the wetland.

The amount of classification error improved within each version of the model. The percentage of incorrectly predicted cells misclassified within one wetland community from the actual value increased from 38.4 % in v2.0 to 43.2 % in v2.2. Similar trends were observed in the percentages of incorrectly predicted cells within two communities and more than two communities from the actual values. This indicates the amount of error, or differences between actual and predicted wetland values, are declining with each version. For all versions, a greater proportion of the incorrectly predicted cells were below (wetter than) the actual cell values.

All three versions overestimated the amount of upland area and underestimated lake area. Versions 2.0 and 2.2 also overestimated open water, and Version 2.1 underestimated the area of tall dense dry emergent vegetation. The models also produced greater percentage differences between the actual and predicted areas of small wetland vegetation communities for 1945 and 1964; marginal differences were observed overall for the other years. For 1945, the predicted areas of floating emergent, tall wet emergent, and treed were more than twice the actual areas. In the simulation results for 1964, short wet meadow was significantly overestimated; the smallest difference was 130.1 ha (754.8 %) in v2.0.

The simulated wetland vegetation maps for the versions of the probability model displayed similar patterns, although there were slight improvements within each version with respect to the model's ability to predict solid and contiguous patches of wetland communities. The amount of noise (i.e. speckling) from other communities within lake and upland decreased with each new version of the model and, although the amount of noise within the lake did increase in some areas for the simulated years prior to and including 1978 in v2.2, the amount of noise did decrease significantly in other areas of the lake.

6.2.5 PROBABILITY MODEL ASSESSMENT

The simulation results of the probability modelling technique were disappointing. Patches within the landscape were not clearly defined, and the entire landscape resembled a mosaic impression of Long Point rather than an actual wetland complex. It should be noted that several filtering techniques were tested in

ARC/INFO to remove some of the noise (e.g. FOCALMEAN, FOCALMAJORITY, FILTER), but no noticeable improvements were observed. Furthermore, adding more water depth classes with smaller interval classes in subsequent versions of the model did not improve the results. In fact, the modifications made in v2.2 reduced the overall accuracy of model. The one positive outcome of the model was the ability of the model to simulate different responses of the wetland vegetation communities to declining and rising water level conditions; drier vegetation dominated marsh areas during periods of water level decline, and open water communities dominated the wetland during periods of water level rise. Overall, the model depicted the theoretical response of wetland vegetation communities to water level fluctuations, drier-type vegetation communities with lower water levels and wetter-type vegetation with higher levels.

6.3 VEGETATION TRANSITION MODEL

The vegetation transition model simulated wetland vegetation response based on the likelihood of wetland vegetation communities changing to other wetland vegetation types for periods of declining and rising water levels. The simulation results of the vegetation transition model are presented. The accuracy of the model was determined by averaging the simulated results for four years. In addition, two of those years were averaged to determine the accuracy of the model in predicting vegetation response for declining water level conditions (1964, 1999) and the two other years were averaged for rising water level conditions (1955, 1978). The results for 1985 and 1995 were excluded since these years were used to derive the transitions for the different water level conditions. A discussion of the results follows.

6.3.1 VERSION 3.0

Overall, the vegetation transition model was 83.4 % accurate in simulating wetland vegetation response. The percentage of cells correctly predicted ranged from 82.1 % for 1978 to 85.6 % for 1999. The model was marginally better at predicting vegetation response for declining water level conditions than for rising water level conditions. The average number of cells that were correctly predicted for periods of declining water levels was 84.1 % compared to 82.4 % for periods of rising levels.

The percentage of cells correctly predicted varied according to the wetland community. Overall, the highest percentage of cells correctly predicted occurred for the lake community; 99.1 % of lake cells were correctly predicted. The model was also successful at predicting open water (79.5 %), upland (77.4 %), and tall dense dry emergent vegetation (54.9 %). The model was moderately successful at predicting meadow and treed vegetation; the percentages of cells correctly predicted for these communities were 22.9 % and 22.6 %, respectively. The model was least accurate overall at predicting emergent (5.3 %) and tall wet emergent vegetation (4.4 %). Generally, the transition probabilities representative of declining water level conditions were more accurate at predicting the individual wetland communities correctly. However, the transition probabilities for rising conditions were better at predicting open water and tall wet emergent vegetation communities. The average percentage of cells correctly predicted for these communities were

7.0 % and 6.8 % greater than the percentage predicted for periods of declining water levels. Whereas, the percentage of cells correctly for the other communities ranged from 0.7 % (lake) to 27.6 % (treed vegetation) less for rising water level conditions

Of the 16.6 % of cells that were incorrectly classified, an average of 19.0 % of the cells were within one wetland community class from the actual value. Another 30.1 % were different by two communities. Therefore, a total of 49.1 % of cells were within one or two communities from the actual wetland community class. A greater number of incorrectly predicted cells were drier than (above) the actual wetland community value.

There were no consistent trends in the percentage differences between the actual and predicted areas of the wetland communities. Although, open water was generally underestimated and tall dense dry emergent vegetation was overestimated for all simulated years except for 1955 (Figure 6-11). There were also greater differences between the predicted and actual areas for the wetland communities reported for 1964. Furthermore, when examining the percentage differences between actual and predicted areas for 1964, the amounts of short wet meadow, treed, and floating emergent vegetation were largely overestimated. These actual areas of these communities were generally smaller compared to others, thus moderate differences between the actual and predicted values result in larger percent differences.

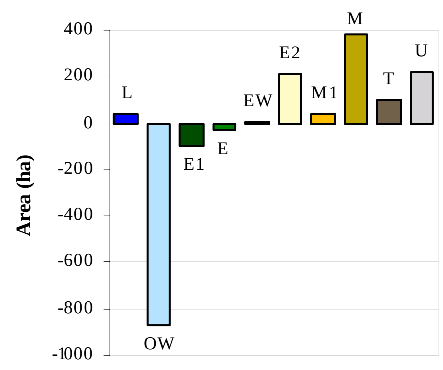


FIGURE 6-11: DIFFERENCE BETWEEN PREDICTED AND ACTUAL AREAS V3.0, 1999

The simulated wetland community distributions of the model were better than expected (Figure 6-12a). The lake area was well defined and there were few occurrences of other wetland vegetation cells within the lake. The majority of the remaining wetland communities were delineated with satisfactory results. Patches within the landscape were delineated successfully, but the patches were dotted with cells classified as other communities. Patches of emergent and floating emergent vegetation are present but occur in smaller and less significant patches, especially for years when water levels have been declining.

The model was also successful is simulating the pattern of change during rising and declining water level conditions. The simulated results show that for years of declining water level conditions, drier vegetation communities of tall dense dry emergent and meadow were dominant within the wetland, and for years of rising water levels, wetter open water and emergent communities dominated vegetated areas. This pattern of community response is what should theoretically occur during declining and rising water level conditions. The spatial distribution of errors exhibited similar trends for all simulated years (Figure 6-12b).

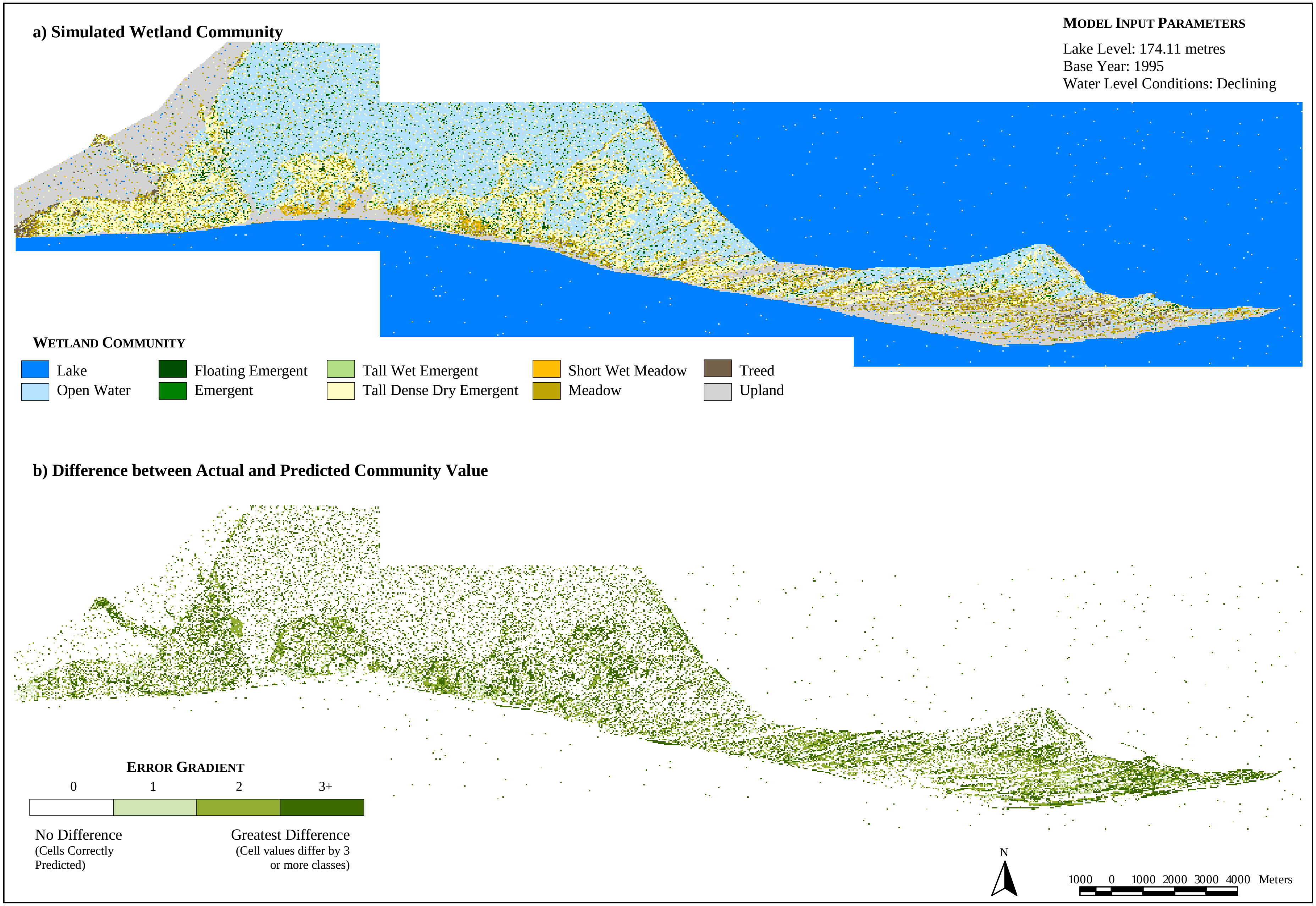


Figure 6-12: Results of the Vegetation Transition Model v3.0, 1999

Higher concentrations of greater errors were evident in the outer peninsula, in the Crown and Company Marshes, Long Point NWAs, and along the shoreline. This trend was particularly prominent during periods of rising water levels.

6.3.2 VEGETATION TRANSITION MODEL ASSESSMENT

The simulation results of the vegetation transition modelling technique were reasonable. The model was able to successfully define the more dominant vegetation communities in the landscape. The model was able to delineate patches of contiguous cells of lake, open water, tall dense dry emergent, and upland communities well. The model successfully simulated the response of the wetland during periods of declining and rising water level conditions; during declining water levels, the communities tended to drier vegetation, and during rising water levels, the wetland vegetation tended towards wetter communities. The differences between the actual and predicted areas of the wetland communities were minimal relative to the results from the other modelling techniques, although the predicted areas for 1964 were noticeably different from the actual areas.

The results of the model were unsatisfactory in two regards. Although the larger wetland communities were well-defined, the model did not define the extent of smaller wetland vegetation communities within the landscape well. It was often hard to delineate these communities visually in the model output. The differences between actual and predicted wetland vegetation community values were also substandard. Only 19.0 % of the incorrectly predicted cells were within one wetland vegetation community from the actual value; a greater percentage of cells had more error between the predicted and actual community values.

6.4 MODEL ASSESSMENT

This section provides an overall assessment of the wetland vegetation simulation modelling efforts for Long Point. The simulation results from the different modelling techniques are compared and the overall modelling efforts for Long Point are discussed.

6.4.1 COMPARISON OF MODELS

Of the three different models that were implemented to simulate wetland vegetation response, the vegetation transition model (v3.0) outperformed the rule-based and probability models. The transition probability model had the highest overall accuracy in predicting cells correctly and a greater number of the individual wetland communities had percentages of correctly predicted cells with a good success rating (i.e. greater than 50 % accuracy); these communities were lake, open water, tall dense dry emergent vegetation, and upland. The model also produced smaller differences between the aggregate totals of the predicted and actual areas of the wetland communities. The wetland communities within the landscape were generally well defined. The extent of smaller patches and some features in the study area, however, were hard to distinguish because of noise in the simulated results; applying filtering techniques, such as those applied to the probability models, may eliminate some of the noise produced by the transition model. The accuracy of the model does need improvement with regard to the amount of error between the actual and predicted community values; there was a smaller percentage of incorrectly predicted cells within one or two community values above or below the actual value compared to the other models.

The simulations from v1.1 of the rule-based model are quite good and comparable to the vegetation transition model. The overall accuracy was marginally less than the accuracy of v3.0. The model was successful at predicting three wetland communities with good rating (lake, open water, upland), and a greater number of communities with moderate rating compared to v3.0. Furthermore, the minimum level of accuracy for any of the wetland vegetation communities in v1.1 was 11.2 % for short wet meadow, which is notably better than the minimum level of accuracy in the vegetation transition model and probability models. The results of the rule-based model were superior visually to both the vegetation transition and probability models. The spatial distributions simulated by the model were more realistic and matched the patterns of a real wetland system compared to the results of the vegetation transition model and probability models. The model was able to depict features within the landscape, such as linear channels of water and islands, with great success. Patches of all sizes were delineated successfully and easily identifiable on the simulated output. Although, the output was visually more realistic, there were obvious errors in the spatial distribution of the wetland communities predicted by the model, especially during periods of extremely low water levels. In 1964, much of open water in the Company Marsh was depicted as floating emergent or emergent vegetation and during periods of extremely high water levels, upland areas along the shore were washed out by the lake; modifications to the decision rules may eliminate some of these problems. The model also produced larger differences between the aggregate totals of the predicted and actual areas of the wetland communities compared to the vegetation transition model.

Version 1.0 of the rule-based model produced satisfactory results. The overall accuracy of the model was significantly less compared to v1.1 and the vegetation transition model v3.0. The lake and upland communities were predicted with good success and three communities (open water, floating emergent, emergent) were predicted with moderate success. The percentages of correctly predicted floating emergent and emergent vegetation cells were slightly better than the percentages in v1.1; the percentages were around seven percent higher in v1.0. Visually, there were also a greater number of cells misclassified along the shoreline and in the Inner Bay compared to v1.1. There were two redeeming qualities of this version of the model. There was a smaller range in the minimum and maximum percentages of correctly predicted cells for the individual wetland communities. Although the most successful community (lake) was predicted will less success compared to the other models, the least successful community (tall dense dry emergent vegetation) was predicted with greater success compared to the other models. In addition, v1.0

also had the highest number of incorrect cells within one and two communities from the actual values. The measurement can be a bit misleading since a significant portion of lake area along the shore was misclassified as open water, and large areas of open water in the Inner Bay were misclassified as lake.

The probability models produced the worst results. Versions 2.0 and 2.1 performed almost identically and v2.2 performed the poorest. The probability models had the lowest overall accuracy in predicting communities correctly, had the fewest number of communities that were predicted with good and moderate success, and had the lowest percentage of cells correctly predicted for the community with the poorest success. The models did perform well in minimizing the amount of error between the actual and predicted wetland community values. The percentages of incorrectly predicted cells within one or two communities from the actual values reported by the models were higher compared with those values reported by the vegetation transition model. There was also a greater range in the differences between the actual and predicted aggregate areas reported by the models compared to the other models. Generally, the differences were smaller for the results from 1978 and 1999 and larger for 1945, 1955, and 1964. The smaller differences are comparable to those reported overall by the vegetation transition model, but the larger differences are similar to those reported by the rule-based models. Visually, the spatial distributions of the wetland communities in the probability models were the least comparable with the aerial photography-interpreted maps; the models contained a lot of noise and contiguous areas of communities were not well defined. A summary of the key results of the models is provided in Table 6-1.

6.4.2 DISCUSSION

The results of the modelling efforts for Long Point were somewhat disappointing, although the models were effective at simulating the response tendencies of the wetland communities to water level fluctuations. During periods of declining water levels, the vegetation tended towards drier communities and during periods of rising water levels, the vegetation tended towards wetter communities. The models were also effective at simulating the spatial distribution and extent of the lake, open water, and upland communities, but the models could not predict the distribution of more important wetland vegetation communities with a high degree of accuracy. The wetter and drier emergent and meadow vegetation communities were often the least accurately predicted wetland communities by the models. There are several reasons why this may have occurred. First, the lake, open water, and upland communities may be more likely to respond to fluctuating water level conditions and their response is mainly influenced by the topography in the wetland. Alternatively, the other communities are too small in area relative to the other communities, and thus it is hard to characterize and model the response of these communities strictly according to water level fluctuations and topography. Or perhaps even, the errors in the topography model did have some negative impact on the performance of the models as wetland vegetation may have been incorrectly classified according the erroneous topographic information in the modelling process.

TABLE 6-1: SUMMARY OF THE WETLAND SIMULATION MODELS

MODEL	CONSIDERATIONS	PERCENT (%) OF CELLS CORRECTLY PREDICTED: OVERALL	BY COMMUNITY **: GOOD	BY COMMUNITY **: MODERATE	BY COMMUNITY **: POOR	AMOUNT OF ERROR*	OBSERVATIONS
RULE-BASED MODEL							
v1.0	• Existing wetland vegetation • Strict water depth tolerance ranges • Cell adjacency between lake and upland	65.9 %	L: 84.7 % U: 68.7 %	OW: 47.8 % E1: 29.4 % E: 29.2 %	E2: 13.7 %	1: 68.6 % 2: 87.3 %	• Overestimates area of E1, E, T • Underestimates area of E2, U • Misclassifications along shore of peninsula and Inner Bay • Horizontal edges of communities in outer peninsula for wet years • Simulates wetter emergent vegetation for marshes in Big Creek and Inner Bay • Good definition of patches and features in landscape
v1.1	• Existing wetland vegetation • Lenient water depth tolerance ranges • Cell adjacency between lake, open water and upland	81.8 %	L: 98.5 % OW: 84.0 % U: 63.8 %	M: 29.6 % E2: 25.3 % E: 22.3 % E1: 22.2 % EW: 20.3 %	T: 11.3 % M1: 11.2 %	1: 39.9 % 2: 71.5 %	• Overestimates area of E, T • Underestimates area of E2, U • Shoreline areas and deeper sections of Inner Bay correctly classified • Landward areas along the shore washed out in wet years • Horizontal transitions between communities in outer peninsula • Simulates drier emergent vegetation for marshes in Big Creek and Inner Bay • Good definition of patches and features in landscape
PROBABILITY MODEL							
v2.0	• Water depth ranges • Ranges from > 200 cm in depth to > 100 above lake level • Depth intervals based on tolerance ranges	55.7 %	L: 75.1 % U: 56.9 %	OW: 37.8 %	M1: 1.0 %	1: 38.4 % 2: 48.5 %	• Poor definition of features and patches in landscape; significant noise • Few contiguous patches of similar communities, except for L and U • Overestimates area of OW, U • Underestimates area of L • Band in northeast corner of lake in 1964 results
v2.1	• Water depth ranges • Ranges from > 200 cm in depth to > 100 cm above lake level • Depth intervals of 10 cm	55.7 %	L: 76.1 % U: 58.0 %	OW: 35.8 %	M1: 2.0 %	1: 39.9 % 2: 49.6 %	• Marginally less noise in L and U • Overestimates area of U • Underestimates area of L, E2 • Band in northeast corner of lake in 1964 results
v2.2	• Water depth ranges • Ranges from > 320 cm in depth to > 1200 cm above lake level • Depth intervals of 10 cm	49.0 %	L: 61.3 % U: 57.8 %	OW: 42.7 %	EW: 1.1 % M1: 1.1 %	1: 43.2 % 2: 52.0 %	• Significantly less noise in L and U communities • Overestimates area of OW, U • Underestimates area of L • Larger contiguous areas of L and U • Contour bands of vegetated communities in lake
VEGETATION TRANSITION MODEL							
v3.0	• Existing wetland vegetation	83.4 %	L: 99.1 % OW: 79.5 % U: 77.4 % E2: 54.9 %	M: 22.9 % T: 22.6 %	E: 5.3 % EW: 4.4 %	1: 19.0 % 2: 49.1 %	• Least amount of difference between actual and predicted aggregate areas • Reduction in noise (compared to probability model) • Larger contiguous areas defined well • Fair definition of smaller patches and some features in landscape

* Amount of Error is the total percentage of incorrectly predicted cells within one or two communities from the actual value

** The percent of cells correctly predicted by community are listed according to good success (over 50 % of the cells were correctly predicted), moderate success (20 to 50 % correctly predicted) and poor success (< 20 % correctly predicted; only the least accurate community or communities are listed

L = Lake; OW = Open Water; E1 = Floating Emergent; E = Emergent; EW = Tall Wet Emergent; E2 = Tall Dense Dry Emergent; M1= Short Wet Meadow; M = Meadow; T = Treed; U = Upland

In addition to the performance of the models, there were several assumptions and caveats within the modelling analyses that should be addressed. The inherent structure of the modelling techniques prohibited the models ability to simulate or account for one-time events in the wetland. During the period of study from 1945 to 1999, there have been several notable changes in the landscape related to both natural and anthropogenic events. The impacts of intense storm systems, such as Hurricane Hazel in the fall of 1954 and other storms in 1975 and 1985, on the vegetation and physical structure of the wetland system were evident on the aerial photographs and the interpreted maps, but simulating these impacts on the wetland communities was not possible within the wetland models. In addition, the models could not simulate the construction of the NWA dike in Big Creek in 1985, changes in land use patterns such as the creation of the Long Point Provincial Park in 1956 or new roads and built-up areas, fires that have occurred along the peninsula, or the expansion of invasive species in the wetland. Furthermore, the structure of the models limited simulating wetland community response dynamically through time. All three models were static models that simulated wetland community change from one year to another year with no regard to the number of years between the base and output years. There was no consideration of inter-annual variability in water levels between the two years, the response times of the wetland vegetation to the changes, or seiche effects.

The tolerance ranges established for the decision rule models were based on the tolerance ranges of vegetation communities along the wetland continuum of a typical marsh. Therefore, the models assumed that the physical environment within the wetland was uniform. In fact, the Long Point wetland is quite complex with many different landforms including dune and swale, lagoon and barrier, delta and riverine, and embayed and shoreline wetlands (Bolsenga and Herdendorf, 1993). These geomorphic forms, along with soil or substrate type, will influence how the wetland vegetation responds to water level fluctuations. For example, the Big Creek Marshes are separated from the lake by the causeway and thus are not directly influenced by water level fluctuations in the lake. The marshes in the Big Creek delta are greatly influenced by seiches and rainfall events, while water levels in the Big Creek NWA dike are controlled. As a result, the vegetation in these areas will respond differently than the vegetation in the Inner and Outer Bay of the peninsula. Consequently, the models could not simulate the wetland vegetation communities within these areas successfully. The tolerance ranges also assume that there are strict boundaries lines that mark the transitions between these communities, although some of this problem is addressed in the modifications to v1.1. Version 1.0 strictly adhered to the tolerance range limits of the pre-existing vegetation communities in relation to changes in the water depth of the cell caused by rising and declining water levels, but in v1.1 the tolerance range limits were not as strict.

The models are simple and were developed with few input variables. The rule-based and probability models only considered the water depth of the wetland communities in relation to the topography. The vegetation transition model was based on transitions that occurred over time for representative periods of declining and rising water levels, and thus the transitions for rising and declining conditions assume that those transitions occurred in relation to water level fluctuations only and that these transitions are constant over time; other land use changes and processes of change may be accounted for in the transitions inadvertently. It is unrealistic to assume that wetland vegetation changes are only influenced by water level changes. There are other important factors that influence vegetation response such as the frequency, timing, and duration of the water level fluctuations and the geomorphic landforms in the wetland, and the development of the vegetation within the wetland communities, which is influenced by the slope and substrate of the wetland, the reservoir of buried seeds in the seed bank, wave action, and the chemistry of the soil and water. Furthermore, the distribution of the vegetation communities is not based on environmental constraints alone. "Given the same climatic and edaphic conditions, different vegetation states are possible, and the present state of any particular patch may be the result of its particular history of response to climactic events, disturbances and/or management" (Hobbs, 1994, 347). Other researchers have noted the importance including more variables.

It is also acknowledged that additional statistical methods could have been used to measure the accuracy of the models. For example, simulated spatial patterns in the wetland could have been compared with actual patterns using statistical descriptors of landscape structure and composition. A number of metrics used in the temporal trend analysis could have been computed for the simulated wetland distributions and compared with the actual wetland distribution (Turner, 1987). This was the preferred method of validation here, but given the poorly simulated spatial distributions of the wetland communities in the probability models, it would have been futile. Furthermore, analysis of variance could be performed on these descriptors or on the percentage differences between the actual and predicted wetland community areas to provide an indication of the significance of the differences (Payn *et al.*, 1999; Poiani and Johnson, 1993a). The methods implemented here to assess the performance of the models were used to provide a simple and basic means of comparing the models. These methods may not have been optimal for comparing the simulated and observed wetland values, especially for the probability models. The probability and vegetation transition models involve some degree of randomness, which was inherent in the models' design. Therefore, the models are not or can never be 100 % accurate in simulating the landscape. Some of this problem was addressed, however, by looking at the aggregate sums of the actual and predicted areas for the wetland vegetation communities. Nevertheless, the randomness in these models may have had an impact on some of the accuracy measurements.

It is difficult to make any effective comparisons of the modelling results for Long Point to similar studies. There have been numerous wetland models developed to assess wetland response to climate change with greater accuracy in simulating vegetation response to change, but these models were developed for wetlands of smaller spatial extent and did not include as many wetland vegetation communities in the simulations. In addition, these models were often more realistic in simulating response, as many of the models incorporated more environmental variables that influence wetland vegetation response to water level fluctuations. Despite these limitations, there was one comparison of note. Turner (1987) observed that random simulations based solely on transition probabilities resulted in a highly fragmented landscape with many small patches of complex shapes. Although the wetland vegetation communities simulated in the vegetation transition model were well-defined, a highly fragmented pattern was observed in the wetland landscapes simulated by the probability models.

The models developed for this analysis were mainly developed as an investigative comparison to determine the applicability of the different modelling techniques for simulating wetland vegetation response at Long Point. Before any of the models developed in this analysis can be applied to simulate wetland vegetation response to projected climate change and water level changes resulting from enhanced global warming, it is imperative that the accuracy of the models improves, especially with regards to simulating the various emergent and meadow vegetation communities within the wetland. Future research efforts should focus on methods of improving the results of the models. The decision rule-based model and vegetation transition models should be explored further as these models hold the most promise for future modelling efforts.

6.5 SUMMARY

This chapter summarized the results of the modelling techniques that were implemented to simulate wetland vegetation response in Long Point. The performance of each individual model was reviewed and compared with the other models to determine the optimal method of wetland modelling. Although none of the models predicted wetland vegetation communities with a high level of accuracy, the vegetation transition model produced the best overall results followed closely by the decision rule-based model. Further limitations related to the development of the models and the models ability to accurately simulate wetland vegetation communities are discussed in the concluding chapter. In addition, recommendations for further modelling efforts are presented.

7.0 CONCLUSIONS AND RECOMMENDATIONS

Natural climate variability and associated water level fluctuations within the Great Lakes have been instrumental in the physical and ecological evolution of the Long Point wetland complex. Long Point contains a diversity of wetland vegetation communities that have developed and adapted to the fluctuating water levels in Lake Erie. The spatiotemporal trend analysis indicated that the structure and composition of these wetland vegetation communities have changed significantly during the period of study from 1945 to 1999, and much of this change can be related to historical climatic conditions affecting water level fluctuations in the lake. The historical response of the wetland vegetation also provided a key to how the wetland may respond to future climatic changes from enhanced global warming. Lake Erie water levels are projected to decline due to human-induced climate change, and as a result there may be serious implications on the composition and configuration of the Long Point wetland complex and consequently lead to a decline in the natural diversity and integrity of the wetland. The modelling analysis attempted to simulate historical wetland vegetation community response to water level fluctuations, in hopes that the models could be used to simulate wetland response to future climatic changes. The models were moderately successful in simulating wetland vegetation response to historical water level fluctuations but the accuracy of the models need to improve before the models can be used to simulate future response with greater confidence.

7.1 LIMITATIONS OF THE SPATIOTEMPORAL AND MODELLING ANALYSES

There were a number of constraints related to the spatiotemporal and modelling analyses procedures. First and foremost, was the availability of wetland classification data. There were only seven years of historical wetland community data for Long Point available in digital form that could be included in the analyses. These years of data often spanned significant lengths of time and occurred over irregular intervals. More closely spaced and regular interval wetland data for the analysis would have been ideal so that greater detailed observations regarding wetland vegetation response to climate change could be made. The availability of digital data, however, is limited by the availability of the historical aerial photography of the region. All the years of existing aerial photography for Long Point were interpreted and classified on mylar maps. There were two more years of mylar classification maps that were available for the analyses, 1968 and 1972, but time constraints prevented vectorizing these additional years of data into the geographic information system. In addition, the few years of digital data greatly influenced the type of modelling techniques that could be implemented and the number of years that could be used to derive and validate the models.

The availability of additional spatial data related to biophysical and environmental conditions in the wetland was limited. The response of the vegetation communities to water level fluctuations is partly influenced by slope, soil or substrate type, seed banks, wave action, and water chemistry of the wetland. Many existing wetland models have incorporated a number of different environmental factors to simulate vegetation response to changing water levels (Ellison and Bedford, 1995; Park *et al*., 1993; Nilsson and Keddy, 1988), and the inclusion of some these factors as additional datasets or parameters in the models could potentially enhance the simulation results. The wetland vegetation simulation models predict the amount and distribution of vegetation communities according to water depth and/or pre-existing wetland communities and thus are extreme simplifications of the processes that affect change within a wetland system, but the time, money, and effort in acquiring and incorporating these additional information in the models would be substantial. For example, soil data layers were available for the Long Point region. If the soil data layers were incorporated into the model, the information contained in the layers would have to be analyzed to determine the most useful attributes influencing vegetation response to water level fluctuations; this would have been hard since little information regarding the attributes were available. Further knowledge would also be required to determine how these attributes actually influence vegetation response in the wetland and how the information could be incorporated into the modelling analysis. Other datasets related to seed bank and water quality information would have to be developed.

Another limitation related to the pre-processing of the digital wetland community data. It is acknowledged that the large amount of lake area included in the wetland study area greatly influenced the results of the spatiotemporal trend analysis and performance of the models. Although, the results of the temporal trend analysis of the wetland were similar for the landscapes that included and excluded the lake area, there were notable differences in the magnitude and range for several of the statistics, especially those metrics that involved using area measurements in their derivation. Furthermore, with regards to the modelling procedure, the lake community was the most successful correctly predicted community in the wetland. The large area of lake, combined with a high accuracy of the models in predicting lake cells correctly, skewed the overall accuracy results of the models. The large spatial extent of the Long Point wetland complex was another limitation related to the spatiotemporal and modelling analyses. The wetland complex is quite diverse and includes many different geomorphic landforms that influence how the wetland vegetation communities respond to water level fluctuations. Although the temporal trend analysis and models were applied to the entire wetland complex, the patterns of change and validity of the models varies by location within Long Point.

Issues related to the accuracy of the topographic model were problematic in the modelling analysis. The validity of the digital elevation model provided by OMNR is a major concern, and most likely affected the results of the simulated wetland communities in Long Point. The projection of the elevation model was questionable to begin with, as one section of the model was misaligned or offset from its real-world location, but there may be other areas of concern, such as the elevation data for the Big Creek area. According to the elevation data the area is immersed in deep water but this is inconsistent with OBMs of the area. In addition, elevations in the outer peninsula are questionable; there are horizontal elevation contours in some sections, which appear to be a bit unnatural. There were also gaps and/or inconsistencies between the elevation, bathymetry, and OBM spot data. This degree of uncertainty can affect the performance of the models that use water depths derived from topographic data to simulate wetland communities. It should be noted that once an accurate topographic model of the area is developed, and at a smaller resolution, accurate slope information could be derived, which in turn could then be incorporated into the modelling analysis.

Finally, it is important to note that there are several constraints related to the applicability of the modelling techniques developed here for future modelling efforts. For the rule-based model, the decision rules applied in the model were based on the theoretical framework of the wetland vegetation continuum of a typical marsh wetland, and therefore are only applicable to wetlands with characteristic marsh vegetation communities. It is assumed that the framework of this model would suit other coastal wetland systems with marsh vegetation, but the validity of the framework may only be applicable within wetlands of particular geomorphic forms. For other wetland systems, such as fens, bogs, and swamps, a different set of rules would have to be established. With regard to the vegetation transition model, the transition probabilities applied in this model were derived from inter-community changes that occurred between two successive years of data with gaps of many years. If an identical method is to be applied to other wetlands, historical data are needed to derive the transition probabilities for the model. The model also assumes that the transitions between these two successive years are constant and that the transitions are only driven by a single factor. However, both of these assumptions are unrealistic when considering the complexity of change within a wetland system. Similarly, the probabilities related to water depth ranges applied in the probability model would also have to be derived from historical data. Recommendations that address the aforementioned constraints are discussed in the following section.

7.2 RECOMMENDATIONS FOR FURTHER WORK

There are several recommendations for improving the results of the spatiotemporal trend and modelling analyses. The first recommendation would relate to the concept of less is more. The wetland classification used for the spatiotemporal trend and modelling analyses incorporated ten wetland communities including lake and upland and eight vegetation communities. Perhaps the wetland classification scheme should have been further simplified with a fewer number of wetland vegetation communities, and these communities could be representative of the main wetland vegetation communities: submergent and floating-leaved vegetation, emergent vegetation, meadow, and, treed. The spatial extent of the wetland of Long Point is large and the landscape itself it quite complex. Therefore, many of the different factors and processes that are occurring in various sections throughout the entire wetland complex drive vegetation response within these various sections of the wetland. Perhaps, the analysis should have focused on a smaller study area within Long Point such as the Inner Bay where the wetland vegetation is typical of a marsh wetland and the topography and landscape of the marsh is not so complex.

Future work should also attempt to incorporate additional environmental variables that influence wetland vegetation response to climatic changes and fluctuating water levels. Further statistical analysis should be performed to identify the relationships between the amount and distribution of the wetland vegetation to the other environmental variables in the landscape. Pearson correlation coefficients were derived to assess the relationships of the wetland communities to water level fluctuations. Similar coefficients could be derived for other important environmental variables and incorporated in a multiple step-wise regression analysis to determine the most important variables influencing change within the wetland communities. In addition, other analysis such as Chi-square tests of independence could be used to identify the relationships. These relationships could help the discussion of change within the wetland system. Future model development should also attempt to incorporate some of these important variables, such as soil and slope, into the models to produce more accurate and realistic simulations. Wetland vegetation response is greatly influenced by water level fluctuations, but other factors that affect how the vegetation will respond should be included in the model. It is also imperative that the integrity of these additional datasets are maintained at a standard level of accuracy. The models developed here do not consider many environmental or climatic conditions that influence wetland vegetation response including substrate or soil type, slope, the frequency, duration, and timing or variability of water level fluctuations, or human influences and modifications in the wetland; although some land use changes were inadvertently incorporated into the transition probabilities of the vegetation transition model. Inclusion of these variables may improve the overall performance of the wetland vegetation simulation models.

The accuracy of the models in simulating wetland vegetation community response could be improved by making several modifications to the existing models. The existing probability models could be deconstructed, in a sense, to produce simple conceptual models of the wetland processes. For example, the probabilities for the probability and transition probability models were derived empirically. A synthesis of existing literature and knowledge of how the wetland vegetation typically responds to water level fluctuations could produce theoretical or hypothetical probabilities. Mathematically deriving the probabilities based on the existing data produced a variety of possible transitions between all vegetation communities. Creating theoretical probabilities would remove less likely probabilities of change between certain communities and thereby eliminate the noise in the probability models. Alternatively, some of the noise in the probability models could be filtered out by removing smaller and less significant probabilities of change derived empirically, and then stretching the major probabilities of change within these communities along a distribution curve. The performance of the models using the theoretical probabilities could also be assessed and compared to the results from the empirical probabilities.

The results of the transition probability model could be improved by considering neighbourhood effects or dynamic transition probabilities. Changes within a cell are influenced by changes in the cells surrounding the particular cell. Turner (1987) calculated transition indices by applying a spatially influenced algorithm that considered the influence of four and eight neighbouring cells. A similar technique could be applied to the transitions in the Long Point wetland to account for the influence of neighbouring cells. Furthermore, transition probabilities are not constant from one period to the next. A thorough review of existing modelling literature may determine an appropriate method of accounting for the dynamic nature of change in the transition probabilities. It would also be worthwhile to explore the potential of incorporating an element of time within all the models. The existing models are static not dynamic. Intermediate changes between a base year and output year are not generated. By considering these intermediate changes, lag time responses of the vegetation to water level fluctuations could be modelled.

Besides modifying the existing models, there are several additional modelling techniques that could be explored. If other spatial datasets were incorporated into the modelling analysis, other types of modelling approaches could be used. Linear regression modelling is one such alternative. Linear regression analysis determines important variables to wetland occurrence and then uses those variables to help predict the probability of wetland vegetation occurring within a particular cell within the wetland. Correlation coefficients of these variables are applied to a regression equation or algorithm, which can then be used to predict the probability of a wetland community occurring. The main limitation with regression analysis is that individual probability maps would have to be developed for each individual wetland community. But the individual probability maps could be combined together by taking the vegetation community with the highest probability of occurrence for each cell in the model to produce a single output map depicting the most probable distribution of wetland response.

Spatial autocorrelation modelling is another alternative modelling approach. A spatial autocorrelation analysis could be performed on the wetland community data to determine the correlation of the wetland vegetation to the community's spatial location within the landscape. Moran's I or Geary's C values could be used to assess whether the locations of certain wetland vegetation communities are located close to similar community types or whether the vegetation communities are located close to dissimilar community types. These values could then be incorporated into some sort of spatial autocorrelation model. The validity of the model's usefulness in simulating wetland vegetation response would have to be tested and methods of incorporating the spatial autocorrelation information into a model will have to be determined. At this stage it is unclear if this approach would be applicable to modelling the wetland community response at Long Point, but it would be interesting to try nevertheless.

Future model validation could include a comparison of landscape and class metrics to determine how well the models simulate structural patterns within the wetland. Simple metrics related to area, patch, edge, and shape could be derived from the simulated results of the models and compared to observed values. More complex metrics related to diversity, nearest-neighbour, and contagion and interspersion could also be compared to provide further indication of the models performance in stimulating the composition and configuration of the wetland. As noted earlier, this method of validation was originally preferred to validate the models here, but given the poor performance of some of the models the technique was not implemented. The simulated and real wetland landscapes could be overlaid to produce a contingency table that summarizes the performance of the models. The contingency table would identify the number of cells in the simulated landscapes that were correctly predicted and those that were not. Finally, statistical analysis, including ANOVA, could be used to provide an indication of how well the simulated spatial distributions compare to the actual distributions or the significance of the differences between the actual and simulated wetland distributions.

Although, there were several constraints identified with the spatiotemporal trend and modelling analyses of Long Point, the analyses have demonstrated the usefulness of geographic information systems in wetland studies by providing a practical examination of the applications of geographic information systems for analysing and modelling the impacts of climate change on the coastal wetland complex at Long Point, both on a temporal and spatial scale. By overcoming the limitations identified in the spatiotemporal and modelling analyses procedures, there is great potential to improve the practicality of the wetland models for simulating future wetland vegetation response to projected water level changes resulting from enhanced global warming. Furthermore, the models could then be applied to other coastal wetland systems within the Great Lakes. The spatiotemporal trend analysis and modelling efforts presented here increases the awareness of the capabilities, and advances the role, of geographic information systems in wetland management and analysis.

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LIST OF APPENDICES

APPENDIX A
Wetland Vegetation Tolerance Chart

TABLE A-1: WETLAND VEGETATION TOLERANCE DEPTH RANGES

	Common Name	Scientific Name	Depth Range (cm)*	Comments	Importance to Wildlife	References
Open Water	Coontail	*Ceratophyllum demersum*	35 to 115	• Optimal depth range from 75 cm and deeper		Geis, 1985; Kadlec and Wentz, 1974
				• Maximum depths of 700 cm	• Seeds and leaves eaten by waterfowl • Muskrats occasionally eat leaves • Food and shelter for invertebrates	Newmaster *et al.*, 1997
		C. echinatum, C. submersum	30 to 150	• Intolerant of drainage; may grow from seeds the following year • Can survive in depths of 760 to 915 cm	• Foliage used by invertebrates	Lamoreux, 1970 in Ould and Holbrow, 1987
	Slender Naiad	*Najas flexilis*	35 to 115	• Optimal depth range from 60 to 85 cm		Geis, 1985
			30 to 800			Kadlec and Wentz, 1974
					• Plants and seeds eaten by waterfowl • Food and shelter for fish	Newmaster *et al.*, 1997
	Muskgrass	*Chara vulgaris*	35 to 115	• Optimal depth range from 55 to 85 cm		Geis, 1985; Herdendorf, Hartley and Barnes, 1981
			30 to 800			Kadlec and Wentz, 1974
					• Source of food for waterfowl • Ducks occasionally eat tubers	Lamoreux, 1970 in Ould and Holbrow, 1987
					• Primary source of food for waterfowl • Occasionally eaten by moose	Newmaster *et al.*, 1997
	Wild Celery	*Vallisneria americana*	35 to 115	• Optimal depth range from 95 cm and deeper		Geis, 1985
			30 to 300			Kadlec and Wentz, 1974
					• Source of food for waterfowl, birds, muskrats • Shelter for fish and invertebrates	Newmaster *et al.*, 1997
				• Can also exist in floating form with flowers and floating leaves depending on the season		Snell, 2002
	Sago Pondweed	*Potamogeton pectinatus*		• Maximum depth of 55 cm		Herdendorf *et al.*, 1981
				• Optimal depth of 30 cm • 90 cm too deep for optimal growth		Ould and Holbrow, 1987
			5 to 300			Kadlec and Wentz, 1974
				• Maximum depth of 400 cm	• Important food for wildlife, especially waterfowl and marsh birds	Newmaster *et al.*, 1997
				• Negatively affected by high water levels in growing season		EPA, 2000
	Water Milfoil	*Myriophyllum heterphyllym*	55 to 115	• Optimal depth range from 90 cm and deeper		Geis, 1985
				• Occupies shallow water at edge of open shoreline • Less abundant in deeper bays		Walker, 1965 in Ould and Holbrow, 1987
				• Occurs in stands inundated continuously for 4 or more years and converted to open water	• Fruits eaten by waterfowl	Ould and Holbrow, 1987
	Longroot Smartweed	*Polygonum coccineum*		• Maximum depth of 120 cm	• Seeds source of food for ducks	Ould and Holbrow, 1987
			5 to 105	• Optimal depth range from 50 to 70 cm		Geis, 1985
				• Can also exist in floating form if abundant enough		Snell, 2002
	Water Smartweed	*Polygonum. amphibium*		• Floating form	• Seeds are a winter staple for many birds • Mammals eat the plant and fruit	Newmaster *et al.*, 1997

	Common Name	Scientific Name	Depth Range (cm)*	Comments	Importance to Wildlife	References
Floating Emergent	Duckweed	*Lemna* spp.		• Obliterates submerged aquatics when forms superficial mat on water surface • During drawdown, the mat is restricted to germination in emergent marsh		Walker, 1965 in Ould and Holbrow, 1987
				• Occurs in stands inundated continuously for 4 or more years and converted to open water		Miller, 1973 in Ould and Holbrow, 1987
					• Home for snails and invertebrates which are eaten by ducks	Ould and Holbrow, 1987
					• Source of food for waterfowl, birds • Occasional food for muskrat and beavers • Food and shelter for aquatic invertebrates	Newmaster *et al.*, 1997
	Floating-Leaf (Broadleaf) Pondweed	*Potamogeton natans*	30 to 150	• Grows in very moist soil during drawdown • Seeds produce more efficiently when roots are submerged		Lamoreux, 1970 in Ould and Holbrow, 1987
			90 to 200			Kadlec and Wentz, 1974
					• Seeds and roots source of food for waterfowl	Ould and Holbrow, 1987
					• Source of food for waterfowl; habitat and food for fish	Newmaster *et al.*, 1997
	Yellow Pond Lily	*Nuphar variegatum*		• Maximum depth of 120 cm	• Seeds eaten by waterfowl	Ould and Holbrow, 1987
					• Waterfowl eat seeds • Beaver, muskrat, porcupine, deer eat rhizomes and leaves • Moose favor stems and rhizomes • Leaves provide food and shelter for fish and invertebrates • Similar to white water lily	Newmaster *et al.*, 1997
	American Lotus	*Nelumboa lutea*	45 to 100			Bellrose and Brown, 1941
				• Maximum depth of 150 cm; but mostly found in shallower areas		GLIN, 1988a
				• Narrow range of moisture conditions		Walker and Coupland, 1968 in Ould and Holbrow, 1987
				• Flourished where water was above soil surface for part of the growing season		Tolstead, 1942, Marks, 1942, Walker, 1965 in Ould and Holbrow, 1987
Emergent	Wild Rice	*Zizania aquatica*		• Maximum depth of 70 cm; produces high yield up to depth of 215 cm	• Food for waterfowl	Ould and Holbrow, 1987
			5 to 180			Kadlec and Wentz, 1974
				• Intolerant of low water levels, flooding, fluctuations		EPA, 2000
					• Important food for waterfowl and marsh birds	Newmaster et al., 1997
	Hardstem Bulrush	*Scirpus acutus*		• Mean depth of 109 cm; maximum 150 cm • Found in shallower water		Dabbs, 1971 in Ould and Holbrow, 1987
				• Maximum depth of 45 cm		Lamoreux, 1970 in Ould and Holbrow, 1987
				• Develops in water over 90 cm deep		Martin and Uhler, 1939 in Ould and Holbrow, 1987
			45 to 75	• Mean depth of 110 cm, maximum depth of 150 cm		Dirschl and Dabbs, 1969 in Ould and Holbrow, 1987

	Common Name	Scientific Name	Depth Range (cm)*	Comments	Importance to Wildlife	References
Emergent			20 to 25	• Similar to pickerel weed and burreed		Herdendorf et al., 1981
				• Optimal depth of 45 cm • Maximum depth from 50 to 60 cm		Koshida, 1988
				• Optimal depth of 110 cm • Maximum depth of 150 cm	• Seeds eaten by waterfowl • New shoots eaten by geese and swans • Rootstocks eaten by ducks and muskrats • Nesting area for ducks • Shelter for muskrats	Ould and Holbrow, 1987
				• Maximum depth of 150 cm		Kadlec and Wentz, 1974
	Softstem Bulrush	*S. validus*		• Dies if flooded in 40 cm of water for 2 years or more; complete destruction after 3 years		Harris and Marshall, 1963
				• Intolerant of deep water		Walker, 1965 in Ould and Holbrow, 1987
			45 to 65	• Mean depth of 45 cm, maximum depth of 65 cm • Forms continuous bands of emergent vegetation from shoreline to 45 cm depths	• Seeds eaten by waterfowl • Nesting area for ducks	Dabbs, 1971 in Ould and Holbrow, 1987
				• Restricted to shoreline where depth is less than 60 cm		Dirschl and Dabbs, 1969 in Ould and Holbrow, 1987
				• Maximum depth of 120 cm		Kadlec and Wentz, 1974
	Three-Square Bulrush	*S. americanus*		• Optimal depth of 30 cm	• Nublets eaten by waterfowl • Shelter for ducks	Ould and Holbrow, 1987
				• Maximum depth of 60 cm		Kadlec and Wentz, 1974
	Burreed	*Sparganium eurycarpum*		• Grows on semi-dry banks in depths less than 10 cm		Liang, 1941, MacDonald, 1955 in Ould and Holbrow, 1987
					• Leaves and seeds eaten by muskrats • Shelter for muskrats	Ould and Holbrow, 1987
				• Maximum depth of 120 cm		Kadlec and Wentz, 1974
				• Maximum depth of 100 cm • Can also exist in floating form	• Fruit eaten by birds, waterfowl • Stems and leaves eaten by muskrats and deer	Newmaster et al., 1997
	Pickerelweed	*Pontederia cordata*		• Optimal depth of 90 cm		Lamoreux, 1970 in Ould and Holbrow, 1987
				• Maximum depth of 120 cm		Kadlec and Wentz, 1974
				• Maximum depth of 100 cm	• Seeds eaten by muskrats and waterfowl • Leaves eaten by deer and muskrat occasionally	Newmaster et al., 1997
	Sweetflag	*Acorus calamus*	60 to 90		• Possible duck nesting cover	Ould and Holbrow, 1987
					• Rhizomes eaten by muskrats	Newmaster et al., 1997
	Arrowhead	*Sagittaria* spp.		• Maximum depth of 30 cm • 10 to 15 cm depth in canals and creeks • Sparse in water over 75 cm deep	• Seeds and tubers eaten by waterfowl	Sharp, 1951, Lamoreux, 1970 in Ould and Holbrow, 1987
				• Can also exist in floating form		Newmaster et al., 1997
	Broad-Leaved Arrowhead	*S. latifolia*		• Maximum depth of 30 cm		Kadlec and Wentz, 1974
					• Seeds and tubers valuable food for waterfowl and birds • Leaves and tubers eaten by muskrats and porcupines	Newmaster et al., 1997

	Common Name	Scientific Name	Depth Range (cm)*	Comments	Importance to Wildlife	References
Tall Emergent (Wet and Dense and Dry)	Cattail	*Typha* spp.	30 to 50	• Maximum depth of 150 cm • Abundant growth with low summer water levels • Dies during prolong high water levels		Walker, 1965 in Ould and Holbrow, 1987
				• Dies if continually flooded in 30 to 120 cm deep water; greatly decreases in 3rd year of flooding in depths over 30 and 40 cm • Common cattail is least tolerant, white cattail is more tolerant and narrowleaf most tolerant of flooding		Harris and Marshall, 1963
	Narrowleaf Cattail	*T. augustifolia*		• Maximum depth of 90 cm • Can survive flooding up to30 cm depths		MacDonald, 1955
			- 90 to 60			Kadlec and Wentz, 1974
	White (Hybrid) Cattail	*T. glauca*	10 to 30	• Optimal depth range from 10 to 15 cm	• Food source and building material for furbearers	Herdendorf et al., 1981
				• Maximum depth of 70 cm • Dormant shoots die if submerged since they need air during winter to survive		Liang, 1941, MacDonald, 1955 in Ould and Holbrow, 1987
			5 to 55	• Optimal depth range from 25 to 40 cm		Geis, 1985
	Broadleaf (Common) Cattail	*T. latifolia*	30 to 50	• Maximum depth of 150 cm • Dies during persistent depths of 30 cm	• Cover for waterfowl • Nesting area and food for ducks	Walker, 1965 in Ould and Holbrow, 1987
				• Maximum depth of 30 cm • Shoots need air during winter to survive		Sharp, 1951 in Ould and Holbrow, 1987
					• Nesting material for birds • Rhizomes eaten by geese • Food and shelter for muskrat	Newmaster et al., 1997
	Bluejoint	*Calamagrostis canadensis*	0 to 60	• Optimal depth range from 15 to 40 cm		Geis, 1985
					• Deer, muskrat, moose graze on young shoots • Wildlife cover in winter	Newmaster et al., 1997
	Reed Canary Grass	*Phalaris arundinacea*			• Flooded strands used by waterfowl, unflooded strands provide nesting for ducks and cover for wildlife	Haworth-Brockman, 1989
				• Occurs in less moist areas than cattail • Less biomass in depths of 30 cm		EPA, 2000
					• Little value to wildlife except for cover	Newmaster et al., 1997
	Sedge	*Carex* spp.		• Maximum depth range of 15 cm		Kadlec and Wentz, 1974
			15 to 20			Herdendorf et al., 1981
	Awned (Slough) Sedge	*C. atherodes*		• Good growth and seed production in 2nd year of flooding up to 70 cm; reduced in 3rd year of flooding over 30 to 40 cm; gone in 4th year		Harris and Marshall, 1963
				• Narrow tolerance range	• Nesting cover for duck • Food and shelter for muskrat	Ould and Holbrow, 1987
	Uptight Sedge	*C. stricta*	0 to 25	• Maximum depth of 55 cm		Geis, 1985
			-45 to 30			Herdendorf et al., 1981

	Common Name	Scientific Name	Depth Range (cm)*	Comments	Importance to Wildlife	References
Tall Dense Dry Emergent	Common Reed Grass	*Phragmites australia*	-200 to 100	• Maximum depth of 50 cm	• Duck nesting cover, muskrat food and shelter • Cover for wildlife and waterfowl • Shelter and food for muskrats	Haslam, 1970 in Ould and Holbrow, 1987 Kadlec and Wentz, 1974 Haworth-Brockman, 1989
				• Maximum depth of 200 cm • Found in deeper water since grows to heights from 200 to 400 cm	• Rhizomes eaten by muskrats; little value to wildfowl; cover for wildlife	Newmaster et al., 1997
				• Less biomass in depths of 80 cm		EPA, 2000
	Water Persicaria	*Polygonum amphibium*		• Maximum depth of 30 cm • Upright form, can also exist in floating form	• Seeds eaten by waterfowl	Ould and Holbrow, 1987
Short Wet Meadow	Creeping Spikerush	*Eleocharis palustris*		• Wide tolerance amplitude	• Seeds and tubers source of food	Ould and Holbrow, 1987
				• Died after 2 years of flooding in depths greater than 40 cm; can thrive in 30 cm for longer periods		Harris and Marshall, 1963
				• Maximum depth of 50 cm		Kadlec and Wentz, 1974
					• Important source of food for waterfowl • Rhizomes eaten by birds and muskrats	Newmaster et al., 1997
	Meadowsweet	*Spiraea* spp.	40 to 75			Dane, 1959
			35 to 57	• Minimum depth in dormant season • Maximum depth in early growing season		EPA, 2000
	Buttonbush	*Cephalanthus occidentalis*		• Maximum depth of 50 cm		Dane, 1959
Meadow	Alder	*Alnus* spp.	0 to 5	• Maximum depth of 25 cm		Geis, 1985
				• More prevalent in drier sites after high water level fluctuations in early growing season		EPA, 2000
					• Food and shelter for wildlife	Newmaster et al., 1997
	Silky Dogwood	*Cornus amomum*	0 to 5	• Maximum depth of 25 cm		Geis, 1985
	Red-Osier Dogwood	*C. stolonifera*	0 to 15			Geis, 1985
					• Winter staple for many wildlife • Fruit, leaves and wood eaten by waterfowl and small mammals	Newmaster et al., 1997
Treed	Common Winterberry	*Ilex verticillata*		• Maximum depth of 50 cm		Dane, 1959
	Silver Maple	*Acer saccharinum*		• Maximum depth of 45 cm • Tolerant of flooding		Dane, 1959
	Red Ash	*Fraxinus pennsylvania*	30 to 45	• Maximum tolerance range of flooding		Dane, 1959
	American Elm	*Ulmus americana*	25 to 30	• Moderately tolerant		Dane, 1959
	Willow	*Salix* spp.		• Optimal depth of 10 cm		Koshida, 1988
			30 to 45	• Maximum tolerance range of flooding • Tolerant of flooding		Dane, 1959
				• Young died after 2 to 3 years of moderate flooding; willows 4 to 5 years thrived in 30 cm depth flooding; complete death after 4 years in water depths of 70 cm, after 5 years of 45 cm depths		Harris and Marshall, 1963

* Water depths provided in feet and inches were converted to centimetres (cm) and rounded to the nearest 5
Negative "water depth" values indicate height above lake level

APPENDIX B
Wetland Trends Through Time Classification Scheme

TABLE B-1: WETLAND VEGETATION COMMUNITIES (IN APPROXIMATE ORDER OF WETTER TO DRIER CONDITIONS)

CODE*	CLASS	DESCRIPTION
W	11	Some evidence of submergents OR dominantly water with some emergent, floating or flooded vegetation
		Includes open water where submergents are possible but not visible on the photos
E1	25	Flat and/or wet emergents
		If species identification possible E1 can include:
	19	EL *Lemna*
	20	EN *Nuphar* or *Nymphaea*
	24	EZ *Zizania*
		Or E species if relatively sparse with water 25 % - 50 % of the polygon
E	13	Mixture of E1 and E2 or conditions between
		If species identification is possible E can include:
	12	CW Cattail (*Typha*) dominant but with significant interspersed water
	205	EgW Grass/sedge hummocks with significant interspersed water
	15	EB *Sparganium*
	206	EF Light, short E patches especially among cattail in early spring (1964); possibly last year's EG or GS
	22	ES *Scirpus*
	23	ET Trees and shrubs in water
E2	26	Taller, denser and possibly drier emergents
		If species identification is possible E2 can include:
	27	C Cattail (*Typha*)
	82	EG Grass/sedge hummocks
	21	EP *Phragmites*
	207	GS Grass/sedge without hummocks
M1	80	Wet meadow
		Includes very short non-hummocky meadow that in wet years is almost W
M	28	Meadow, can include small shrubs
MT	32	Mixed meadow and trees; or shrub covered
T	29	Trees, large shrubs or scattered trees

* The above codes have a prefix N if the community is within a National Wildlife Area (the diked areas of Big Creek NWA)

TABLE B-2: ADJACENT UPLAND LAND USE CLASSES

CODE	CLASS	DESCRIPTION
A	50	Agriculture, large estate lawns or rural managed open space, including farm buildings (farmsteads included under B)
B	67	General Built-Up
D	54	Pond
I	55	Idle land, rough pasture, scrubby vegetation (< 5 % tree cover)
J	202	Sand exposed but with enough idle vegetation cover to at least partially stabilize it
L	57	Lake
Mr	58	Marsh separated from study wetland by upland
Or	59	Orchard, tree nursery or young reforestation
P	60	Park or park like (e.g., managed grass areas), cemetery; includes non-closed forest parkland (i.e. 5 % - 70 % tree cover)
Pc	61	Park campground, trailer park
Pg	204	Golf course, or other recreation
R	62	River, canal
S	63	Marina, dock, collection of boathouses
Tp	64	Causeway, raised road, weir, breakwall or dike attached to upland
U	31	Sandbar, remnant dike
X	65	Sand, rock, disturbed soil, fill, pavement
Z	66	Woodland (> 70 % tree cover)

TABLE B-3: SIMPLIFIED WETLAND COMMUNITY CLASSIFICATION

SIMPLIFIED CODE	SYMBOL	DESCRIPTION	CODE GROUPING *
1	L	Lake	L
2	OW	Open Water (Possible Submergent)	W
3	E1	Floating (or Sparse) Emergent	E1, EL, EN, EZ
4	E	Emergent	E, EB, EF, ES, ET
5	EW	Tall Wet Emergent	CW, EgW
6	E2	Tall Dense Dry Emergent	E2, C, EG, EP, GS
7	M1	Short Wet Meadow	M1
8	M	Meadow	M, MT
9	T	Treed	T
10	U	Upland (Non-Lake/Non-Wetland)	Includes all upland land use classes (excluding lake)

* The above codes have a prefix N if the community is within a National Wildlife Area (the diked areas of Big Creek NWA)

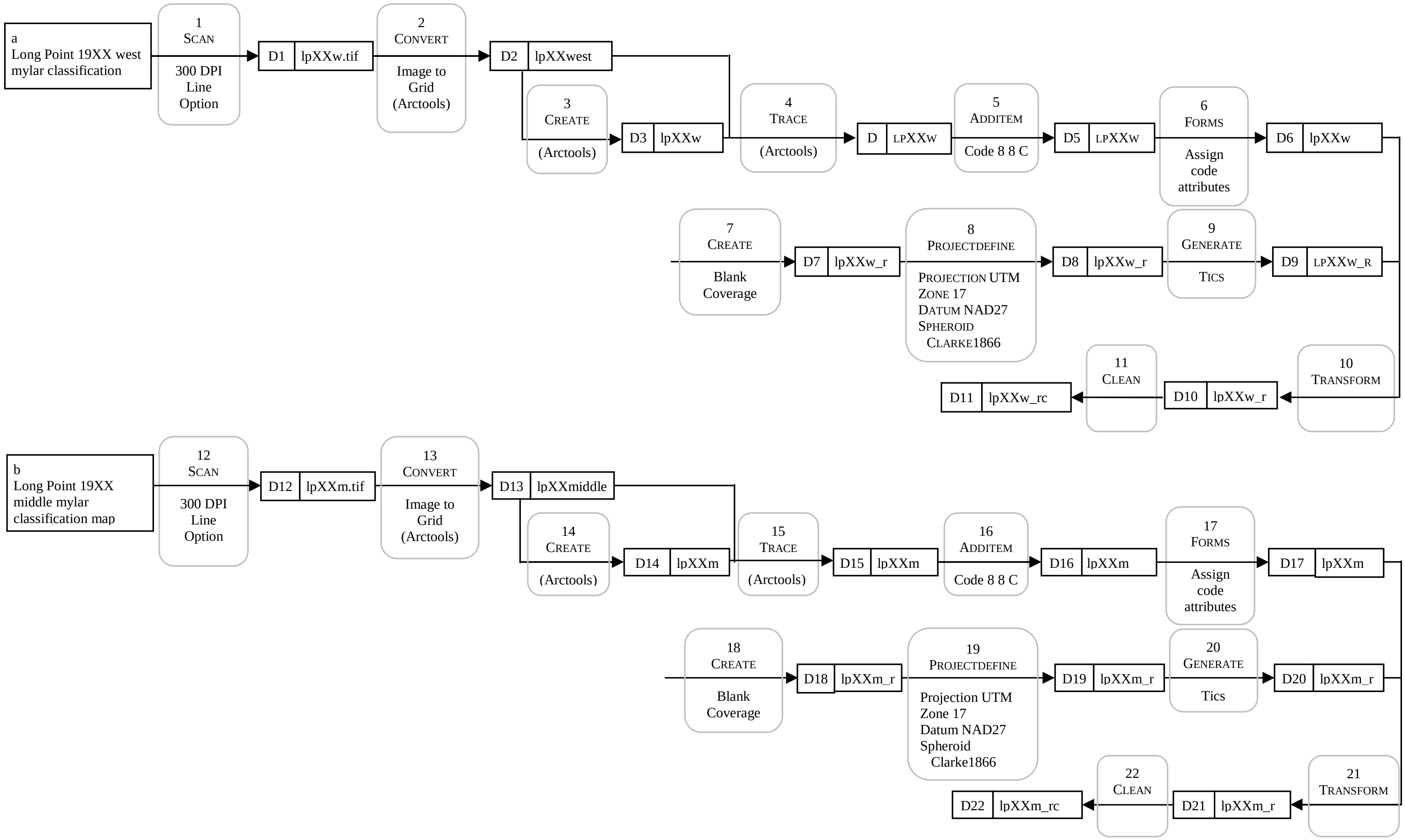
APPENDIX C
Data Processing Flow Diagrams

DATA PROCESSING FLOW DIAGRAMS

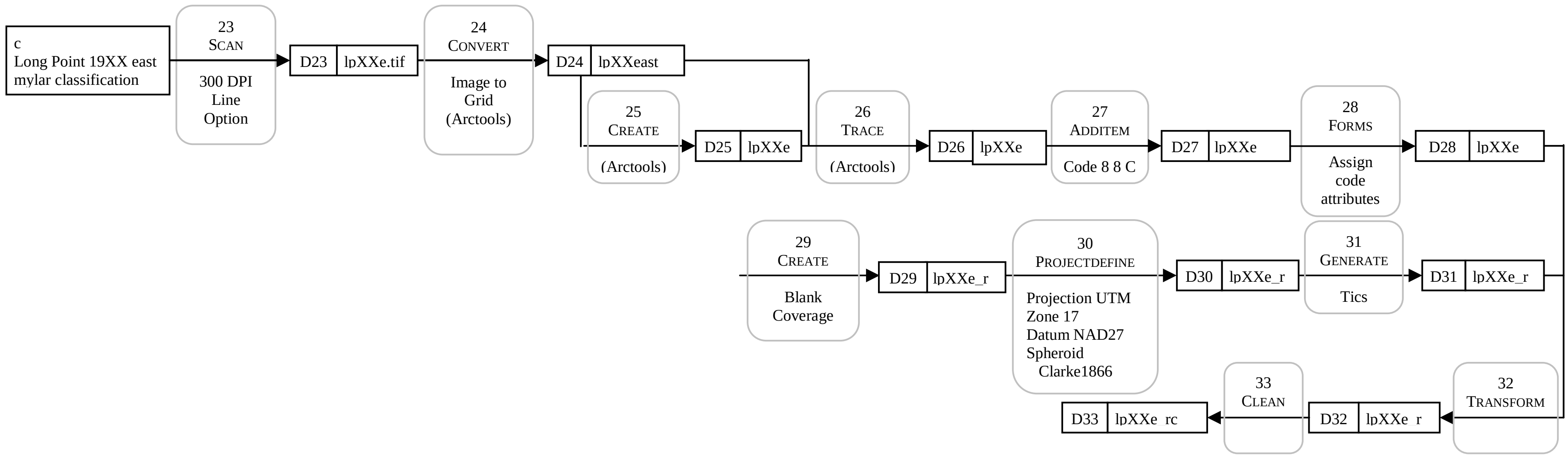
EXPLANATION OF SYMBOLS USED IN THE DATA FLOW DIAGRAM

i INPUT	**INPUT FILE** i indicates the input sequence number. INPUT is the name and type (or description) of the input file.
P COMMAND Details	**PROCESS COMPLETED** P is the process number. Process is the command or process completed on the input file, where applicable the details of the command process are identified.
DP | OUTPUT	**OUTPUT FILE** DP is the output number (correlates to the process number). Output is the name of the output file.
→	**DIRECTION OF PROCESS**
———	**LINKAGES BETWEEN INPUTS**
XX	**YEAR** XX represents the last two digits of the year of data.
YYY	**RUN NUMBER** YYY is output run number assigned by the Fragstats analysis, where 102 is the output for 1945, 103 (1964), 104 (1985), 105 (1995), 106 (1999), 107 (1955), 108 (1978)
waterlevel	**WATER LEVEL** *Waterlevel* is actual water level for each year.

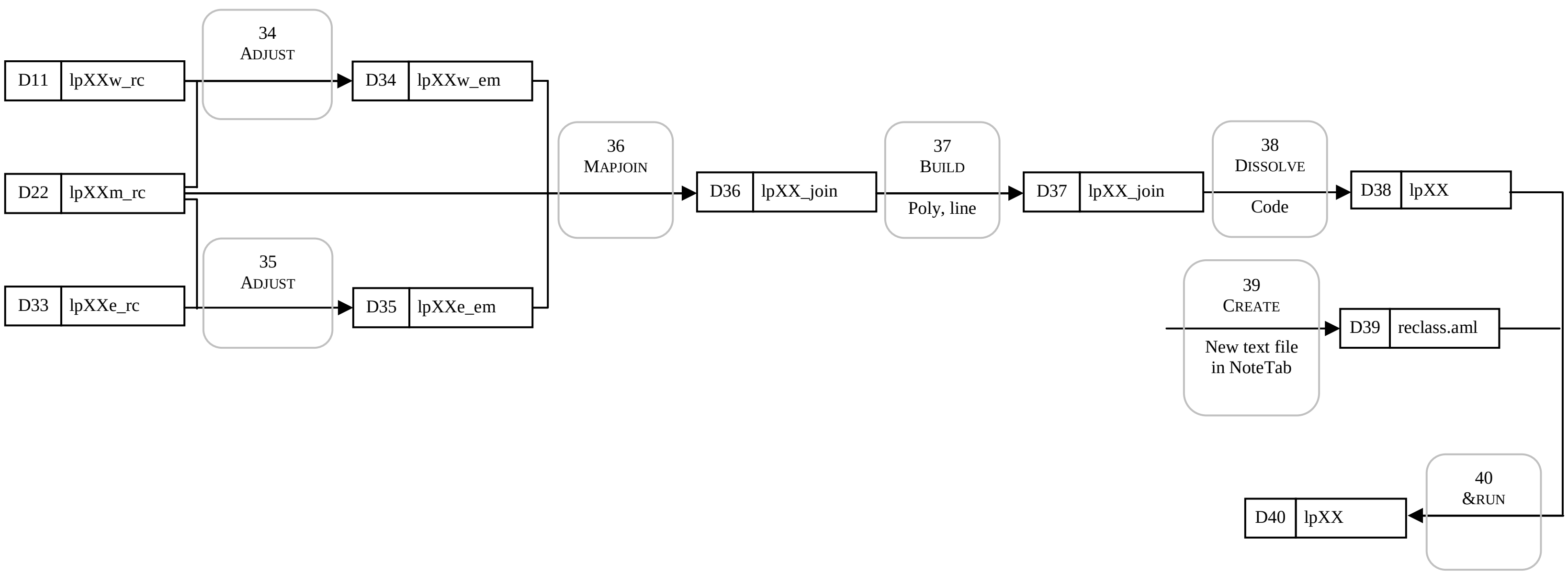
1A. Scan Vectorizing *



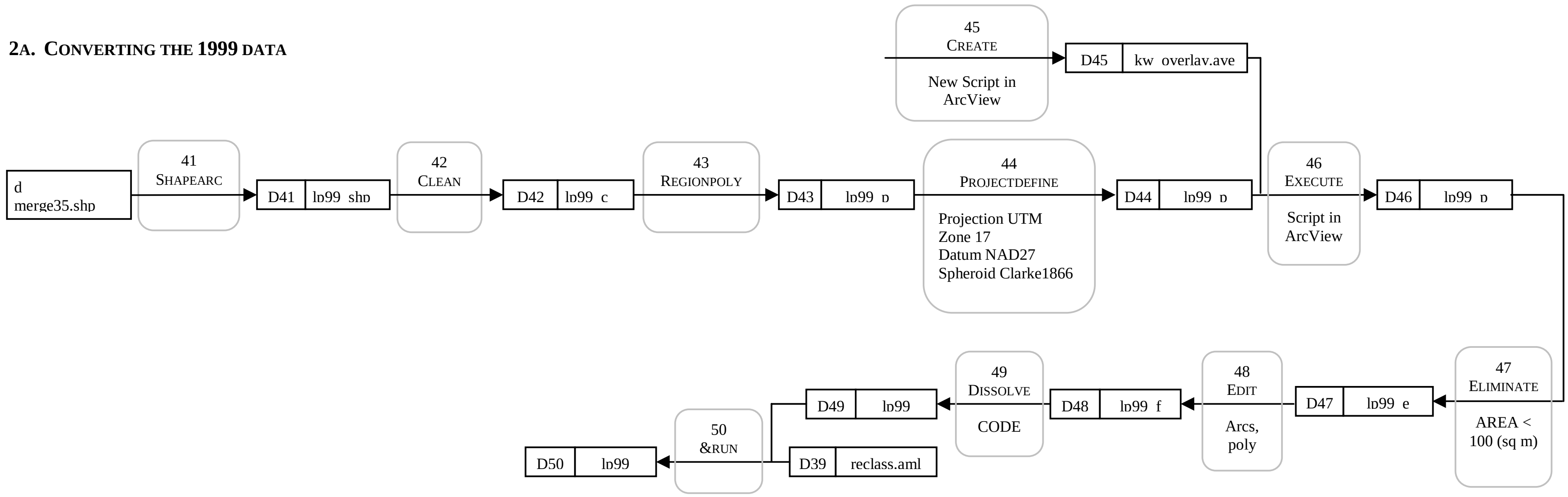
* The same vectorizing process was repeated for the 1945, 1964, 1985 and 1995; XX = 45, 64, 85, 95



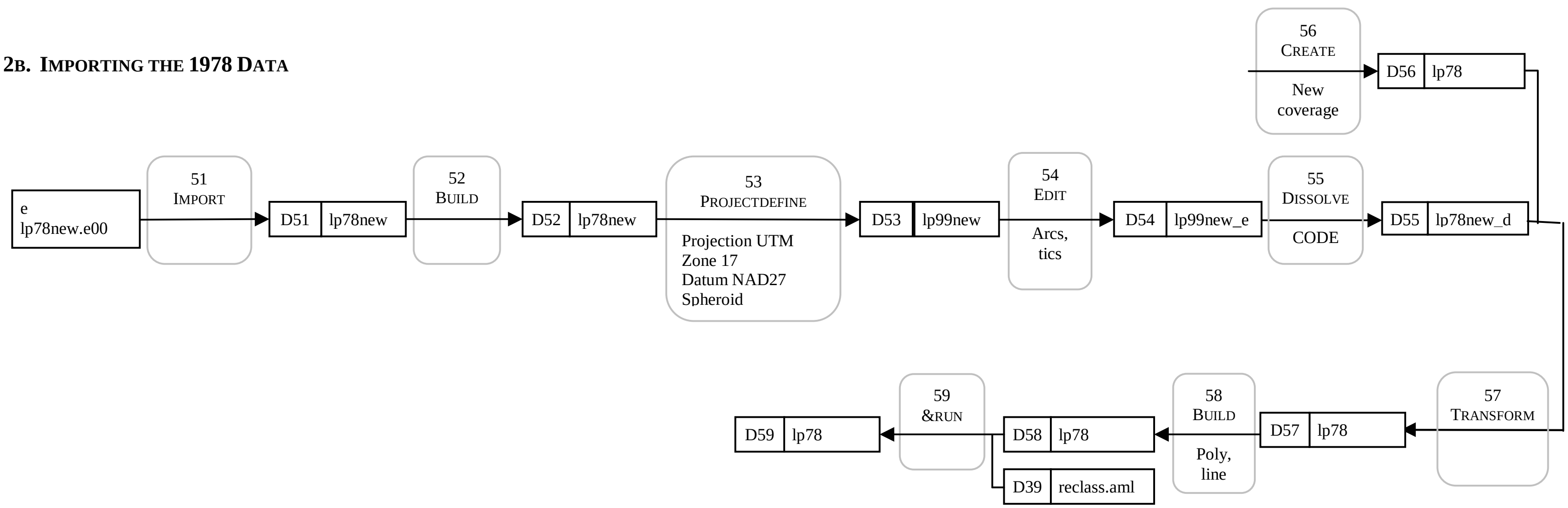
1B. EDGEMATCHING AND JOINING THE VECTORIZED COVERAGES



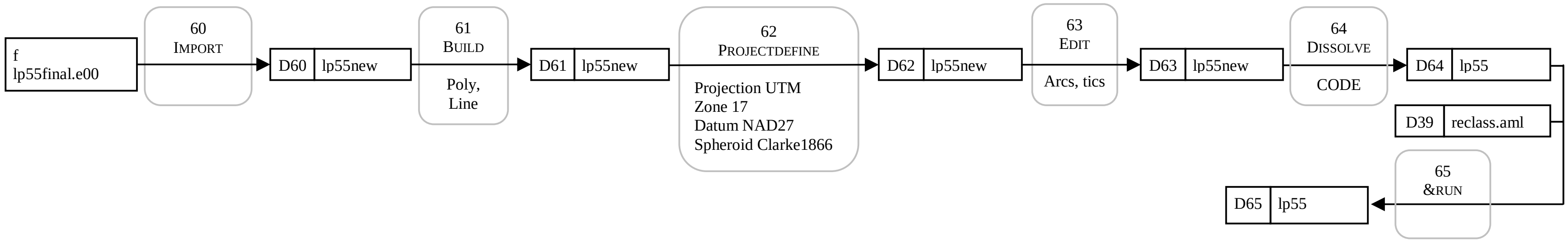
2A. Converting the 1999 data



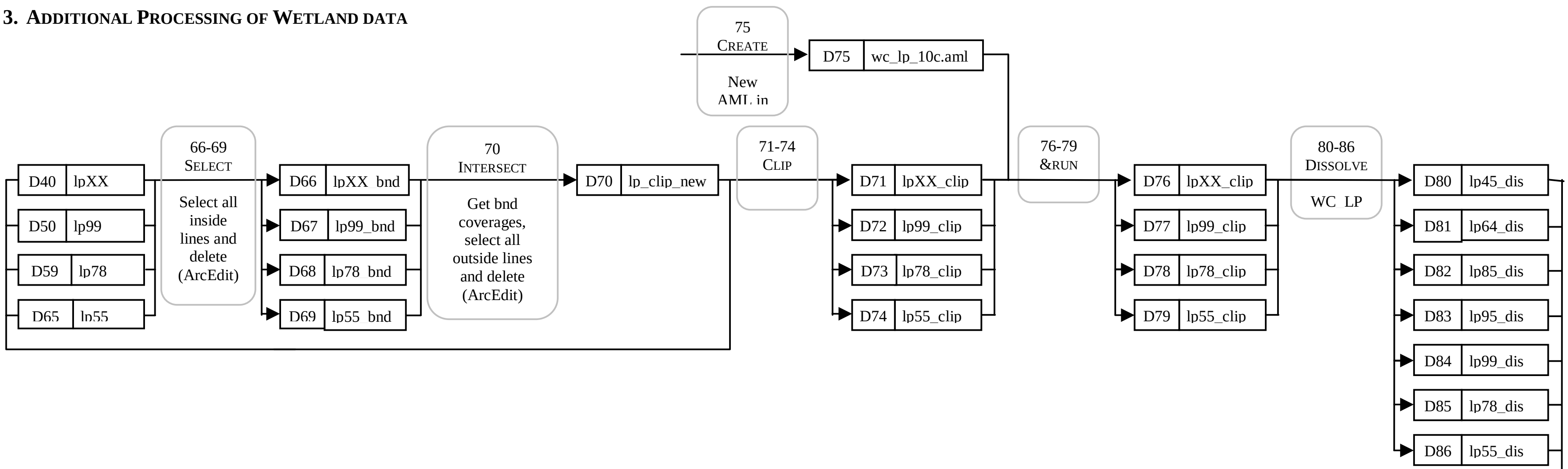
2B. Importing the 1978 Data



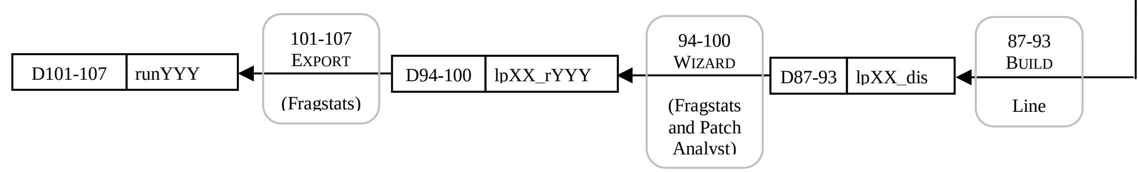
2c. Importing the 1955 Data



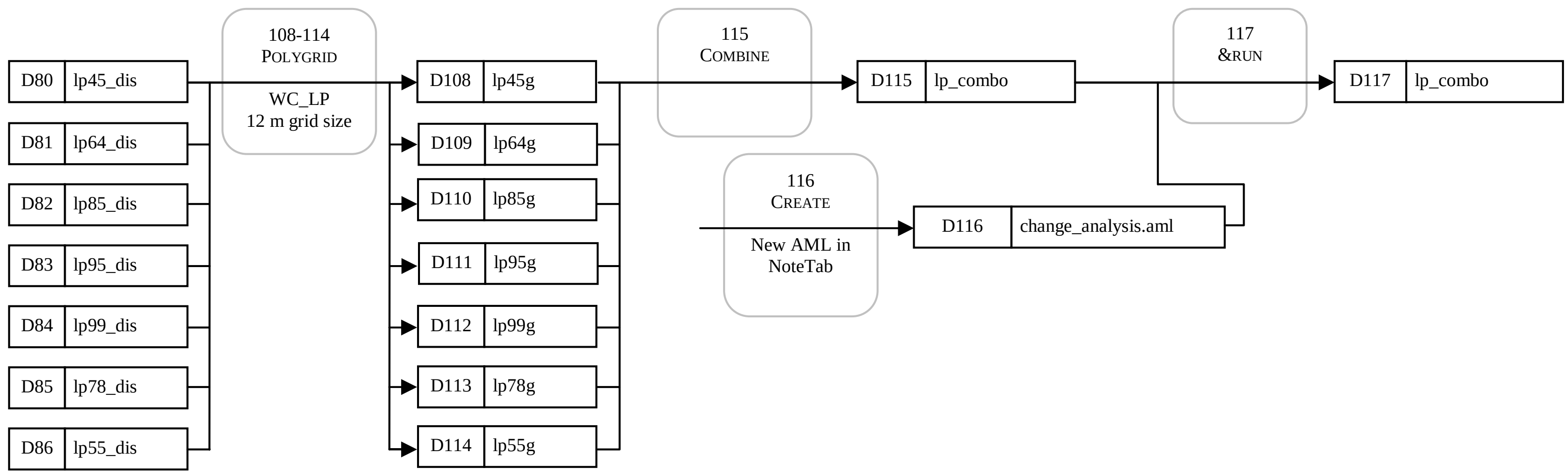
3. Additional Processing of Wetland data



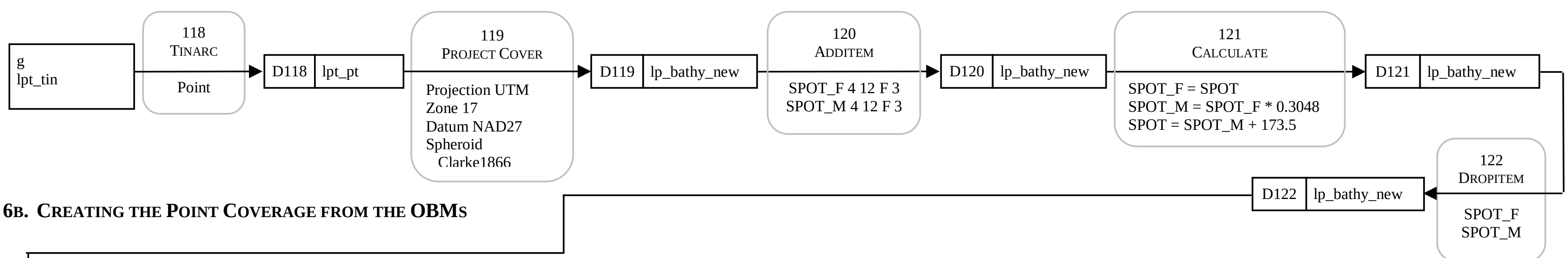
4. Fragstats Analysis



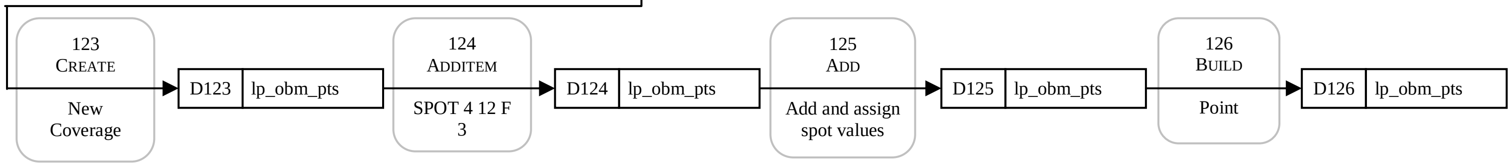
5. Spatial Grid Analysis



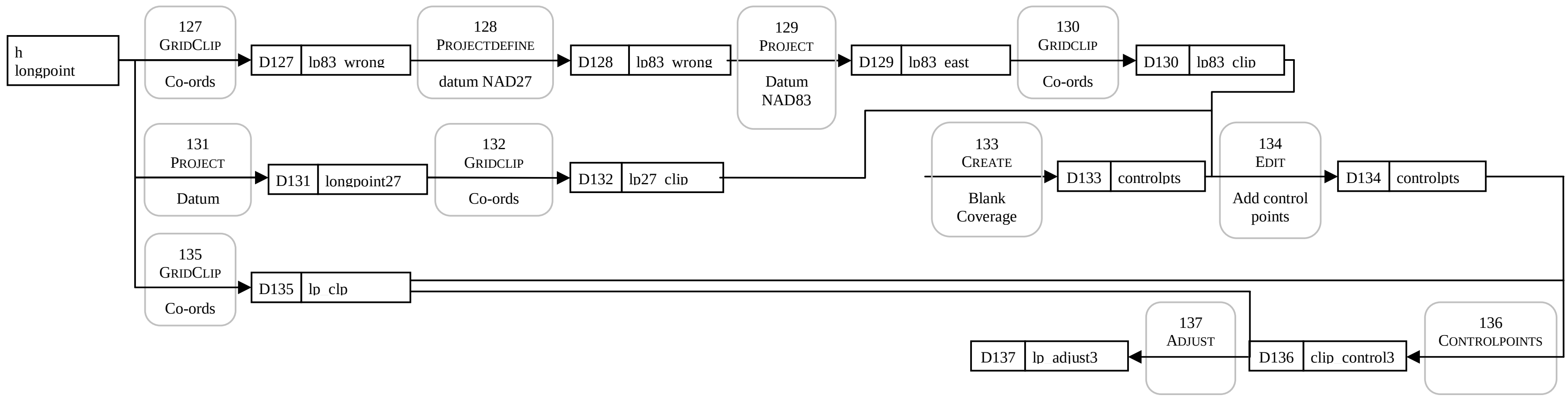
6A. Creating the Bathymetry Coverage



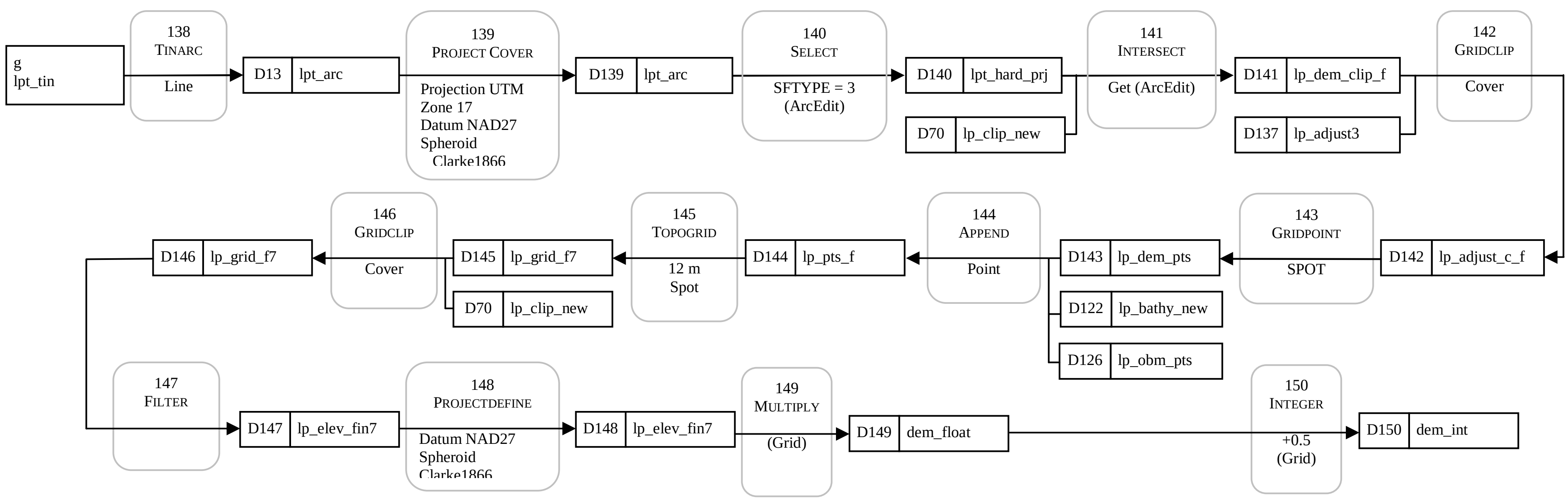
6B. Creating the Point Coverage from the OBMs



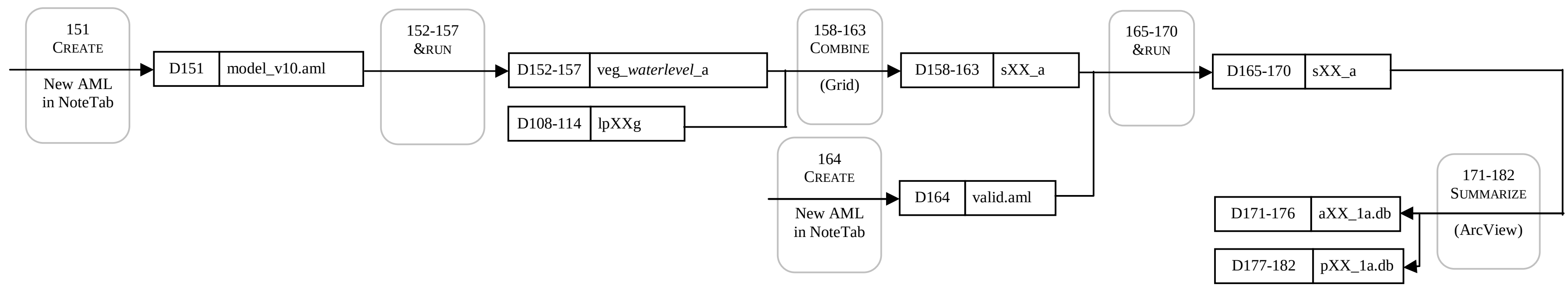
6C. Creating Point Coverage from Elevation Model

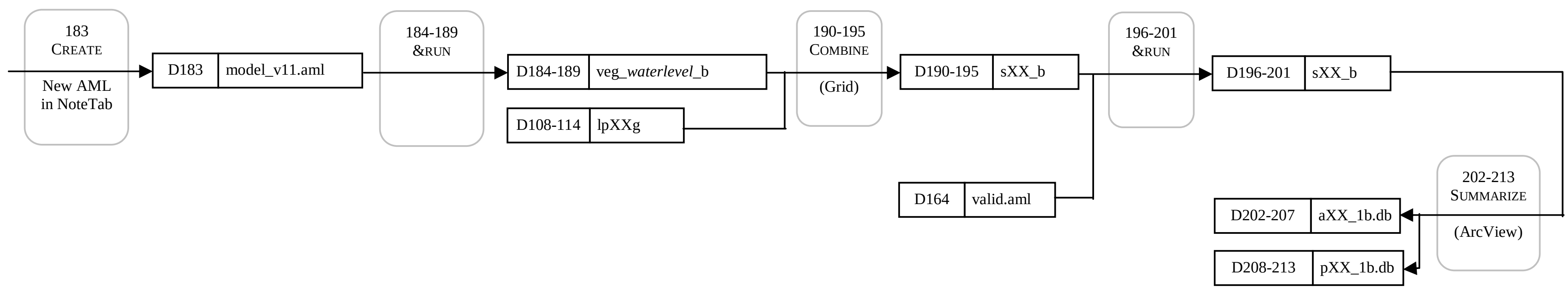


6D. Creating the Digital Elevation Model

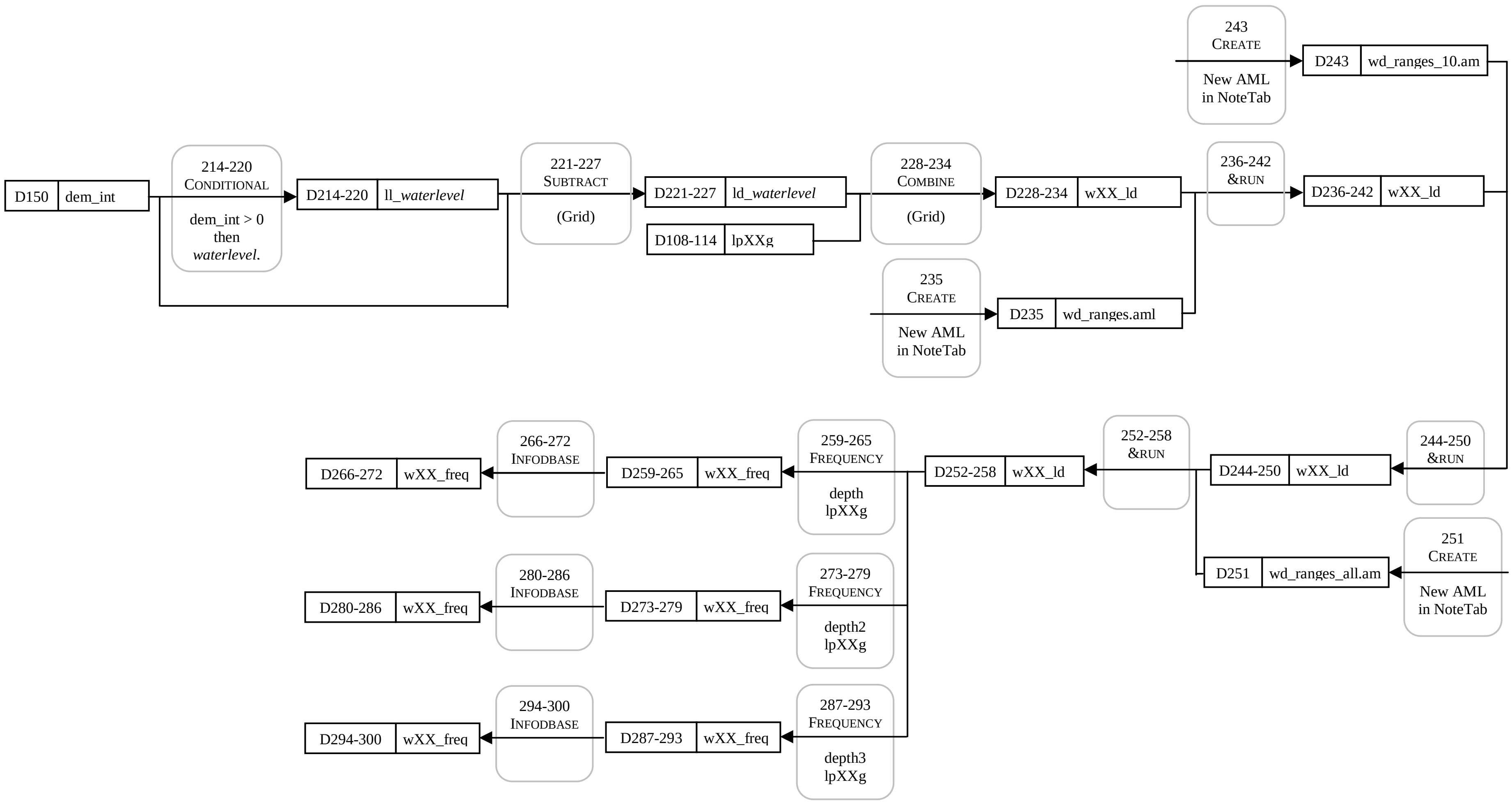


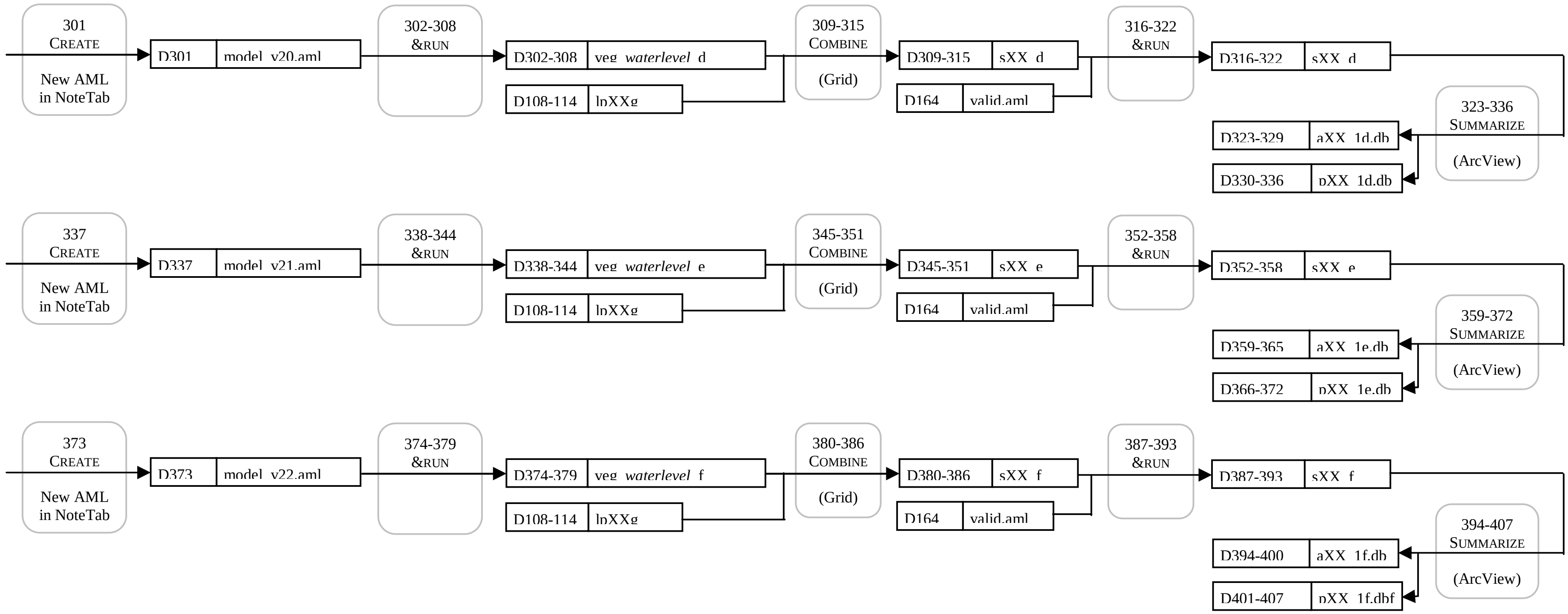
7A. MODELLING ANALYSIS – RULE-BASED MODEL



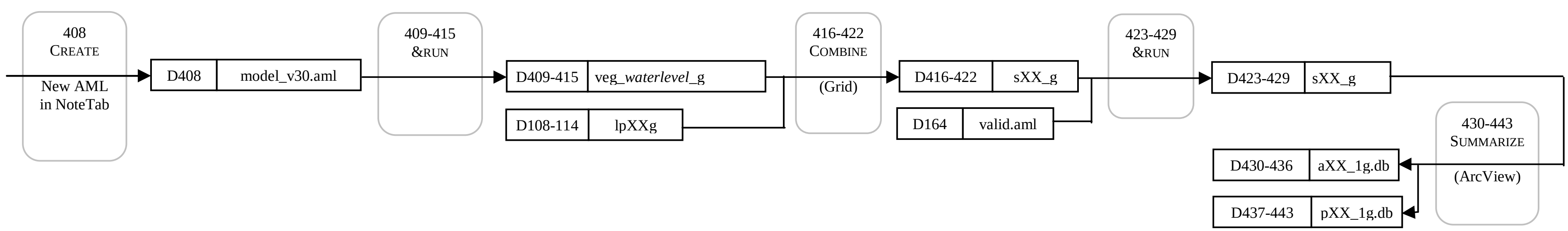


7B. Modelling Analysis - Probability Model





7C. MODELLING ANALYSIS – VEGETATION TRANSITION MODEL



APPENDIX D
Correlation Analysis

CORRELATION ANALYSIS

A correlation analysis was performed in SPSS (SPSS Inc., 2000), a statistical software package. The area values for each wetland vegetation community along with the lake level of each year were used as input into the analysis. After an initial run using the original wetland classification, the analysis was performed several times using simplified schemes. The results of the correlation analysis for the original and final simplified classification are provided in the table below.

TABLE D-1: CORRELATION COEFFICIENTS

WETLAND VEGETATION COMMUNITY CODE	PEARSON CORRELATION COEFFICIENT	SIMPLIFIED WETLAND CODE	PEARSON CORRELATION COEFFICIENT
L	0.719	L	0.719
OW	0.585	OW	0.585
E1	0.843*	E1	0.814*
EL	0.175		
EN	-0.038		
EZ	-0.132		
E	0.110	E	-0.433
EB	0.175		
EF	-0.610		
ES	-0.164		
ET	-0.641		
CW	0.508	EW	0.483
EgW	0.313		
E2	-0.516	E2	-0.519
C	-0.187		
EG	0.162		
EP	-0.589		
GS	-0.162		
M1	0.699	M1	0.699
M	-0.772*		-0.810*
MT	0.249		
T	0.246	T	0.246
U**	-0.710	U	-0.710

* Correlation coefficient is significant at the 0.05 level
** U includes all upland land use classes

Key to interpreting the Pearson Correlation Coefficient:

1) Strength of the relationship is determined by the value of the coefficient (r)
   - $-0.3 \le r \le 0.3$ indicates a weak relationship
   - $-0.7 \le r < -0.3$ or $0.3 \le r < 0.7$ indicates a moderate relationship
   - $-1.0 \le r < -0.7$ or $0.7 < r \le 1.0$ indicates a strong relationship

2) Direction of the relationship is determined by the sign of the coefficient (r)
   - $r = 0$ indicates that there is no relationship
   - $r < 0$ indicates a negative relationship (as water levels increase, class area decreases; as water levels decrease, class area increases)
   - $r > 0$ indicates a positive relationship (as water levels increase, class area increases; as water levels decrease, class area decreases)

APPENDIX E

Simplified Wetland Community Data, 1945-1995

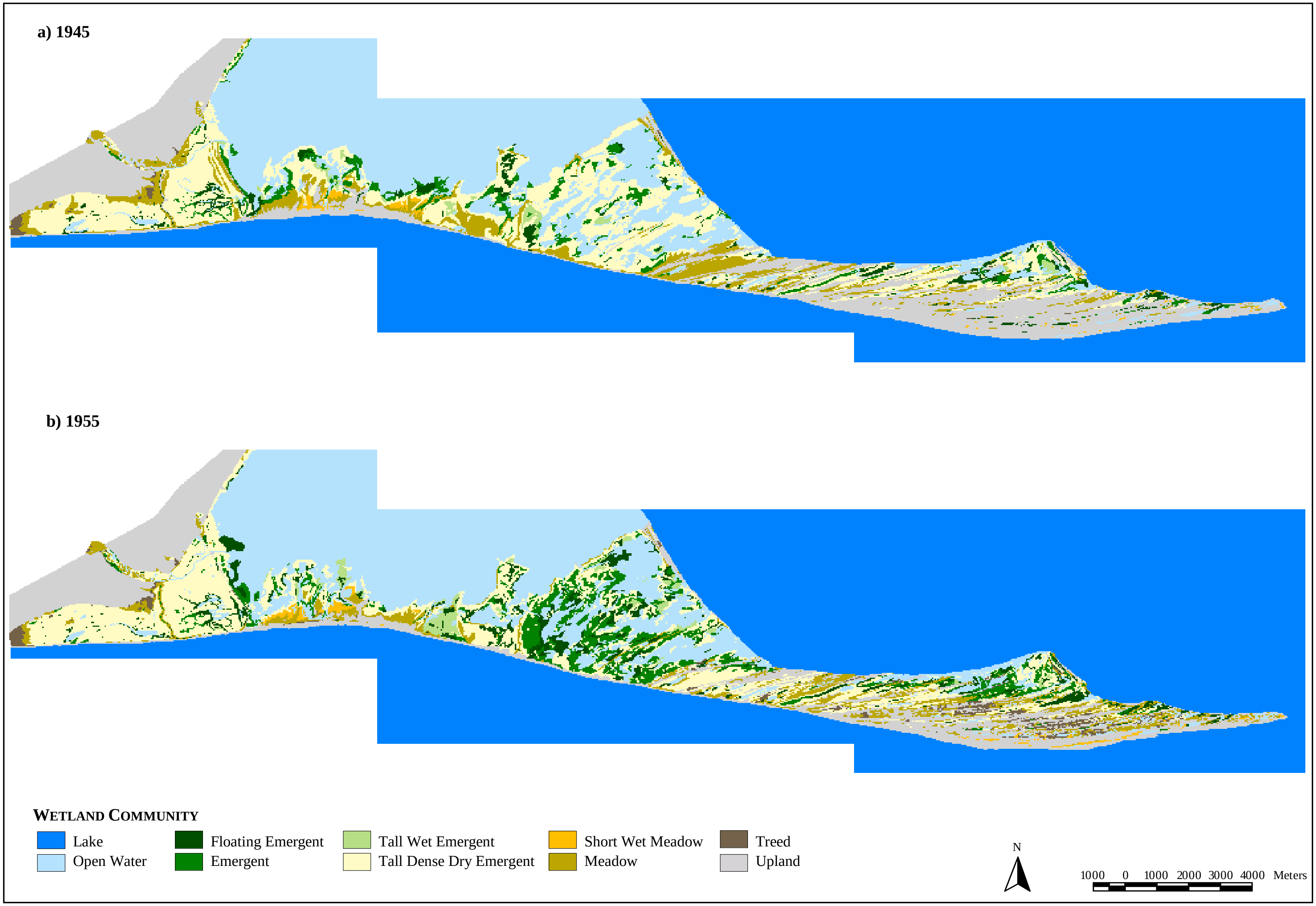


FIGURE E-1: SIMPLIFIED WETLAND COMMUNITY DATA, 1945-1955

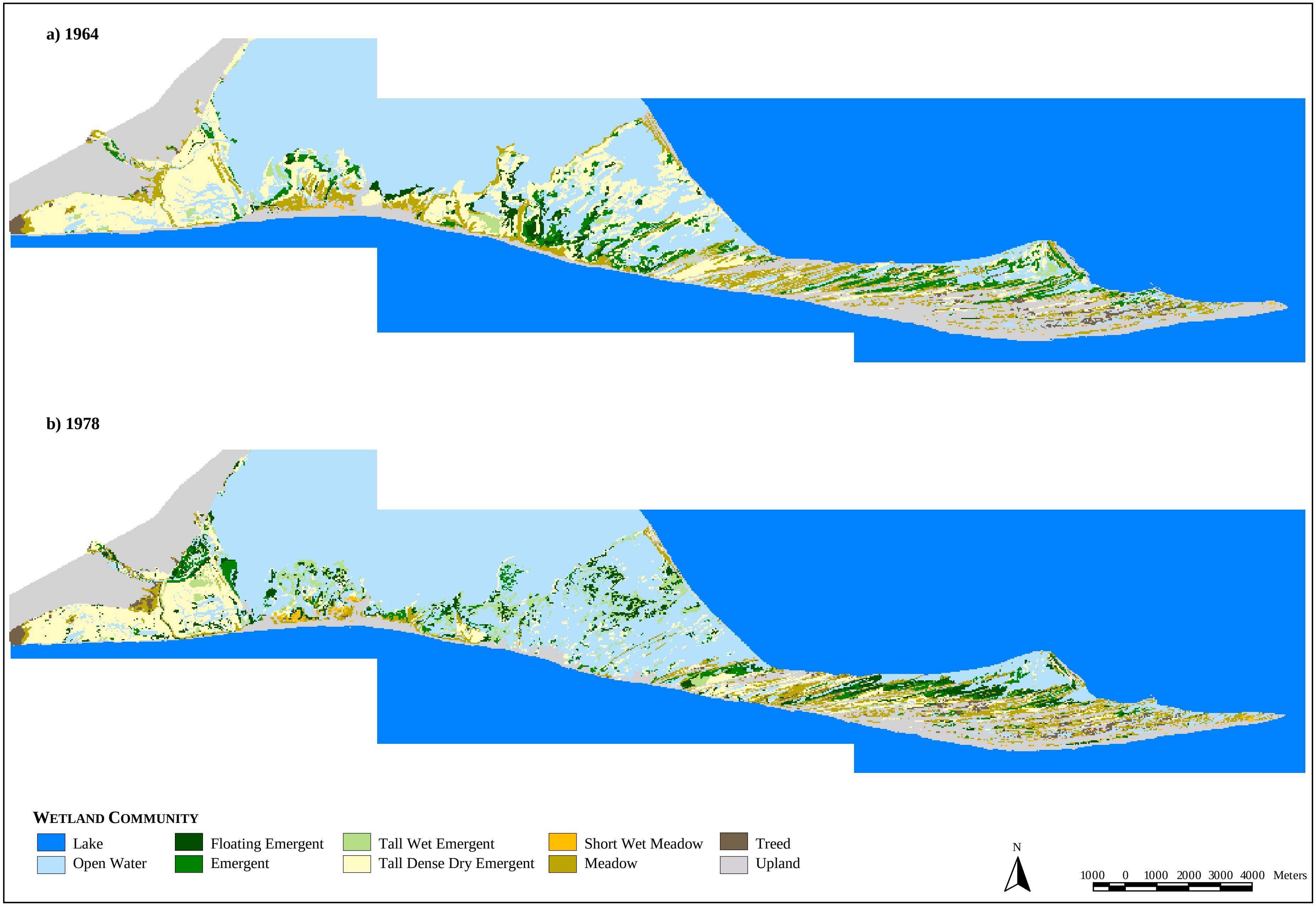


FIGURE E-2: SIMPLIFIED WETLAND COMMUNITY DATA, 1964-1978

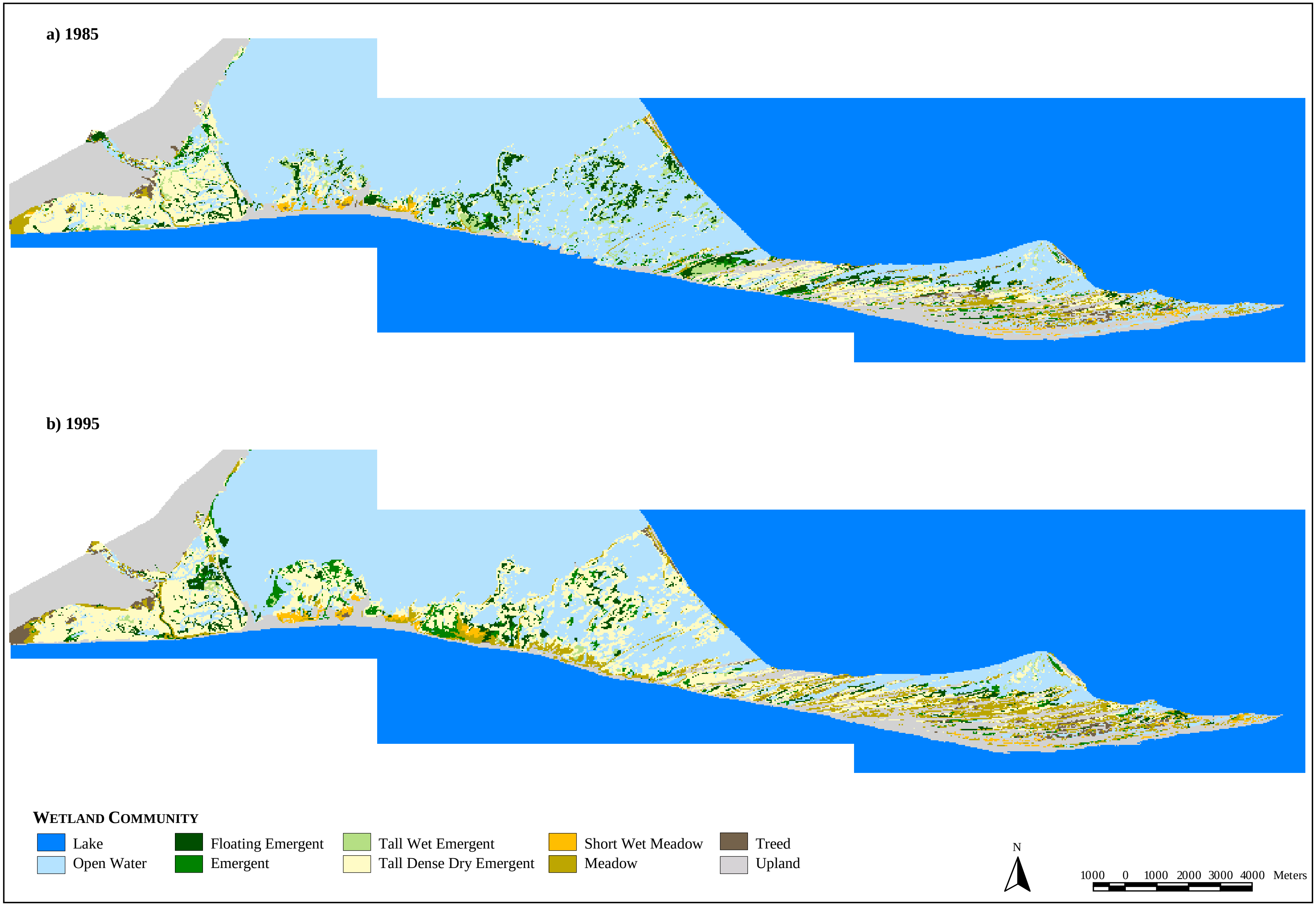


FIGURE E-3: SIMPLIFIED WETLAND COMMUNITY DATA, 1985-1995

APPENDIX F
Class Metrics

TABLE F-1: CLASS METRICS FOR LONG POINT, 1945-1999

YEAR *Mean Water Level*	1945 *174.19*	1955 *174.35*	1964 *173.61*	1978 *174.37*	1985 *174.73*	1995 *174.29*	1999 *174.11*
LAKE							
CA	17240.770	17312.443	17178.920	17445.130	17560.090	17535.280	17479.000
TA	31136.870	31136.858	31136.980	31136.863	31136.874	31137.020	31136.007
PLAND	55.371	55.601	55.172	56.027	56.396	56.317	56.138
LPI	55.371	55.601	55.172	56.027	56.396	56.317	56.138
NP	1	1	1	1	1	1	1
PD	0.003	0.003	0.003	0.003	0.003	0.003	0.003
MPS	17240.770	17312.443	17178.920	17445.130	17560.090	17535.280	17479.000
PSSD	0.000	0.000	0.000	0.000	0.000	0.000	0.000
PSCV	0.000	0.000	0.000	0.000	0.000	0.000	0.000
TE	67073.451	66092.516	68135.817	68306.301	69085.904	67212.086	71115.425
ED	2.154	2.123	2.188	2.194	2.219	2.159	2.284
MNN	0.000	0.000	0.000	0.000	0.000	0.000	0.000
MPI	0.000	0.000	0.000	0.000	0.000	0.000	0.000
LSI	1.441	1.417	1.466	1.459	1.471	1.432	1.517
MSI	3.050	3.030	3.080	3.060	3.070	3.030	3.120
AWMSI	3.050	3.030	3.080	3.060	3.070	3.030	3.120
MPFD	1.250	1.250	1.250	1.250	1.250	1.250	1.250
AWMPFD	1.250	1.250	1.250	1.250	1.250	1.250	1.250
IJI	22.340	28.790	17.650	23.740	39.520	30.930	28.470
OPEN WATER							
CA	5415.527	5632.808	5973.606	6977.494	7558.565	6356.960	6241.567
TA	31136.870	31136.858	31136.980	31136.863	31136.874	31137.020	31136.007
PLAND	17.393	18.090	19.185	22.409	24.275	20.416	20.046
LPI	15.869	15.578	16.500	19.960	20.790	17.511	17.334
NP	130	155	275	240	274	160	148
PD	0.418	0.498	0.883	0.771	0.880	0.514	0.475
MPS	41.658	36.341	21.722	29.073	27.586	39.731	42.173
PSSD	433.226	389.831	310.056	401.481	392.436	432.350	444.890
PSCV	1039.959	1072.703	1427.382	1380.941	1422.591	1088.193	1054.917
TE	331800.591	368537.558	497642.437	669063.410	757957.613	505471.725	484937.703
ED	10.656	11.836	15.982	21.488	24.343	16.234	15.575
MNN	100.740	89.430	72.830	57.290	37.880	54.080	69.230
MPI	31920.600	14591.030	21864.270	43069.230	46555.040	23180.620	30334.530
LSI	12.719	13.852	18.163	22.595	24.593	17.884	17.315
MSI	1.951	1.949	1.778	2.072	2.063	2.273	2.143
AWMSI	7.043	6.556	7.940	13.507	13.250	9.147	9.298
MPFD	1.389	1.392	1.403	1.423	1.449	1.436	1.423
AWMPFD	1.362	1.356	1.366	1.423	1.423	1.386	1.394
IJI	66.650	79.830	70.040	84.250	84.560	66.280	64.480
FLOATING EMERGENT							
CA	329.588	659.635	226.268	599.609	614.517	456.379	507.117
TA	31136.870	31136.858	31136.980	31136.863	31136.874	31137.020	31136.007
PLAND	1.059	2.119	0.727	1.926	1.974	1.466	1.629
LPI	0.116	0.115	0.096	0.089	0.108	0.111	0.065
NP	112	232	61	346	457	273	312
PD	0.360	0.745	0.196	1.111	1.468	0.877	1.002
MPS	2.943	2.843	3.709	1.733	1.345	1.672	1.625
PSSD	4.985	5.053	5.641	3.487	3.063	3.268	2.803
PSCV	169.385	177.735	152.090	201.212	227.732	195.455	172.492
TE	136287.964	256775.740	76786.585	314861.590	361631.009	267209.242	267277.406
ED	4.377	8.247	2.466	10.112	11.614	8.582	8.584
MNN	200.610	127.570	371.550	82.120	65.670	80.640	100.900
MPI	34.230	32.410	87.000	68.060	74.760	68.810	61.650
LSI	21.177	28.203	14.400	36.273	41.152	35.284	33.481
MSI	2.069	1.951	1.889	1.947	1.963	2.126	1.932
AWMSI	2.978	2.643	2.582	3.145	3.350	3.369	2.833

YEAR *Mean Water Level*	**1945** *174.19*	**1955** *174.35*	**1964** *173.61*	**1978** *174.37*	**1985** *174.73*	**1995** *174.29*	**1999** *174.11*
MPFD	1.402	1.401	1.375	1.422	1.442	1.436	1.419
MPFD	1.402	1.401	1.375	1.422	1.442	1.436	1.419
AWMPFD	1.398	1.385	1.381	1.422	1.434	1.437	1.417
IJI	65.710	68.380	62.810	78.620	70.430	59.860	57.100
EMERGENT							
CA	532.718	899.181	694.627	405.126	231.935	319.192	319.997
TA	31136.870	31136.858	31136.980	31136.863	31136.874	31137.020	31136.007
PLAND	1.711	2.888	2.231	1.301	0.745	1.025	1.028
LPI	0.095	0.229	0.139	0.091	0.026	0.150	0.138
NP	164	179	165	322	355	188	192
PD	0.527	0.575	0.530	1.034	1.140	0.604	0.617
MPS	5.097	9.924	6.039	2.786	1.078	4.145	4.064
PSSD	156.927	197.571	143.444	221.463	165.084	244.111	243.791
PSCV	186071.555	279003.185	264471.854	225861.434	174358.163	147246.936	132519.910
TE	5.976	8.961	8.494	7.254	5.600	4.729	4.256
ED	167.120	113.680	103.470	90.430	101.650	138.820	142.270
MNN	40.450	243.940	274.050	82.800	28.550	42.560	53.330
MPI	3.248	5.023	4.210	1.258	0.653	1.698	1.667
LSI	22.742	26.247	28.307	31.655	32.297	23.250	20.898
MSI	1.875	2.015	2.102	1.771	1.727	1.817	1.686
AWMSI	2.480	3.128	3.273	2.891	2.498	2.773	2.383
MPFD	1.386	1.385	1.389	1.429	1.436	1.413	1.405
AWMPFD	1.380	1.386	1.409	1.429	1.428	1.412	1.385
IJI	64.220	69.320	71.630	77.330	77.900	67.620	67.330
TALL WET EMERGENT							
CA	100.792	191.709	148.552	463.796	337.704	63.578	58.071
TA	31136.870	31136.858	31136.980	31136.863	31136.874	31137.020	31136.007
PLAND	0.324	0.616	0.477	1.490	1.085	0.204	0.187
LPI	0.090	0.201	0.105	0.081	0.110	0.025	0.020
NP	18	43	49	377	449	67	73
PD	0.058	0.138	0.157	1.211	1.442	0.215	0.234
MPS	5.600	4.458	3.032	1.230	0.752	0.949	0.795
PSSD	6.597	9.867	4.961	2.521	2.176	1.377	1.018
PSCV	117.804	221.332	163.621	204.959	289.362	145.100	128.050
TE	25508.803	50263.766	47612.187	231405.976	196659.601	36884.277	34554.825
ED	0.819	1.614	1.529	7.432	6.316	1.185	1.110
MNN	597.330	417.280	426.570	66.200	69.080	321.370	356.180
MPI	60.800	37.130	9.240	96.730	25.930	10.960	5.700
LSI	7.168	10.241	11.020	30.311	30.189	13.049	12.792
MSI	1.738	1.674	1.703	1.612	1.532	1.713	1.586
AWMSI	2.108	2.400	2.061	2.437	2.303	1.980	1.783
MPFD	1.342	1.357	1.367	1.392	1.414	1.408	1.397
AWMPFD	1.345	1.357	1.351	1.392	1.389	1.394	1.385
IJI	58.760	67.970	60.680	54.600	57.290	64.010	63.560
TALL DENSE DRY EMERGENT							
CA	2927.794	2619.266	2461.872	1431.805	1568.040	2492.748	2612.112
TA	31136.870	31136.858	31136.980	31136.863	31136.874	31137.020	31136.007
PLAND	9.403	8.412	7.907	4.598	5.036	8.006	8.389
LPI	1.178	1.384	1.300	0.756	0.526	0.597	0.470
NP	212	410	337	598	775	668	727
PD	0.681	1.317	1.082	1.921	2.489	2.145	2.335
MPS	13.810	6.388	7.305	2.394	2.023	3.732	3.593
PSSD	40.531	29.874	29.986	14.586	10.899	13.020	10.426
PSCV	293.490	467.658	410.486	609.273	538.754	348.875	290.175
TE	544337.373	609928.045	540436.217	489273.614	615246.590	804149.483	782713.702
ED	17.482	19.589	17.357	15.714	19.759	25.826	25.139
MNN	51.350	45.030	44.060	52.450	46.150	36.390	37.640
MPI	3827.600	1017.580	1247.420	581.210	729.090	1401.160	1381.080

YEAR	**1945**	**1955**	**1964**	**1978**	**1985**	**1995**	**1999**
Mean Water Level	*174.19*	*174.35*	*173.61*	*174.37*	*174.73*	*174.29*	*174.11*
LSI	28.379	33.619	30.726	36.476	43.829	45.435	43.202
MSI	2.054	1.804	1.842	1.663	1.693	1.809	1.706
AWMSI	3.957	3.861	3.755	4.095	4.863	3.847	3.124
MPFD	1.375	1.379	1.379	1.372	1.418	1.400	1.391
AWMPFD	1.372	1.361	1.367	1.372	1.352	1.368	1.342
IJI	75.620	80.740	70.100	76.340	75.850	75.400	75.240
SHORT WET MEADOW							
CA	68.968	89.221	17.260	46.960	91.573	109.592	77.717
TA	31136.870	31136.858	31136.980	31136.863	31136.874	31137.020	31136.007
PLAND	0.221	0.287	0.055	0.151	0.294	0.352	0.250
LPI	0.102	0.061	0.009	0.026	0.033	0.049	0.043
NP	30	44	20	47	107	49	17
PD	0.096	0.141	0.064	0.151	0.344	0.157	0.055
MPS	2.299	2.028	0.863	0.999	0.856	2.237	4.572
PSSD	5.907	3.971	0.815	1.651	1.494	3.501	4.346
PSCV	256.938	195.809	94.438	165.265	174.533	156.504	95.057
TE	28986.692	38673.417	9769.367	31151.311	65916.879	48997.836	27101.858
ED	0.931	1.242	0.314	1.000	2.117	1.574	0.870
MNN	118.800	119.160	300.600	251.540	151.160	78.020	619.510
MPI	11.290	16.110	10.760	29.230	41.310	107.040	118.420
LSI	9.846	11.550	6.634	12.823	19.432	13.203	8.672
MSI	2.032	1.847	1.595	1.886	1.950	2.056	2.184
AWMSI	3.168	2.754	1.683	2.731	2.731	2.534	2.447
MPFD	1.435	1.415	1.400	1.438	1.458	1.427	1.394
AWMPFD	1.414	1.404	1.378	1.438	1.446	1.405	1.385
IJI	58.720	63.550	54.060	73.780	65.400	67.110	76.820
MEADOW							
CA	1086.254	833.095	1052.151	840.401	458.934	887.075	893.418
TA	31136.870	31136.858	31136.980	31136.863	31136.874	31137.020	31136.007
PLAND	3.489	2.676	3.379	2.699	1.474	2.849	2.869
LPI	0.366	0.219	0.377	0.230	0.120	0.235	0.337
NP	198	268	320	481	614	402	314
PD	0.636	0.861	1.028	1.545	1.972	1.291	1.008
MPS	5.486	3.109	3.288	1.747	0.747	2.207	2.845
PSSD	13.897	6.815	9.138	5.033	2.286	5.895	8.170
PSCV	253.318	219.202	277.920	288.094	306.024	267.105	287.170
TE	335990.044	376671.959	494668.922	538772.446	371979.187	514546.695	426556.203
ED	10.791	12.097	15.887	17.303	11.947	16.525	13.700
MNN	81.290	72.580	51.760	34.270	42.800	44.470	68.440
MPI	151.500	219.000	229.890	257.720	60.130	275.250	405.400
LSI	28.758	36.814	43.020	52.427	48.982	48.735	40.257
MSI	2.288	2.198	2.321	2.247	2.044	2.356	2.181
AWMSI	3.442	3.867	4.693	4.801	3.290	4.761	4.792
MPFD	1.423	1.415	1.429	1.460	1.481	1.454	1.431
AWMPFD	1.388	1.434	1.451	1.460	1.441	1.462	1.445
IJI	61.640	71.830	64.940	75.620	76.580	69.750	68.380
TREED							
CA	45.374	219.298	125.568	184.286	137.638	167.929	151.079
TA	31136.870	31136.858	31136.980	31136.863	31136.874	31137.020	31136.007
PLAND	0.146	0.704	0.403	0.592	0.442	0.539	0.485
LPI	0.070	0.088	0.077	0.078	0.068	0.111	0.112
NP	31	109	79	235	203	125	107
PD	0.100	0.350	0.254	0.755	0.652	0.401	0.344
MPS	1.464	2.012	1.589	0.784	0.678	1.343	1.412
PSSD	4.016	4.567	3.268	2.169	1.730	3.667	4.101
PSCV	274.317	226.988	205.664	276.658	255.162	273.045	290.439
TE	14774.693	101819.494	63457.197	116149.893	102904.589	93454.512	69185.996
ED	0.475	3.270	2.038	3.730	3.305	3.001	2.222

YEAR	1945	1955	1964	1978	1985	1995	1999
Mean Water Level	*174.19*	*174.35*	*173.61*	*174.37*	*174.73*	*174.29*	*174.11*
MNN	723.400	222.990	242.010	109.870	93.910	132.180	174.120
MPI	3.720	81.680	67.840	52.090	37.610	83.360	39.850
LSI	6.187	19.396	15.975	24.136	24.743	20.344	15.879
MSI	1.490	1.894	1.919	1.688	1.806	1.966	1.776
AWMSI	1.629	3.170	2.584	2.614	2.889	2.920	2.465
MPFD	1.397	1.407	1.419	1.413	1.456	1.444	1.424
AWMPFD	1.323	1.414	1.402	1.413	1.460	1.420	1.392
IJI	50.130	65.460	61.210	70.050	79.660	72.570	71.020
UPLAND							
CA	3389.065	2680.214	3258.168	2742.240	2577.866	2748.285	2795.942
TA	31136.870	31136.858	31136.980	31136.863	31136.874	31137.020	31136.007
PLAND	10.884	8.608	10.464	8.807	8.279	8.826	8.980
LPI	10.406	7.916	9.725	3.936	2.989	6.400	6.505
NP	81	143	129	251	293	220	208
PD	0.260	0.459	0.414	0.806	0.941	0.707	0.668
MPS	41.840	18.743	25.257	10.925	8.798	12.492	13.442
PSSD	359.818	206.023	266.489	98.960	77.191	139.591	145.978
PSCV	859.986	1099.200	1055.109	905.812	877.370	1117.443	1085.984
TE	428620.440	419685.724	504429.980	502852.082	492292.383	500057.398	475099.503
ED	13.766	13.479	16.200	16.150	15.811	16.060	15.259
MNN	73.090	79.280	61.380	43.370	43.770	46.500	61.900
MPI	33046.590	17765.870	22816.090	8954.130	10417.470	19120.320	22637.760
LSI	20.770	22.868	24.929	27.088	27.352	26.908	25.346
MSI	2.248	1.941	2.133	2.004	1.973	1.969	1.957
AWMSI	17.159	16.667	18.910	10.616	9.938	17.134	16.077
MPFD	1.427	1.415	1.428	1.394	1.465	1.454	1.447
AWMPFD	1.475	1.472	1.481	1.394	1.396	1.436	1.439
IJI	76.380	80.040	74.190	76.810	88.370	75.620	75.170

APPENDIX G
Model Input Parameters

1.0 RULE-BASED MODEL

The rule-based model simulates wetland vegetation response according to the pre-existing wetland vegetation community, the tolerance depth ranges of the different wetland vegetation communities (synthesized from the literature) and adjacency of lake, open water and upland cells.

TABLE G-1: PARAMETERS FOR RULE-BASED MODEL

RUN NUMBER	MEAN LAKE LEVEL (M)	BASE YEAR	OUTPUT GRIDS		
			VERSION 1.0	VERSION 1.2	VERSION 1.3
1	174.35	1945	veg_17435_a	veg_17435_b	veg_17435_c
2	173.61	1955	veg_17361_a	veg_17361_b	veg_17361_c
3	174.37	1964	veg_17437_a	veg_17437_b	veg_17437_c
4	174.73	1978	veg_17473_a	veg_17473_b	veg_17473_c
5	174.29	1985	veg_17429_a	veg_17429_b	veg_17429_c
6	174.11	1995	veg_17411_a	veg_17411_b	veg_17411_c

2.0 PROBABILITY MODEL

The probability model simulates wetland vegetation response according to water depth ranges. The water depth ranges for rising and declining water level conditions were established based on the likelihood of wetland vegetation communities occurring in specific depth ranges in the 1985 (rising) and 1995 (declining) data sets.

TABLE G-2: PARAMETERS FOR PROBABILITY MODEL

RUN NUMBER	MEAN LAKE LEVEL (M)	WATER LEVEL CONDITION	OUTPUT GRIDS		
			VERSION 2.0	VERSION 2.1	VERSION 2.2
1	174.19	Rising	veg_17419_d	veg_17419_e	veg_17419_f
2	174.35	Rising	veg_17435_d	veg_17435_e	veg_17435_f
3	173.61	Declining	veg_17361_d	veg_17361_e	veg_17361_f
4	174.37	Rising	veg_17437_d	veg_17437_e	veg_17437_f
5	174.73	Rising	veg_17473_d	veg_17473_e	veg_17473_f
6	174.29	Declining	veg_17429_d	veg_17429_e	veg_17429_f
7	174.11	Declining	veg_17411_d	veg_17411_e	veg_17411_f

3.0 VEGETATION TRANSITION MODEL

The vegetation transition model simulates wetland vegetation response based on the likelihood of wetland vegetation communities changing to other wetland vegetation types for periods of rising and declining periods. Inter-community transitions for rising water level conditions were derived from changes that occurred between 1978 and 1985; transitions for declining water level conditions were derived from changes between 1985 and 1995. Note this model does not actually use the mean lake level for any calculations; it is just used as a reference.

TABLE G-3: PARAMETERS FOR TRANSITION PROBABILITY MODEL

RUN NUMBER	MEAN LAKE LEVEL (M)	BASE YEAR	WATER LEVEL CONDITION	OUTPUT GRID
				VERSION 3.0
1	174.35	1945	Rising	veg_17435_g
2	173.61	1955	Declining	veg_17361_g
3	174.37	1964	Rising	veg_17437_g
4	174.73	1978	Rising	veg_17473_g
5	174.29	1985	Declining	veg_17429_g
6	174.11	1995	Declining	veg_17411_g

APPENDIX H
Rule-Based Modelling Results, 1945-1995

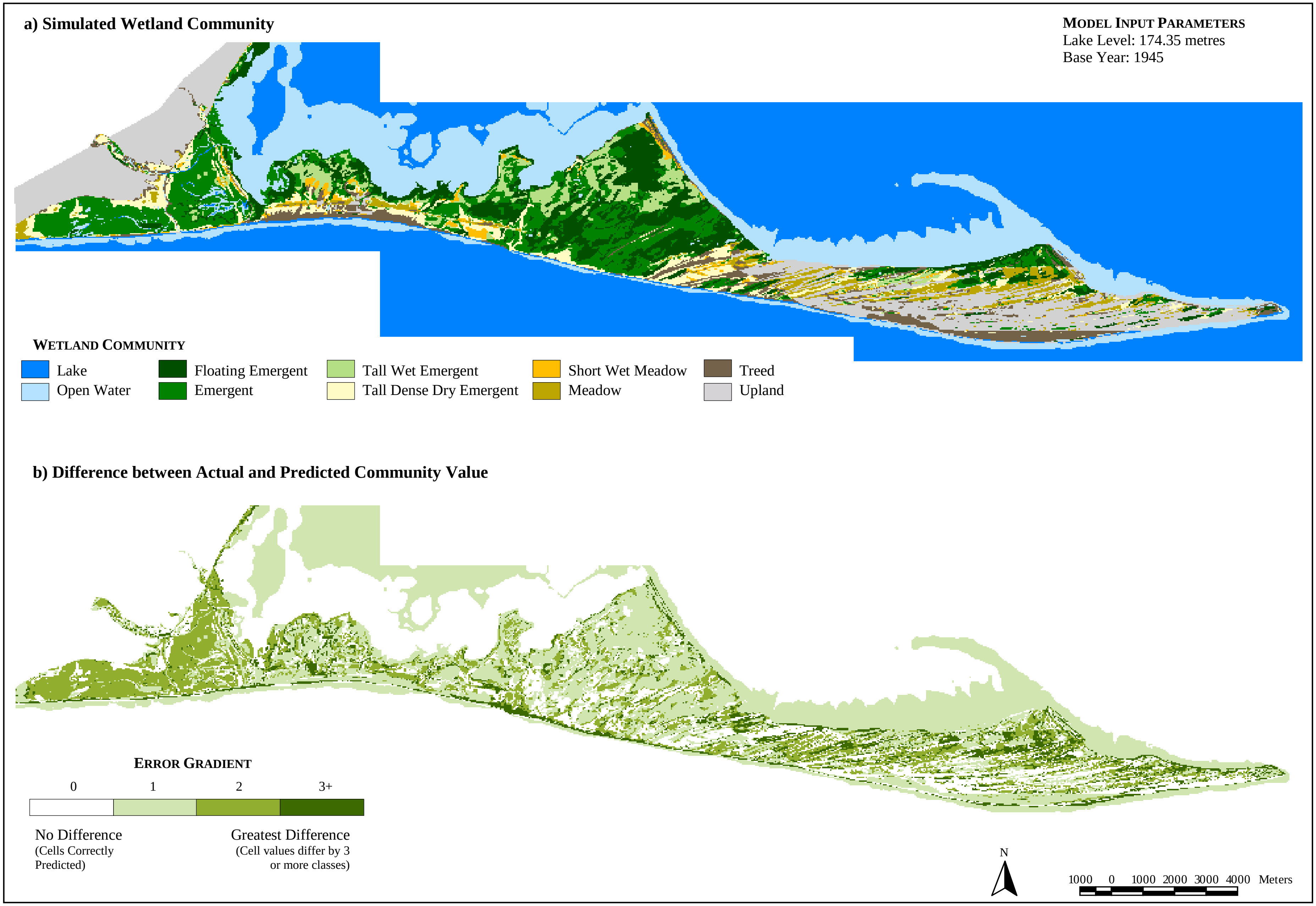


FIGURE H-1: RESULTS OF THE RULE-BASED MODEL V1.0, 1955

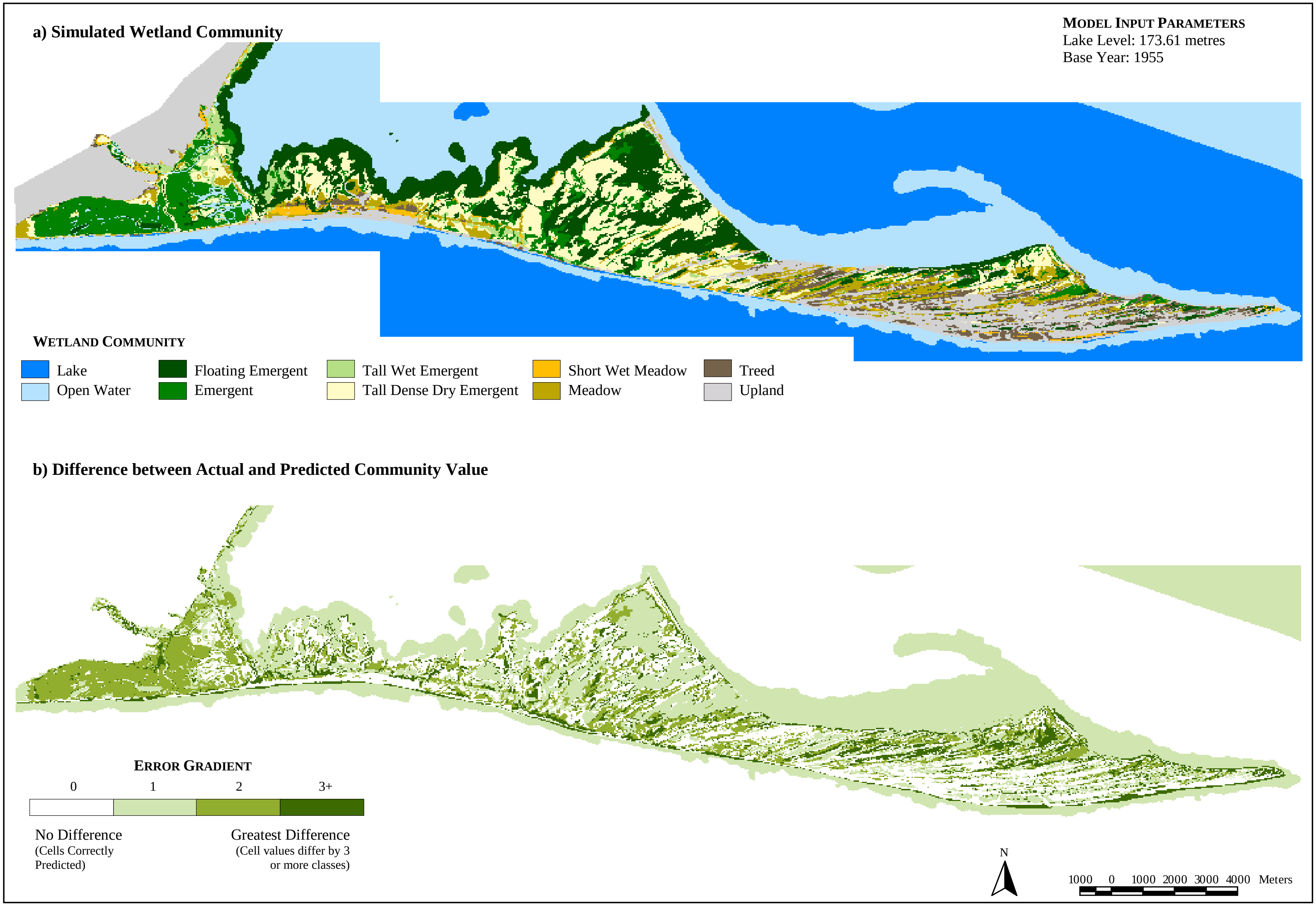


FIGURE H-2: RESULTS OF THE RULE-BASED MODEL v1.0, 1964

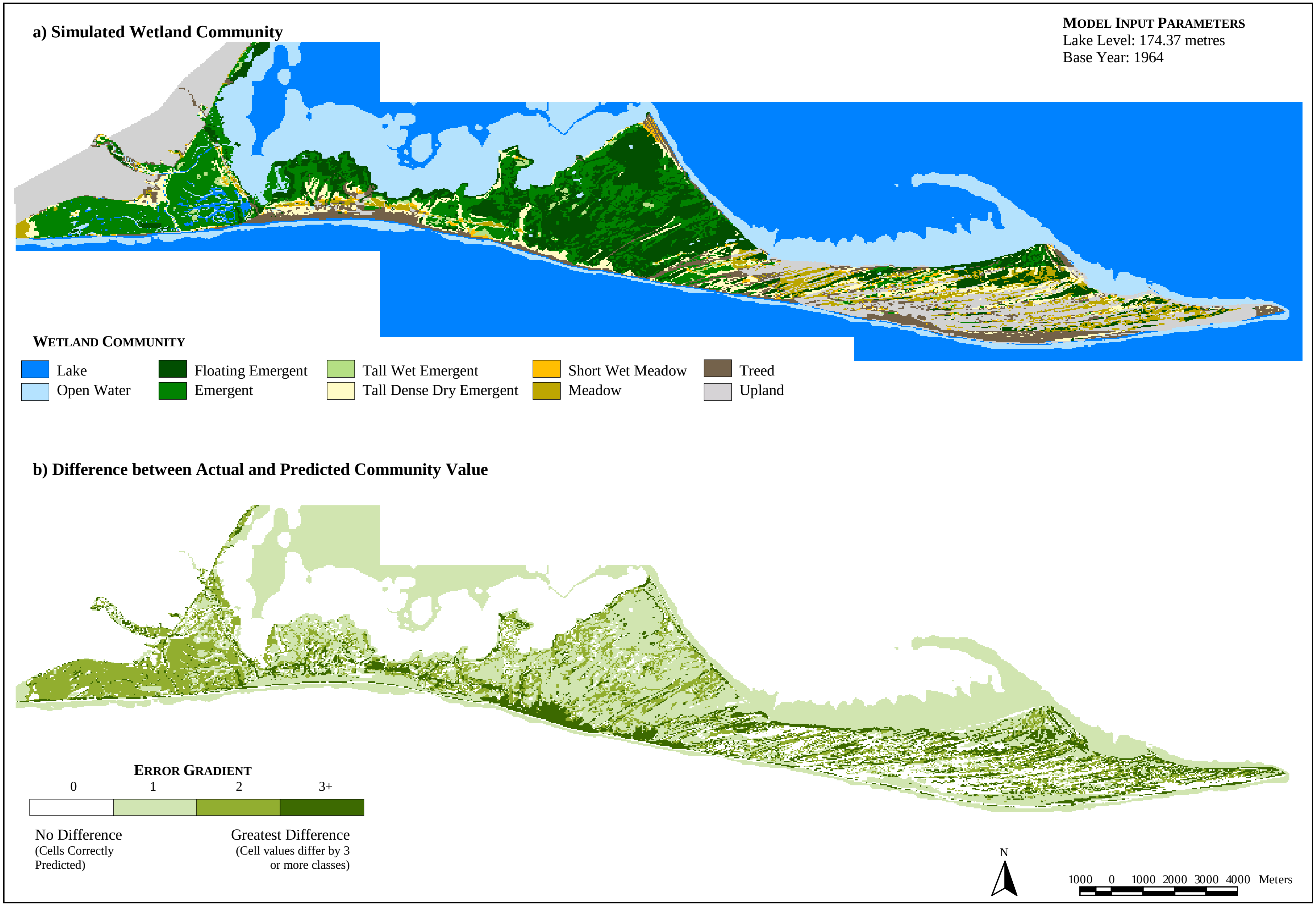


FIGURE H-3: RESULTS OF THE RULE-BASED MODEL V1.0, 1978

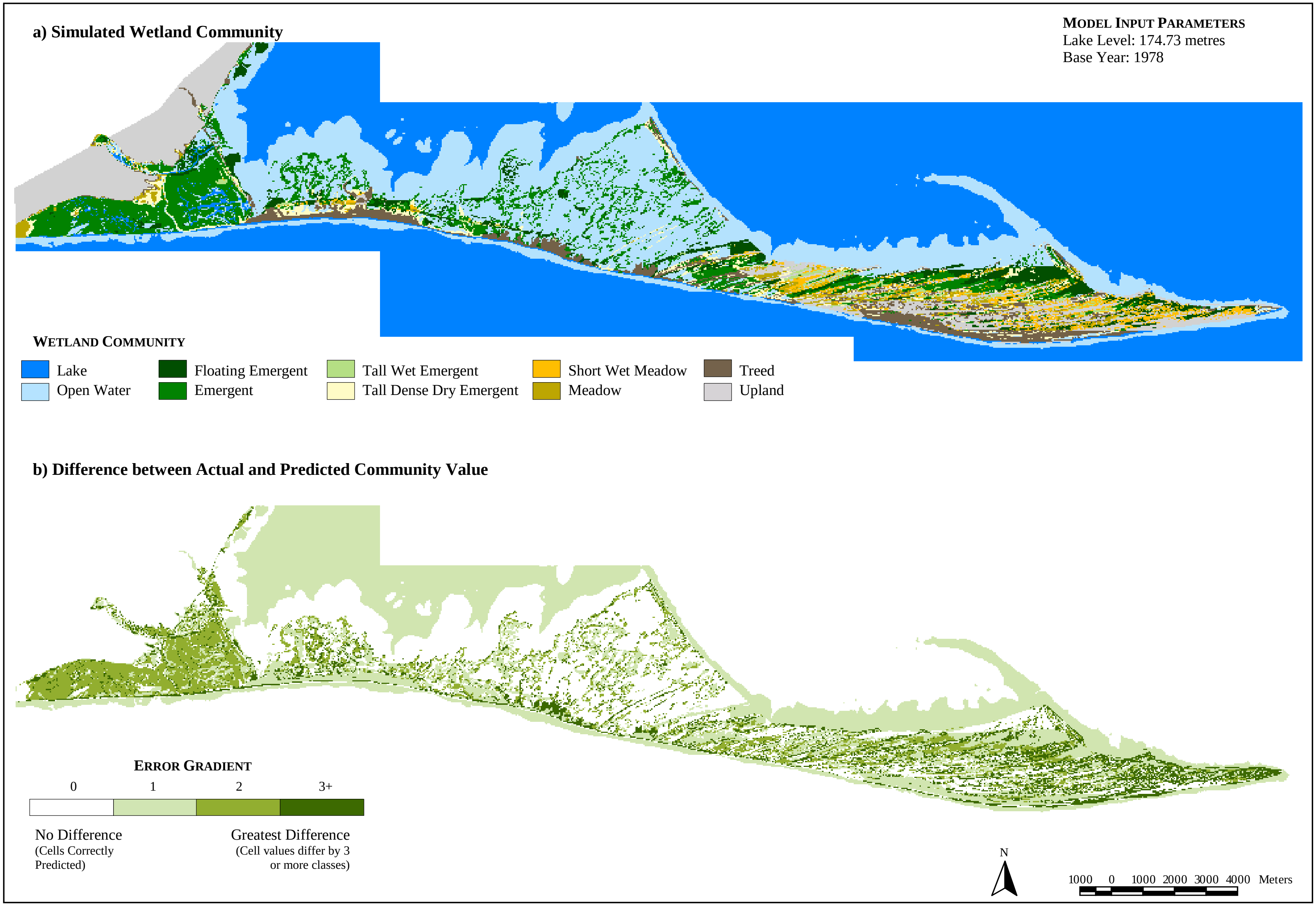


FIGURE H-4: RESULTS OF THE RULE-BASED MODEL v1.0, 1985

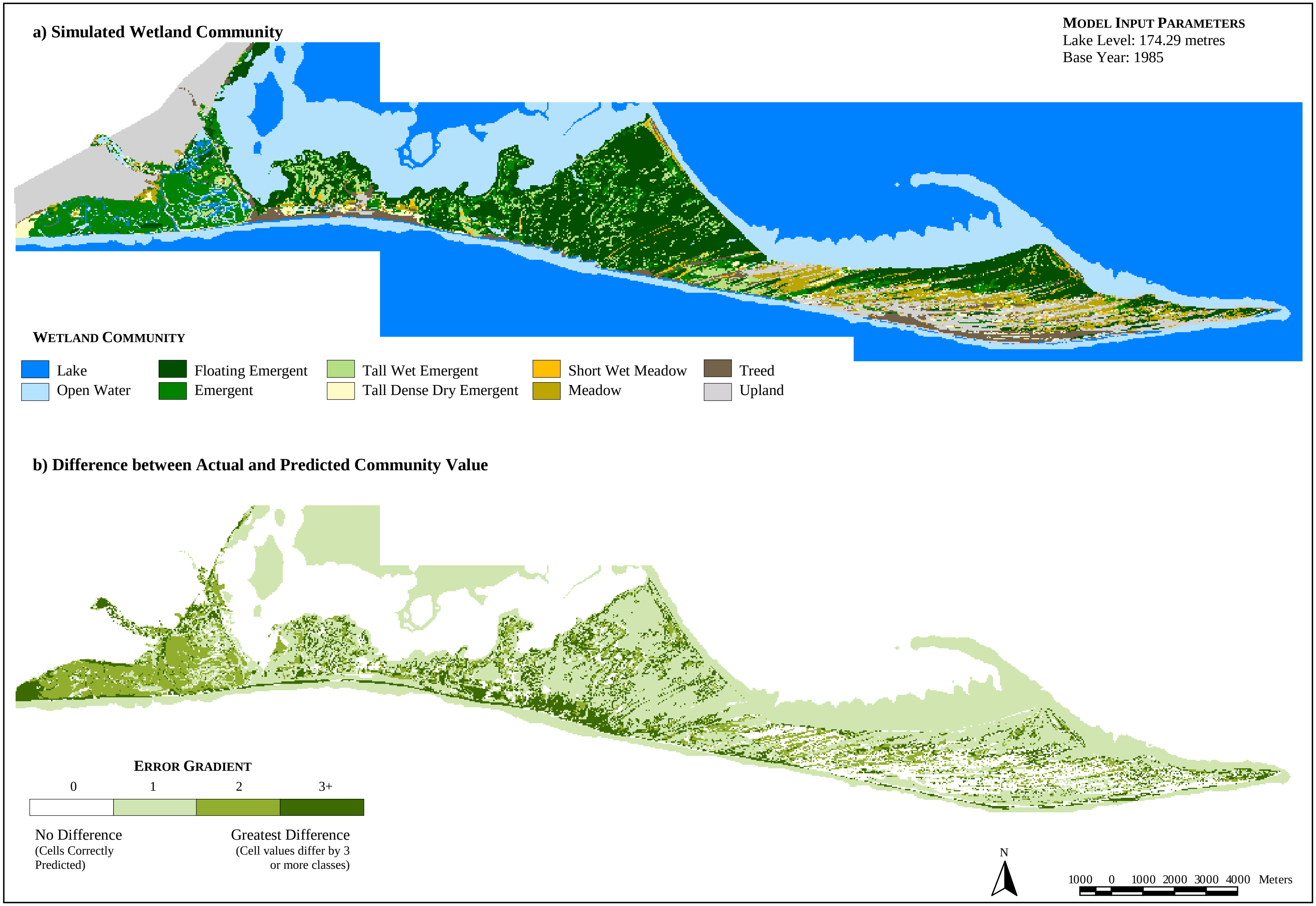


FIGURE H-5: RESULTS OF THE RULE-BASED MODEL v1.0, 1995

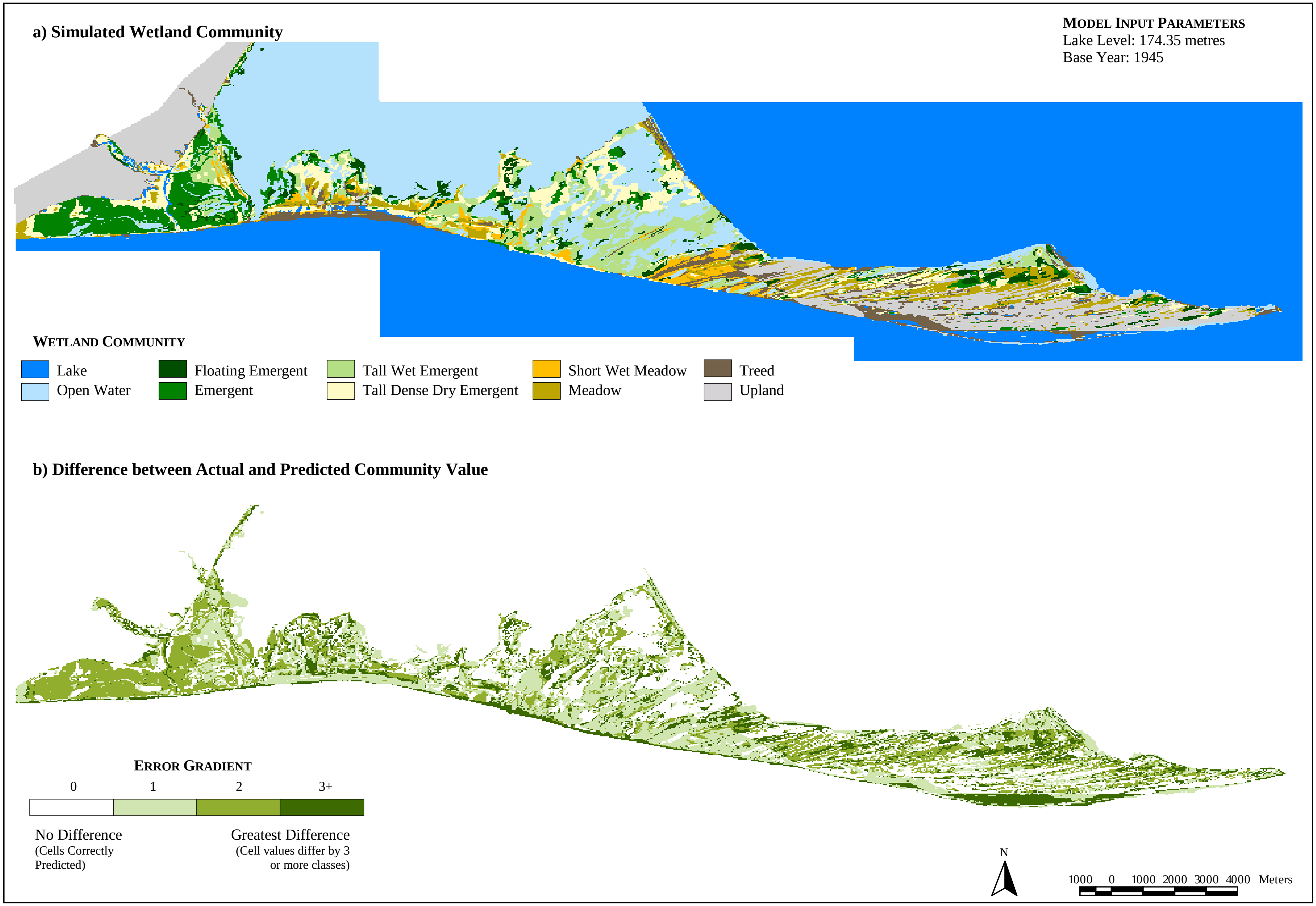


FIGURE H-6: RESULTS OF THE RULE-BASED MODEL v1.1, 1955

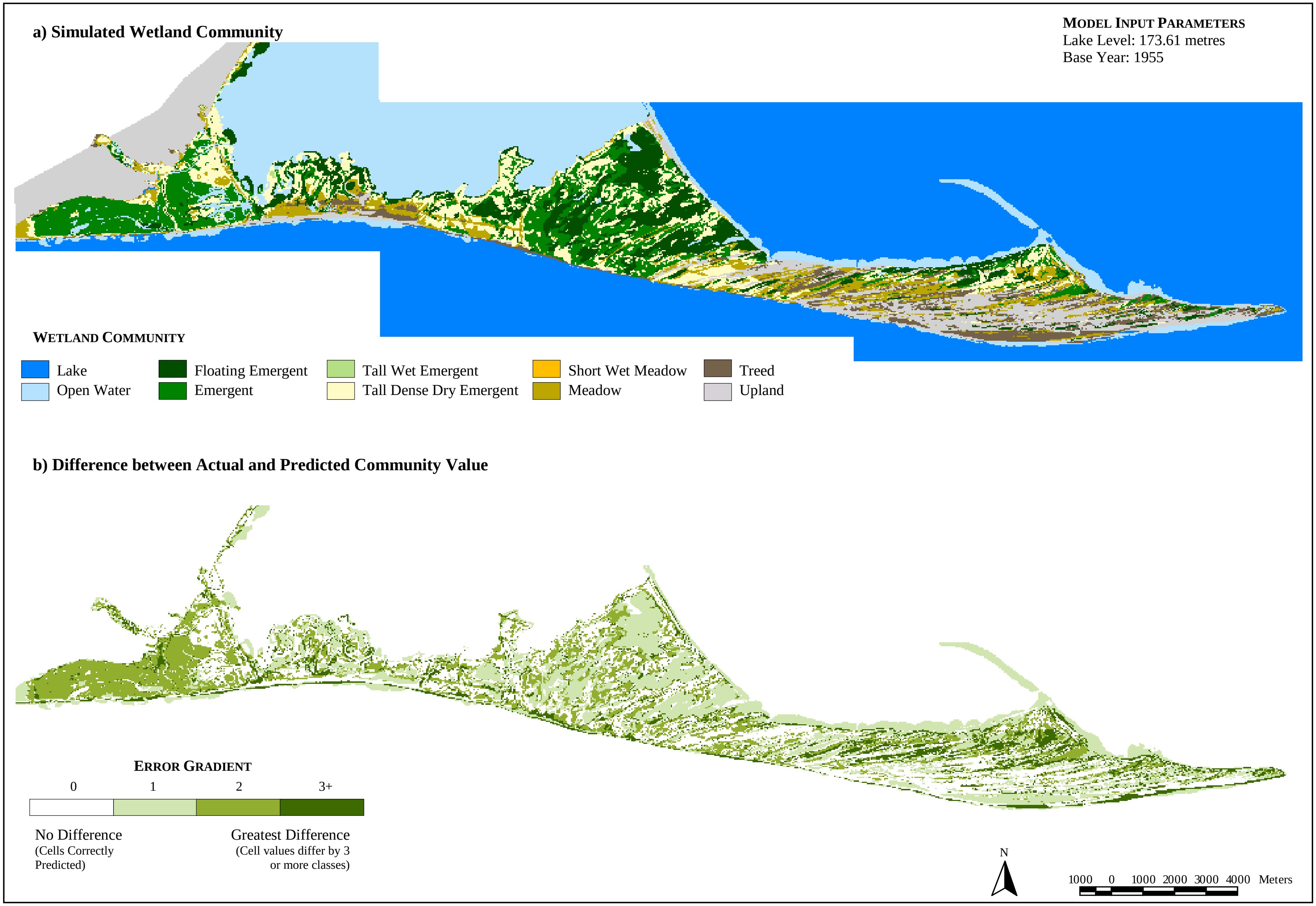


FIGURE H-7: RESULTS OF THE RULE-BASED MODEL v1.1, 1964

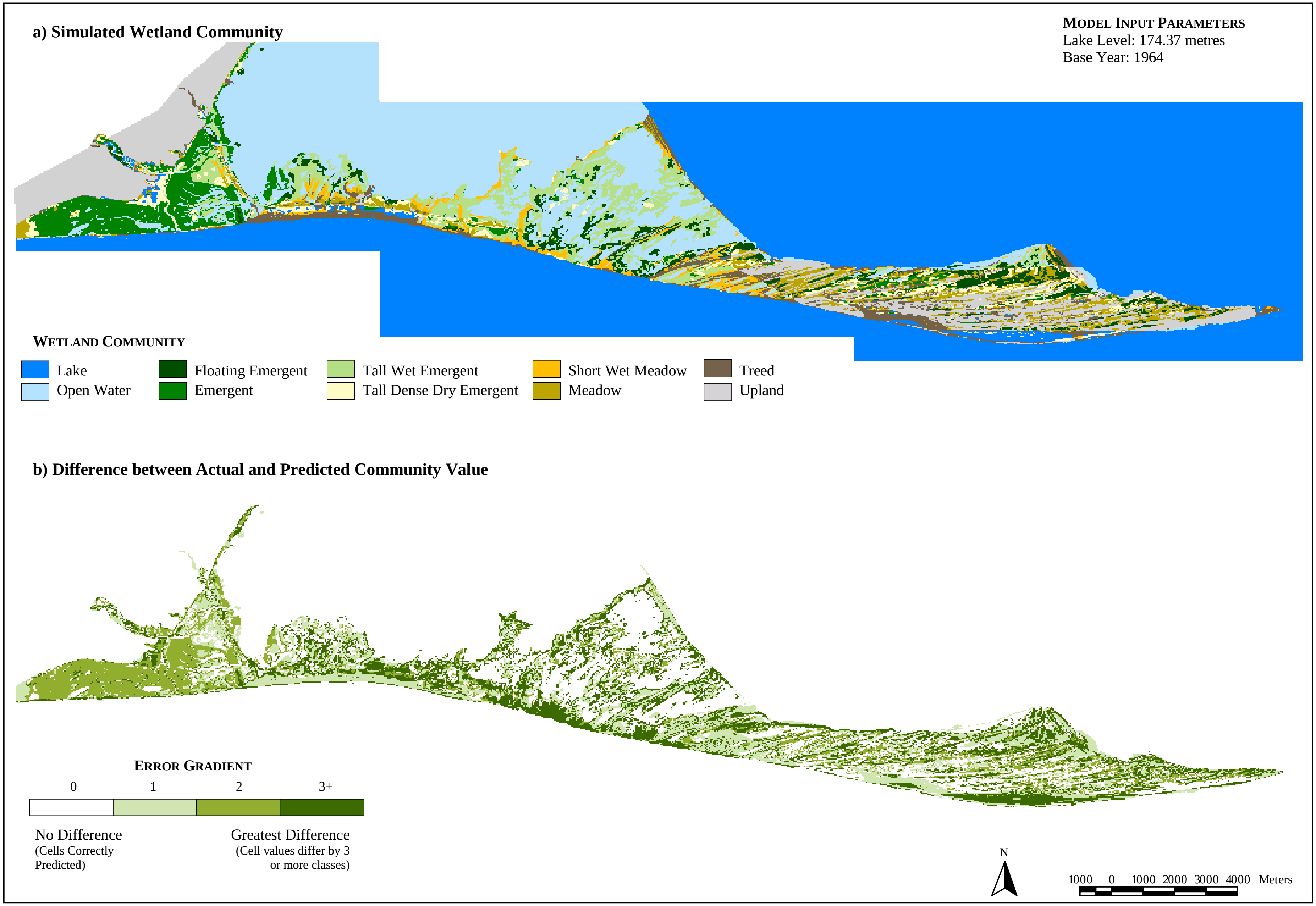


FIGURE H-8: RESULTS OF THE RULE-BASED MODEL v1.1, 1978

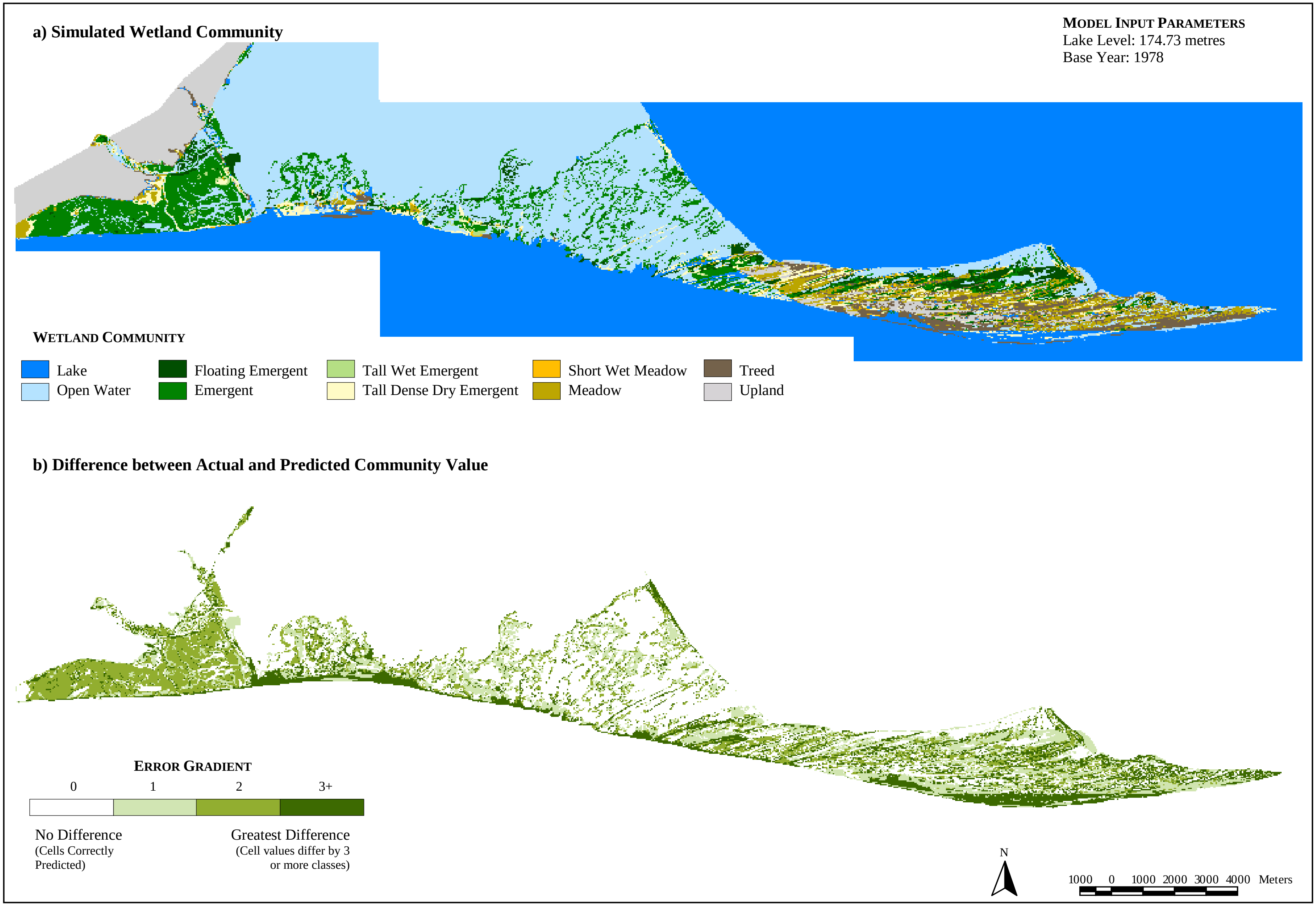


Figure H-9: Results of the Rule-Based Model v1.1, 1985

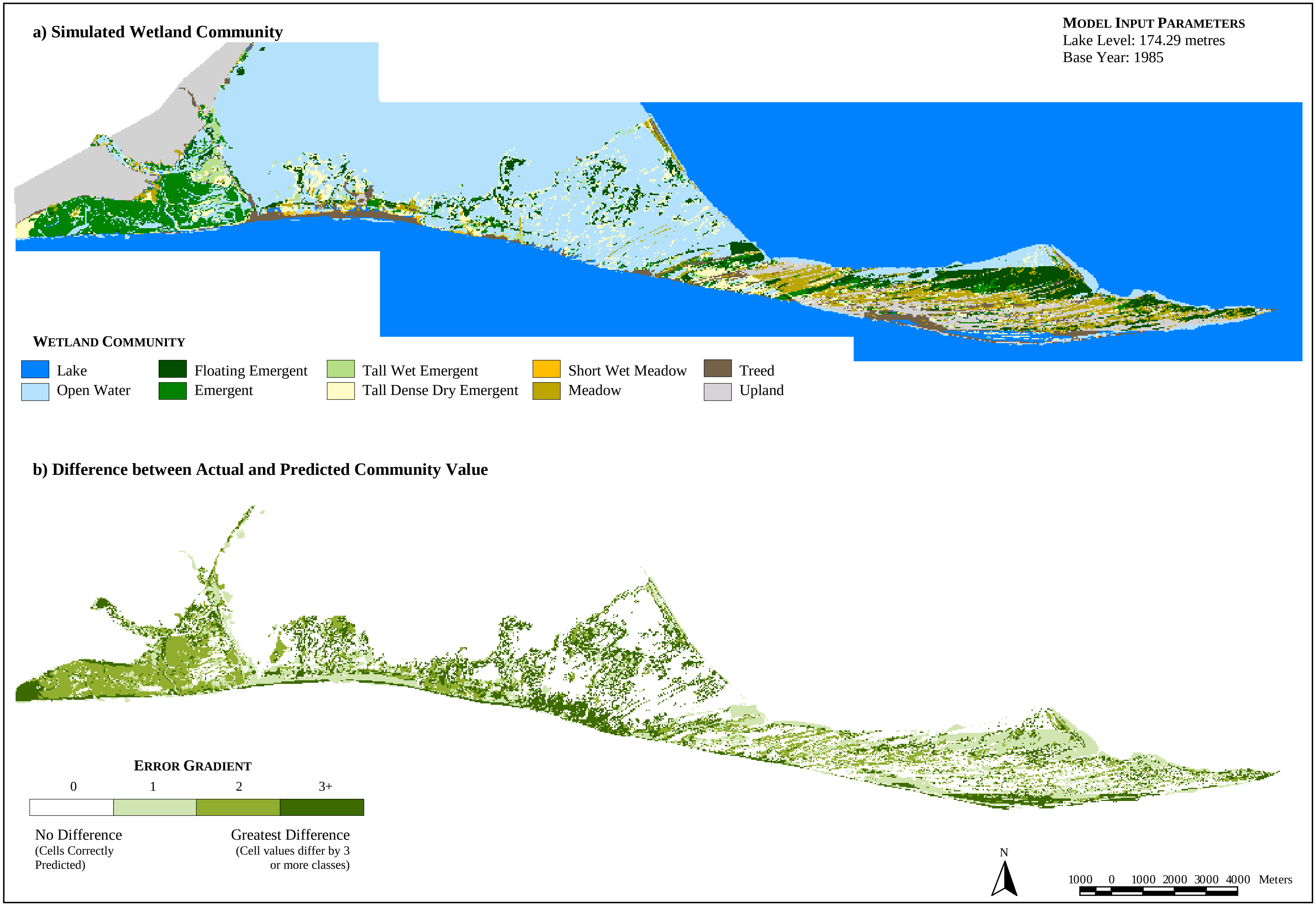


FIGURE H-10: RESULTS OF THE RULE-BASED MODEL v1.1, 1995

APPENDIX I
Probability Modelling Results, 1945-1995

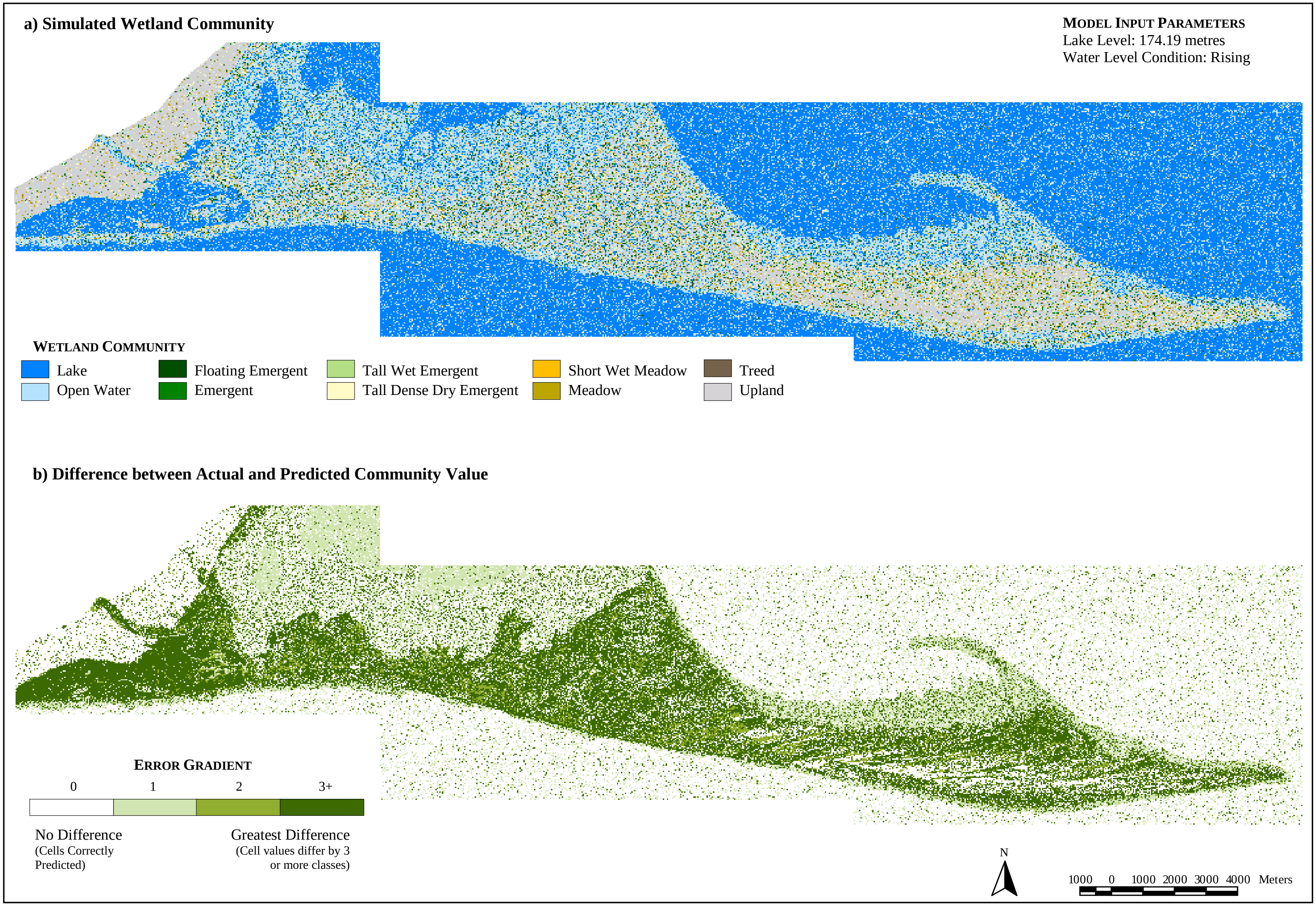


Figure I-1: Results of the Probability Model v2.0, 1945

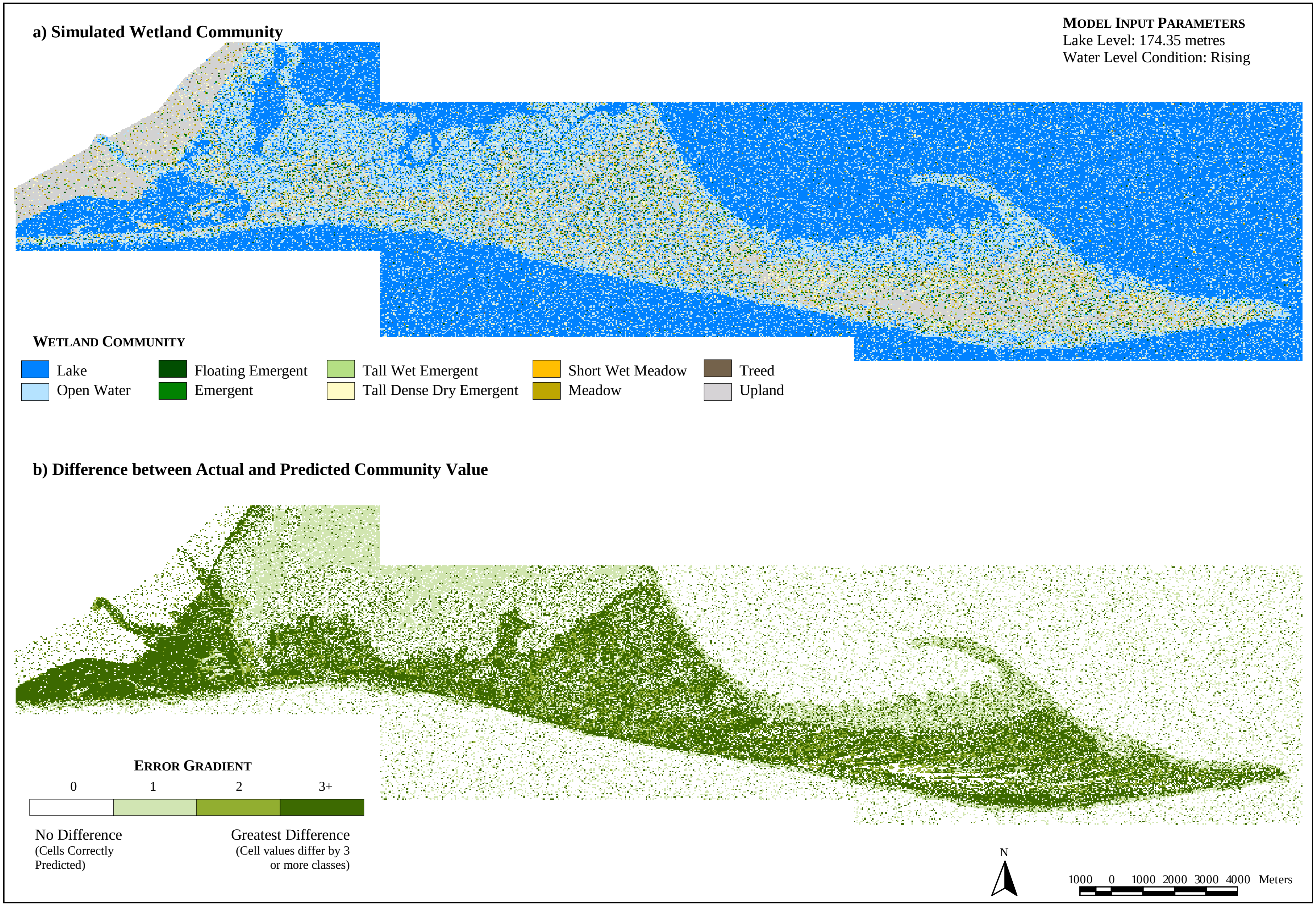


FIGURE I-2: RESULTS OF THE PROBABILITY MODEL v2.0, 1955

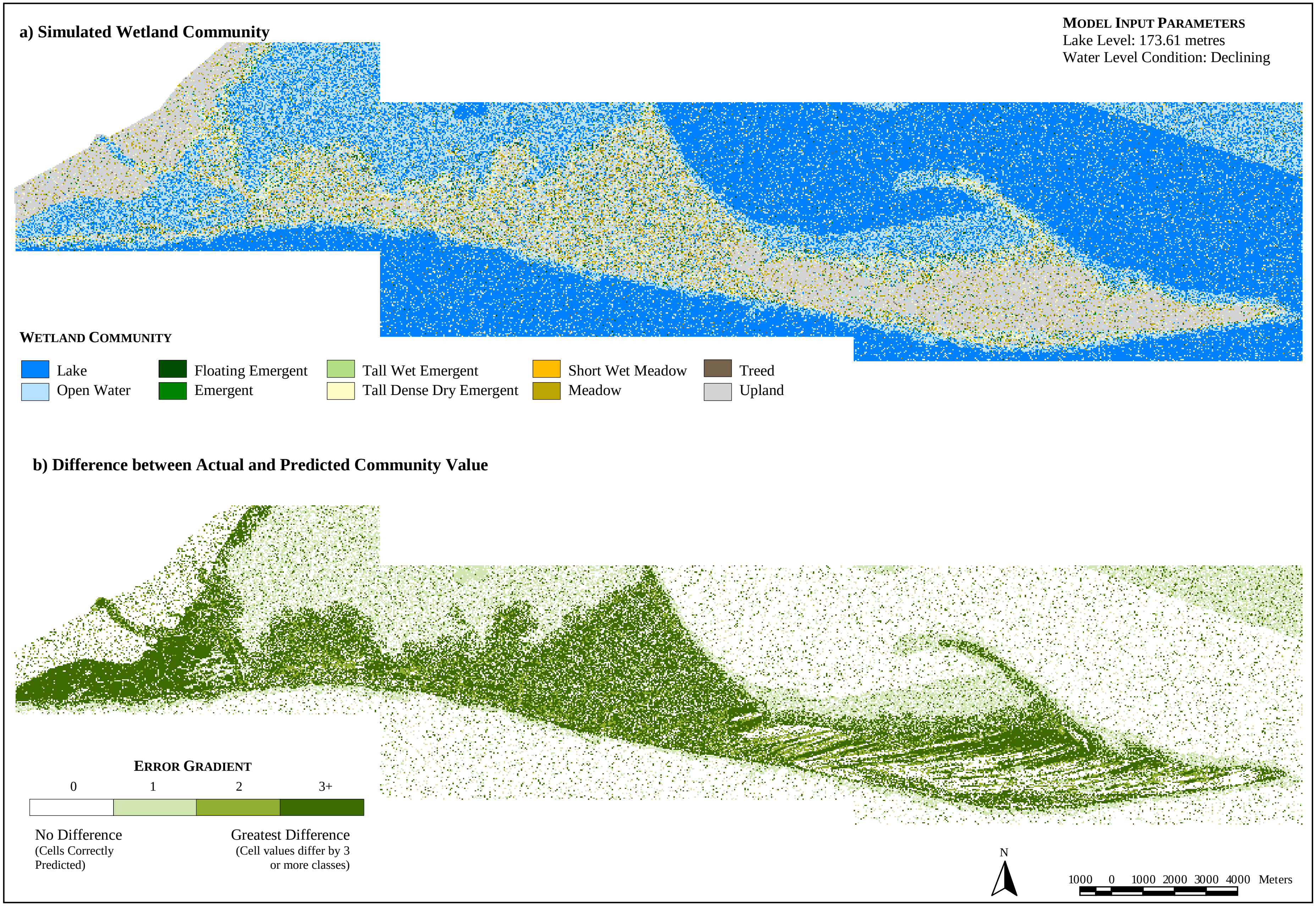


FIGURE I-3: RESULTS OF THE PROBABILITY MODEL v2.0, 1964

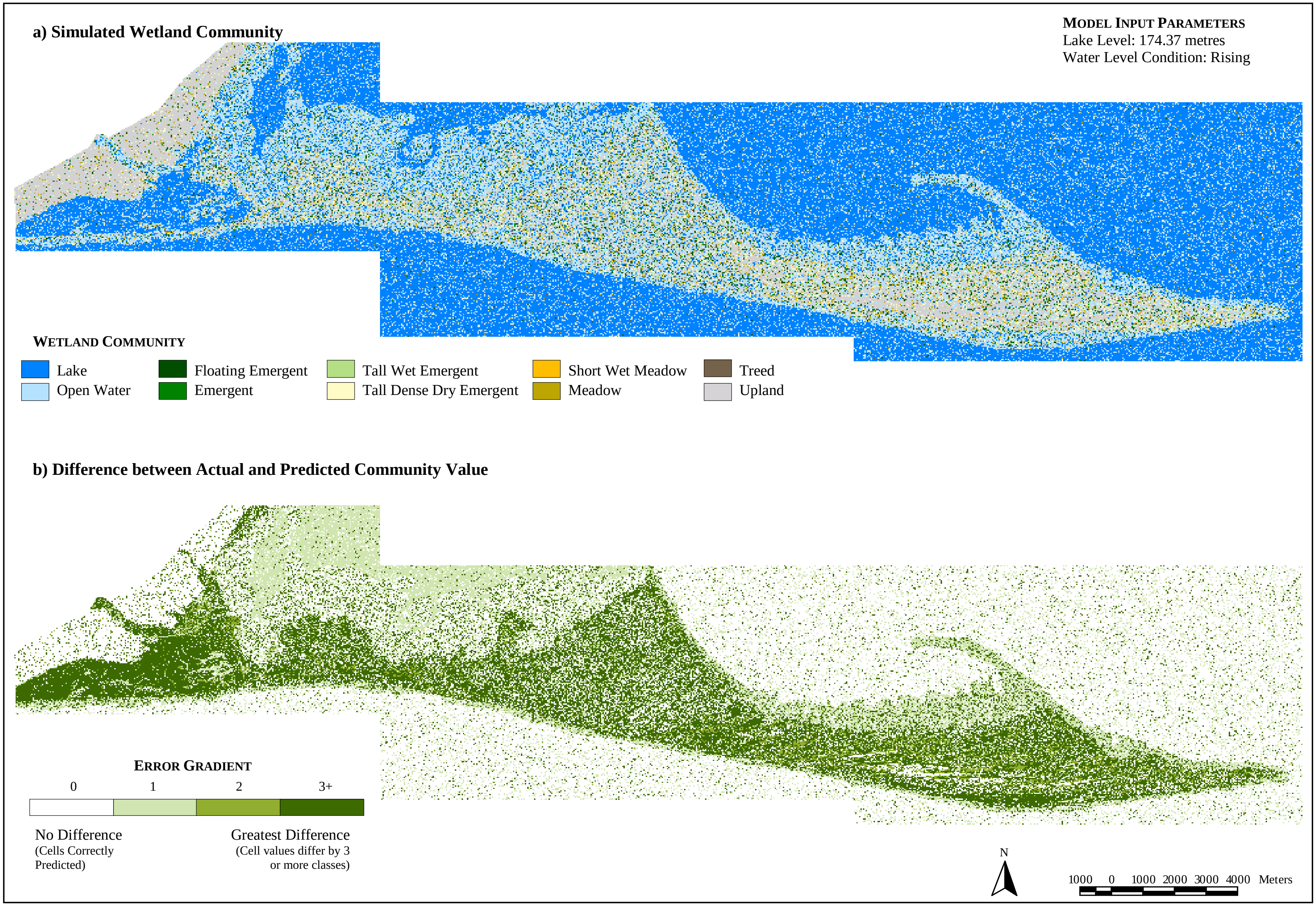


FIGURE I-4: RESULTS OF THE PROBABILITY MODEL v2.0, 1978

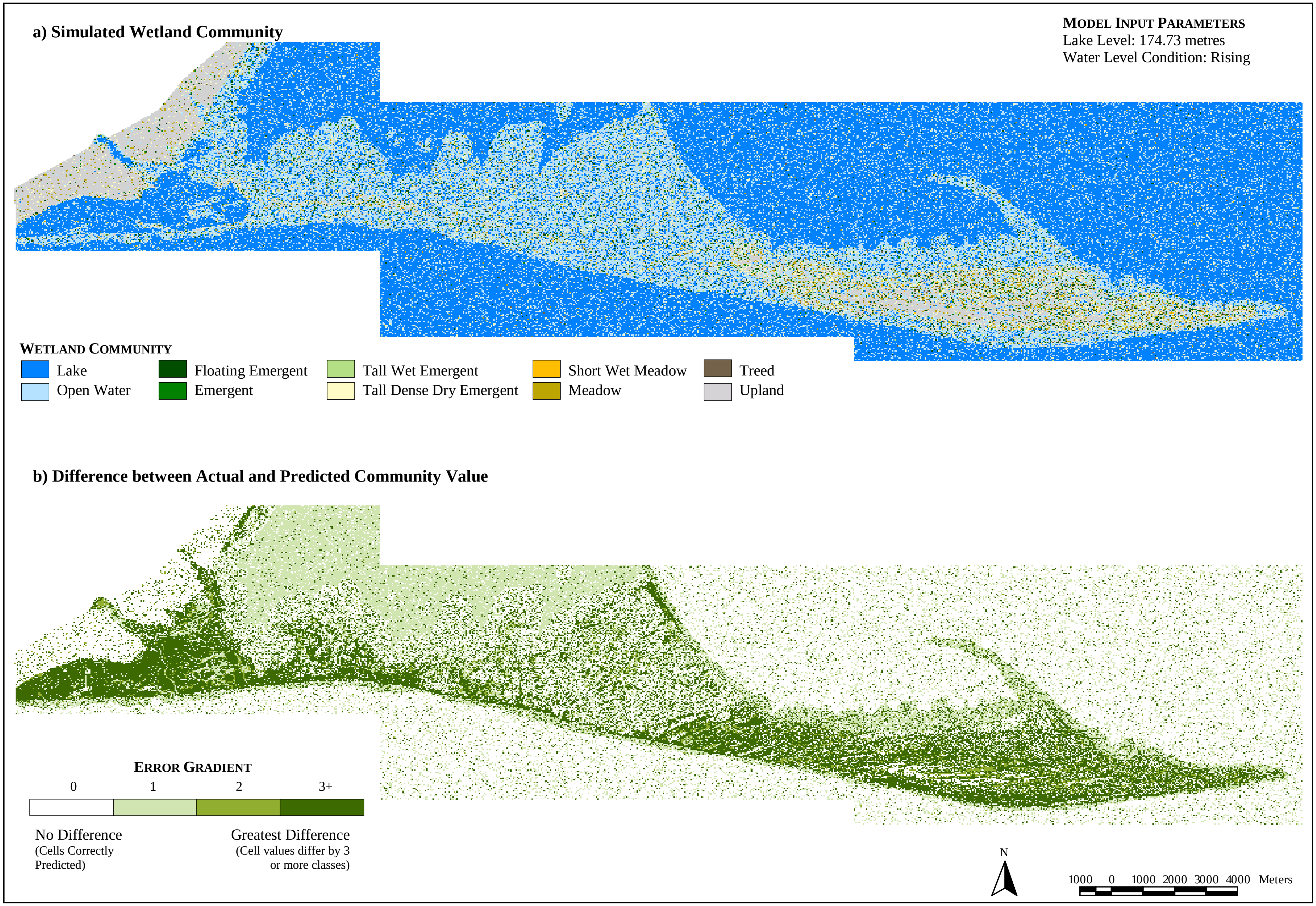


FIGURE I-5: RESULTS OF THE PROBABILITY MODEL v2.0, 1985

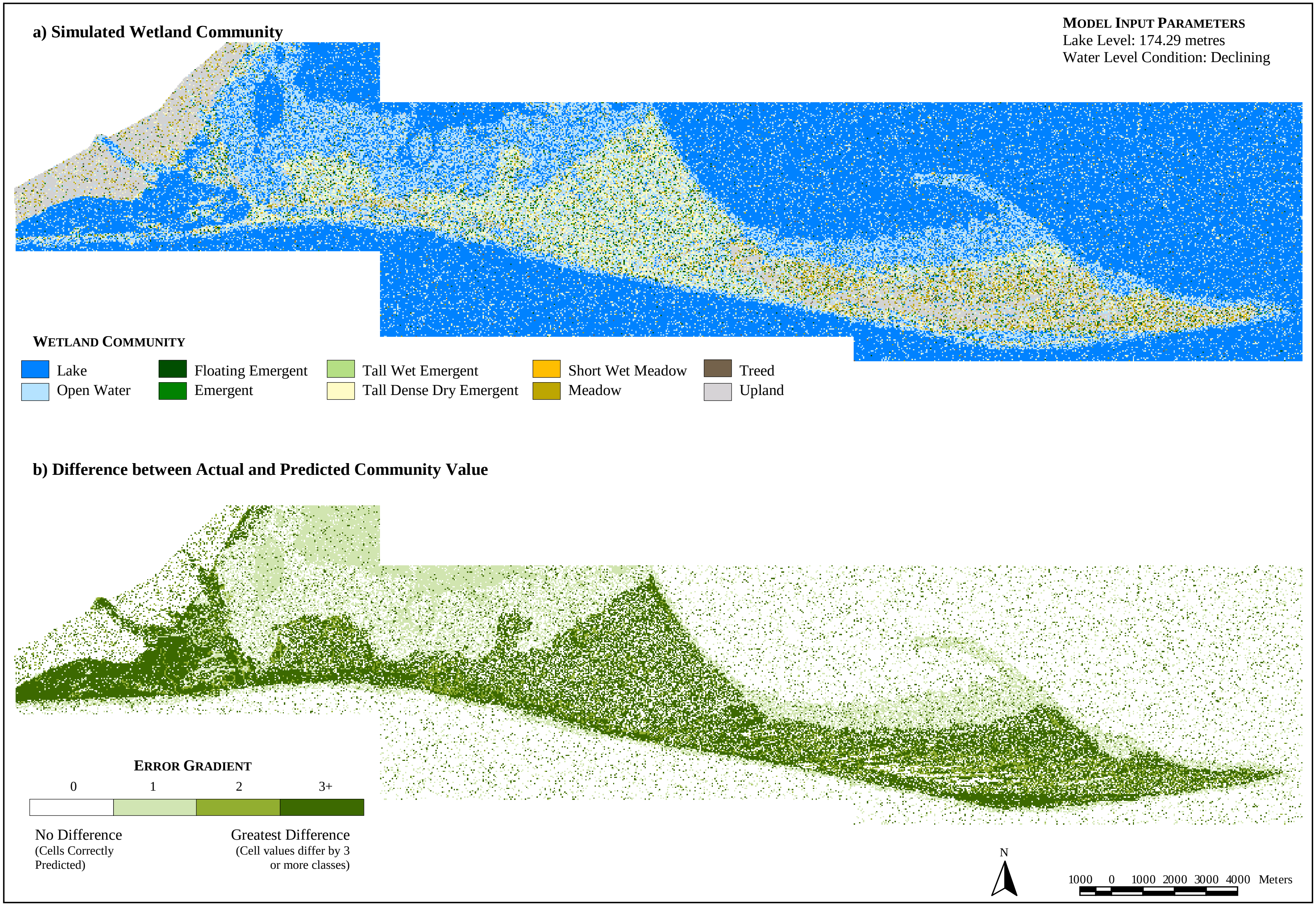


FIGURE I-6: RESULTS OF THE PROBABILITY MODEL v2.0, 1995

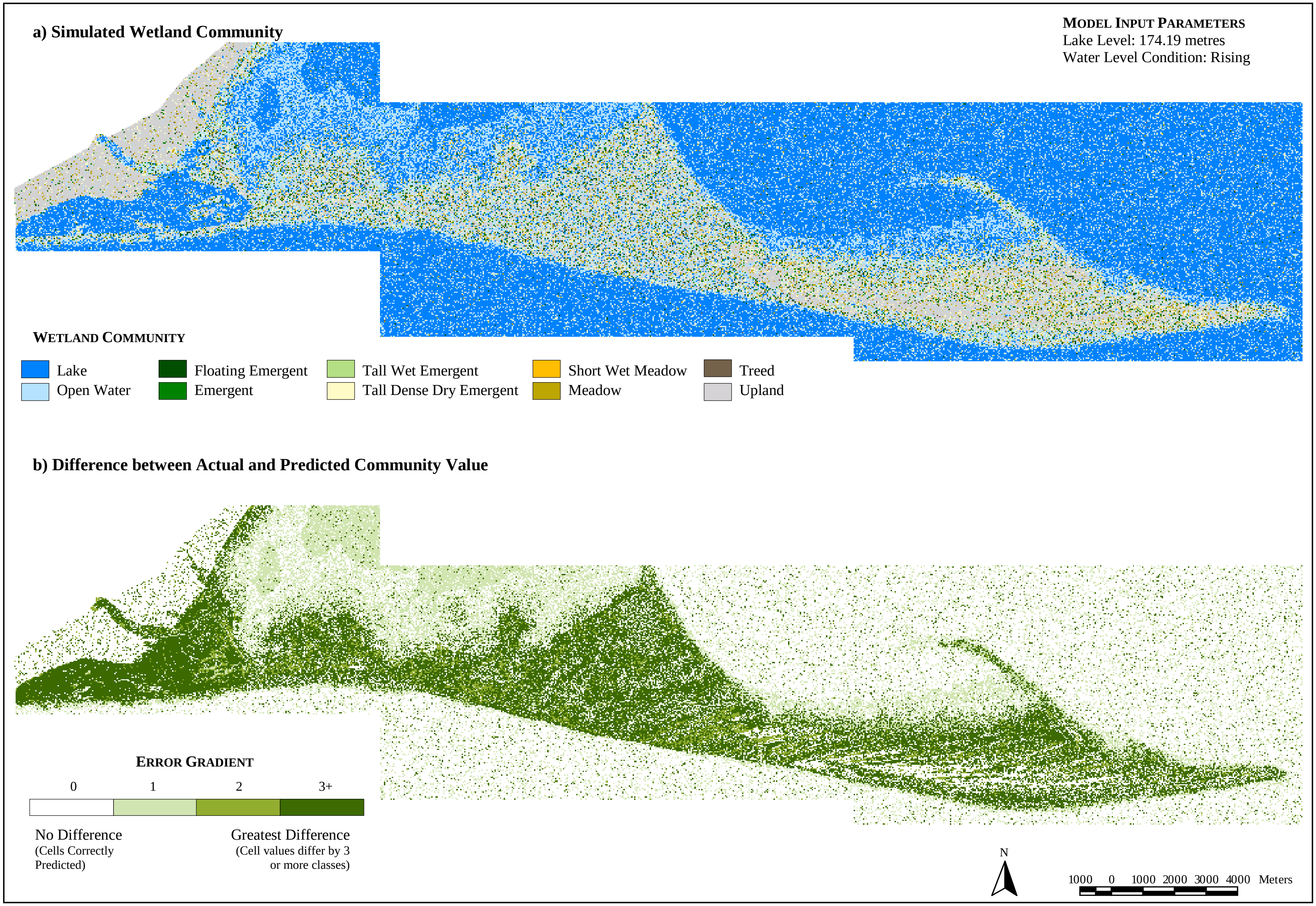


FIGURE I-7: RESULTS OF THE PROBABILITY MODEL v2.1, 1945

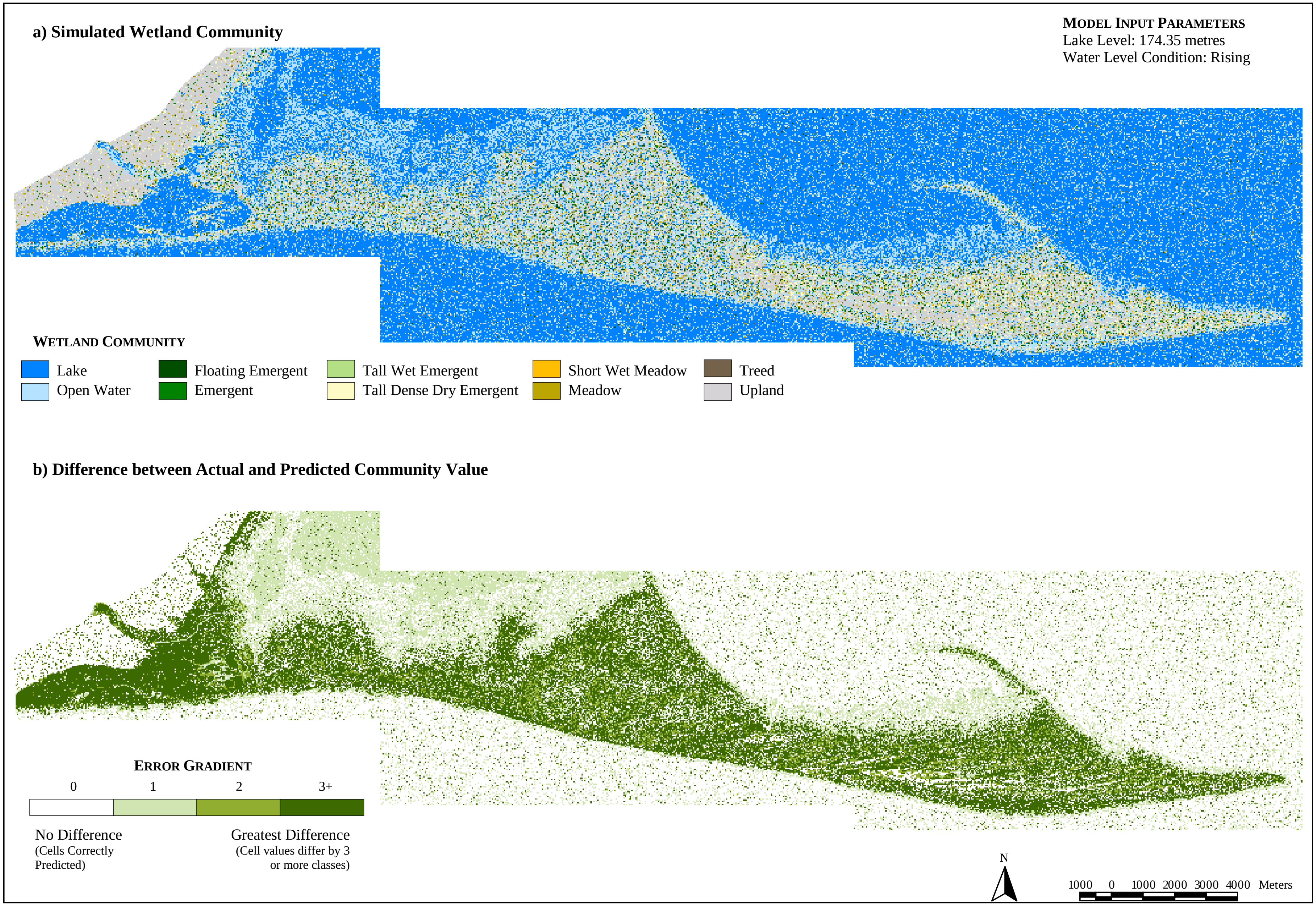


FIGURE I-8: RESULTS OF THE PROBABILITY MODEL v2.1, 1955

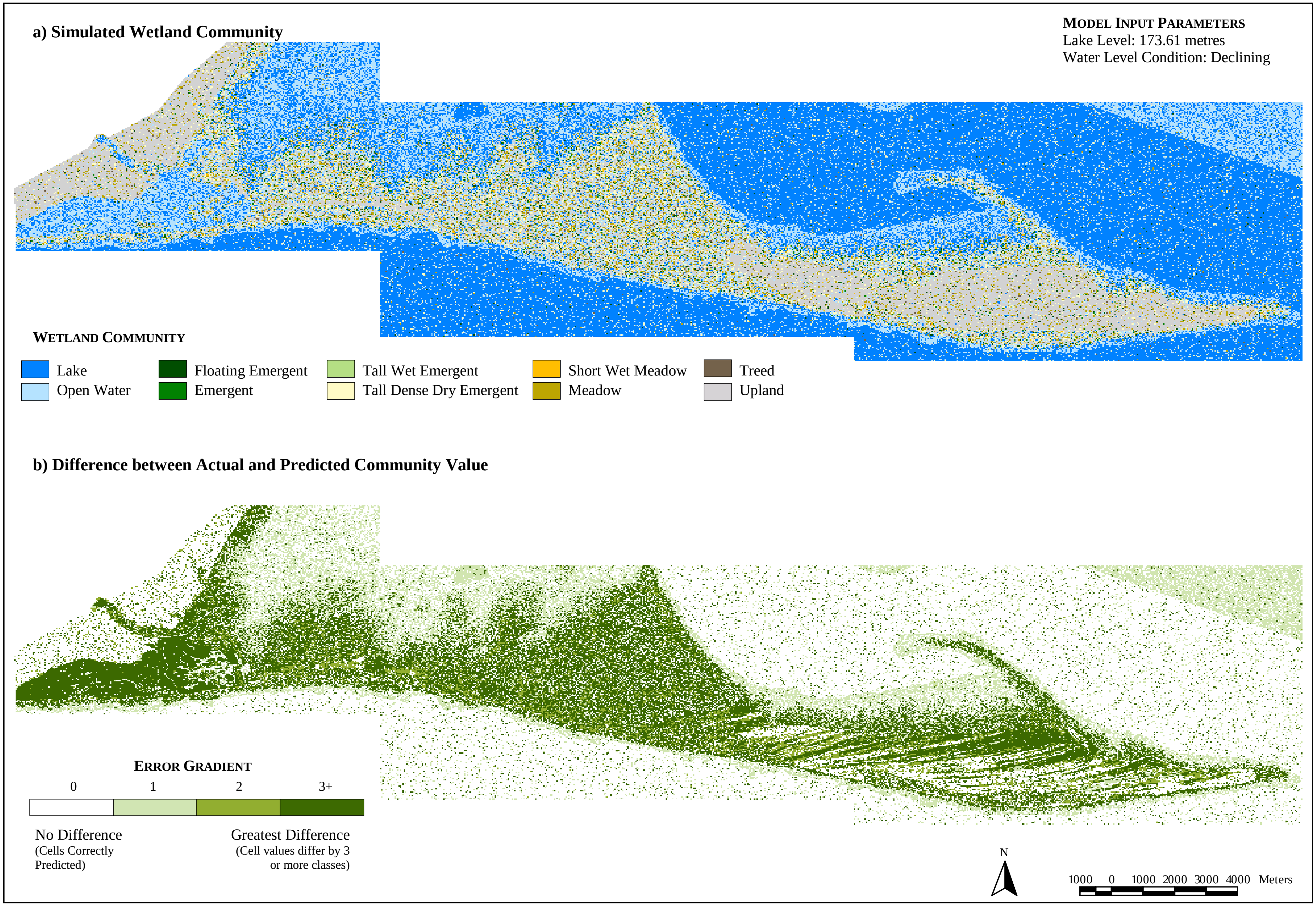


FIGURE I-9: RESULTS OF THE PROBABILITY MODEL v2.1, 1964

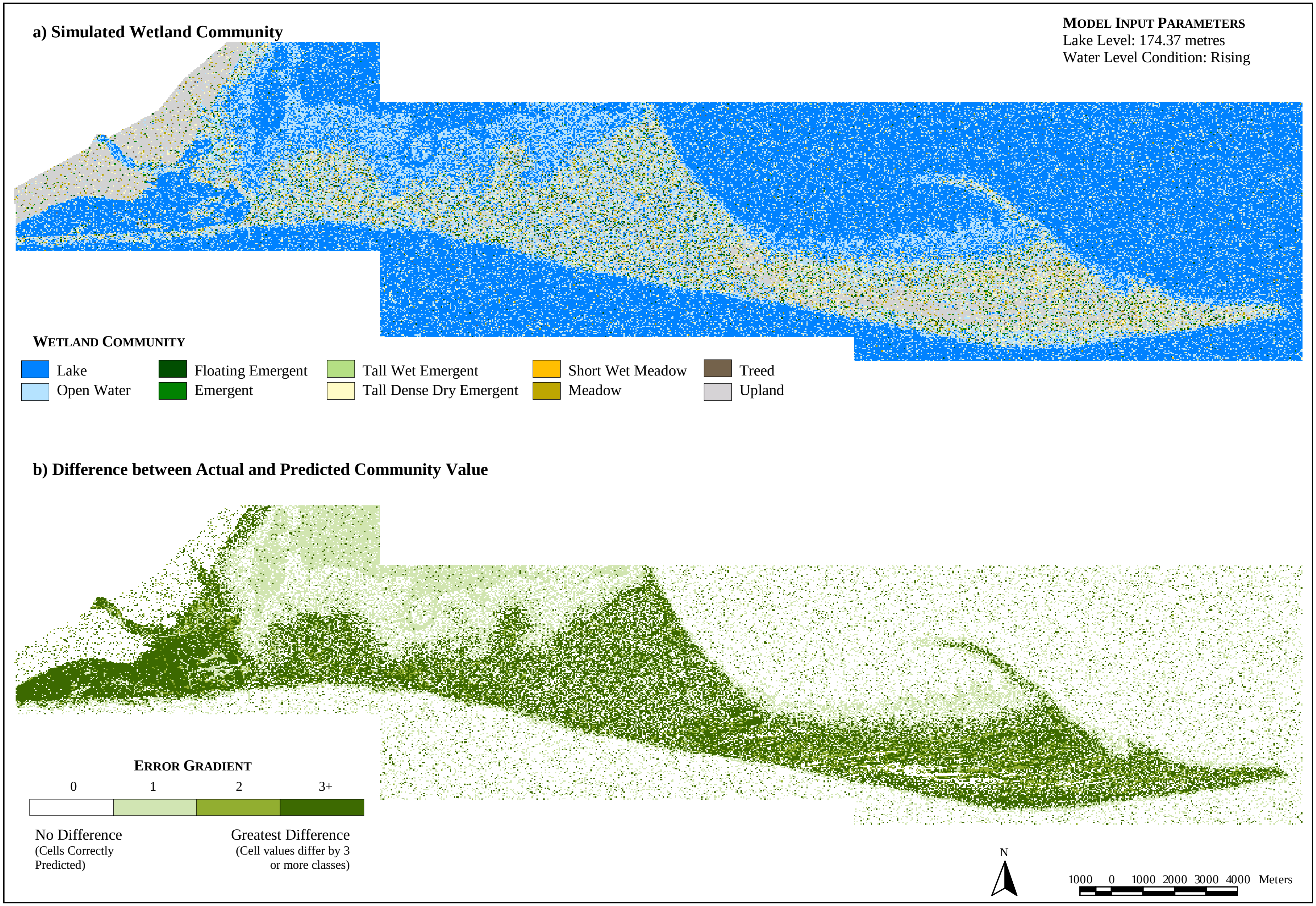


FIGURE I-10: RESULTS OF THE PROBABILITY MODEL V2.1, 1978

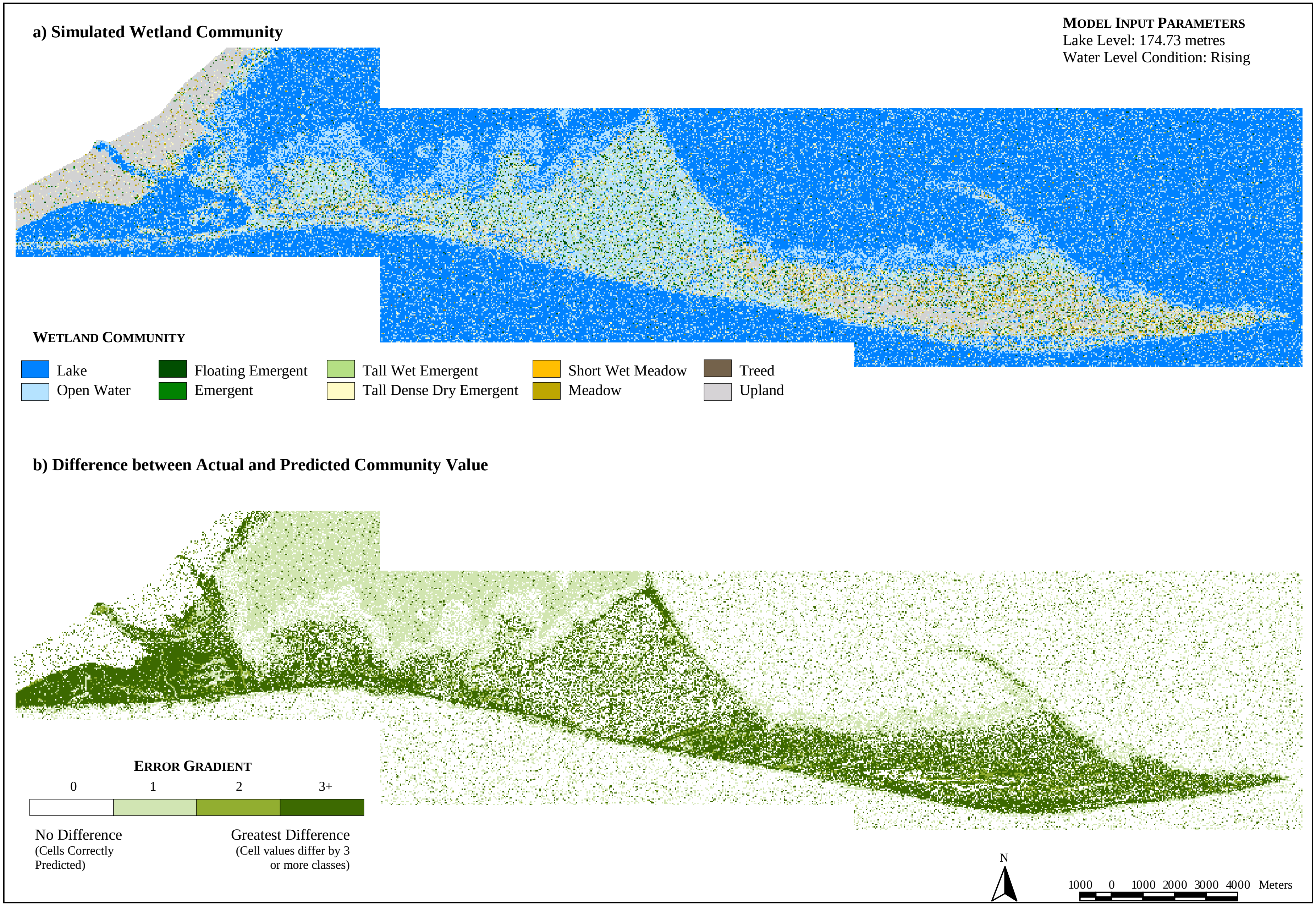


FIGURE I-11: RESULTS OF THE PROBABILITY MODEL v2.1, 1985

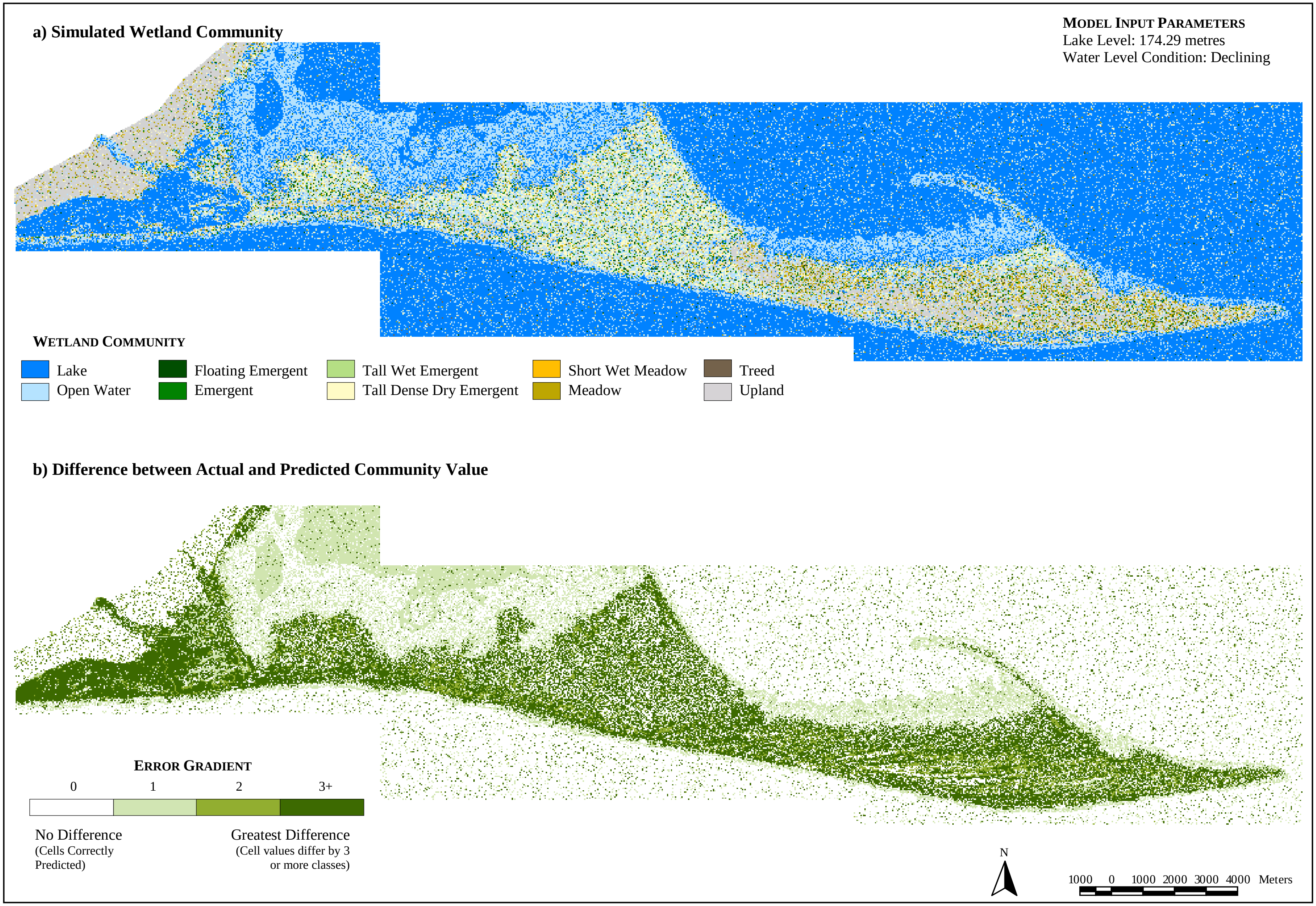


FIGURE I-12: RESULTS OF THE PROBABILITY MODEL V2.1, 1995

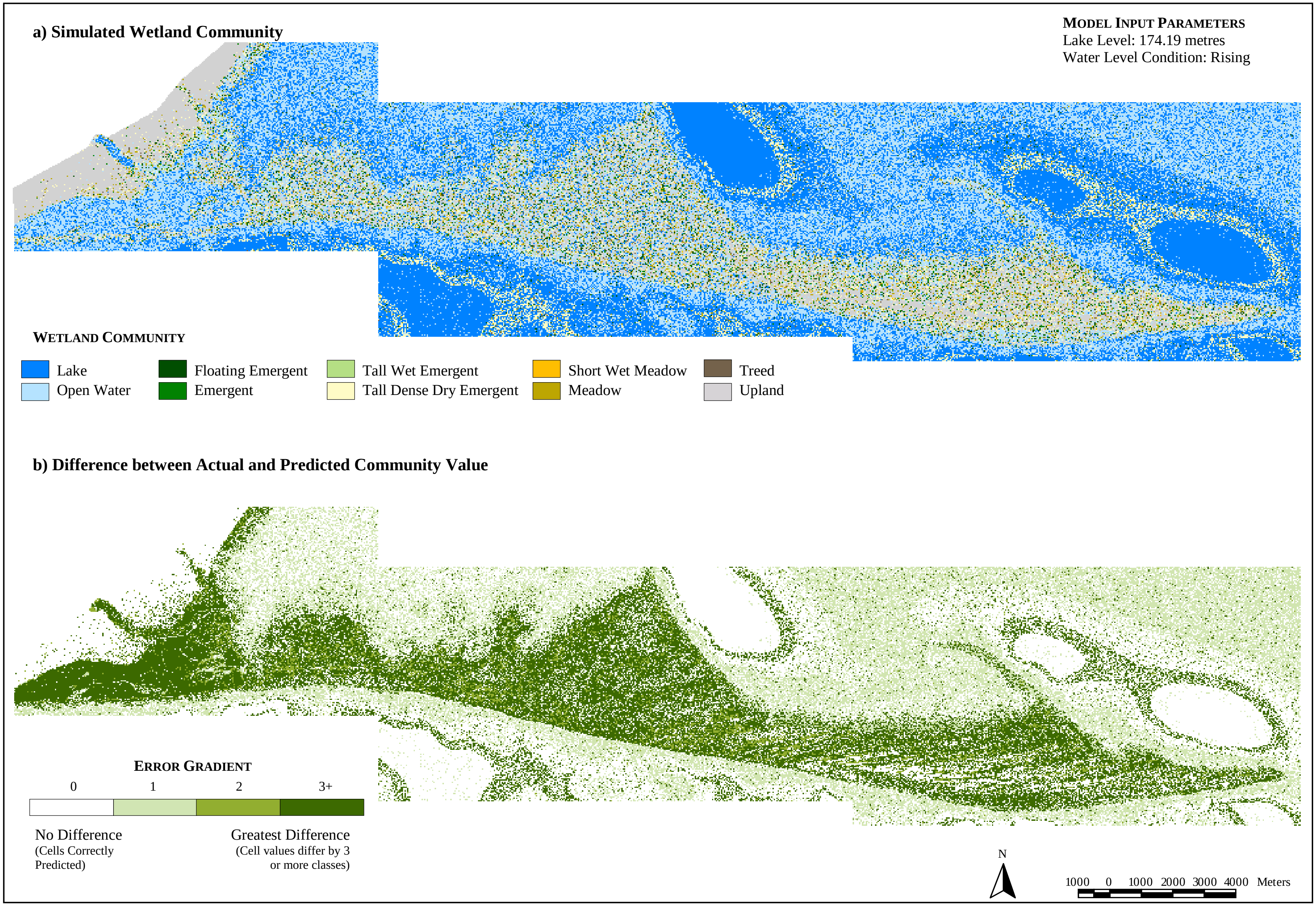


FIGURE I-13: RESULTS OF THE PROBABILITY MODEL v2.2, 1945

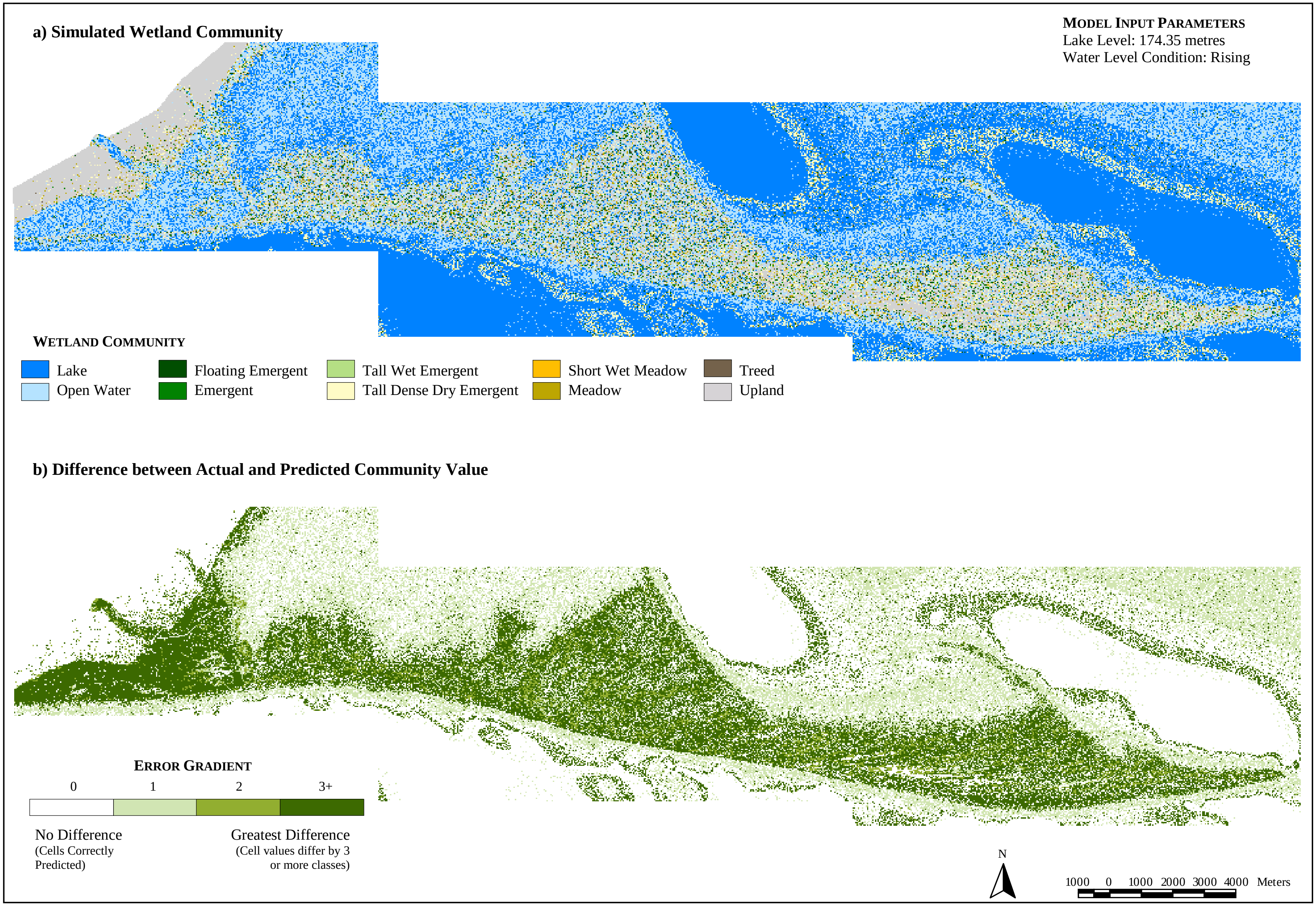


FIGURE I-14: RESULTS OF THE PROBABILITY MODEL V2.2, 1955

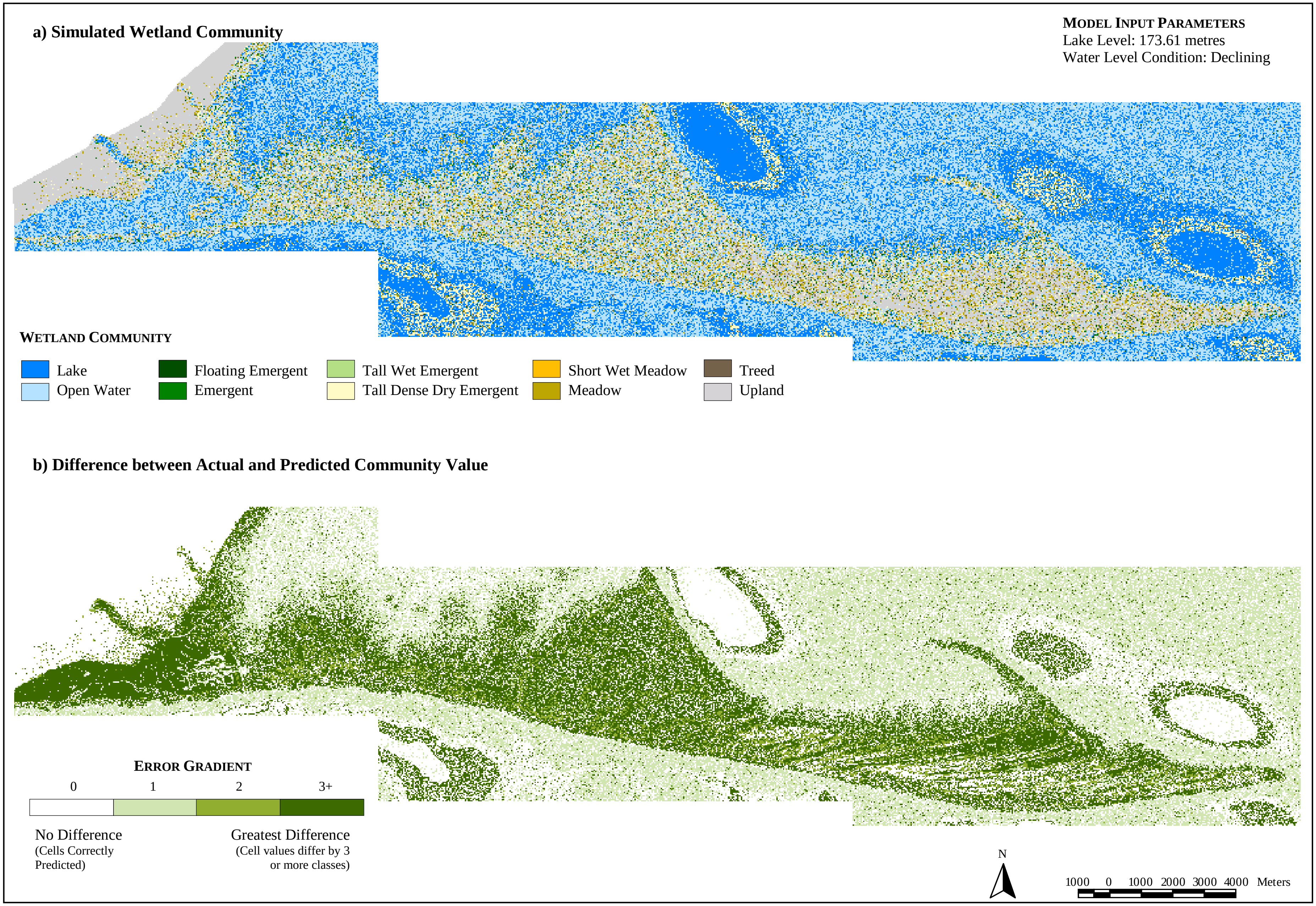


FIGURE I-15: RESULTS OF THE PROBABILITY MODEL v2.2, 1964

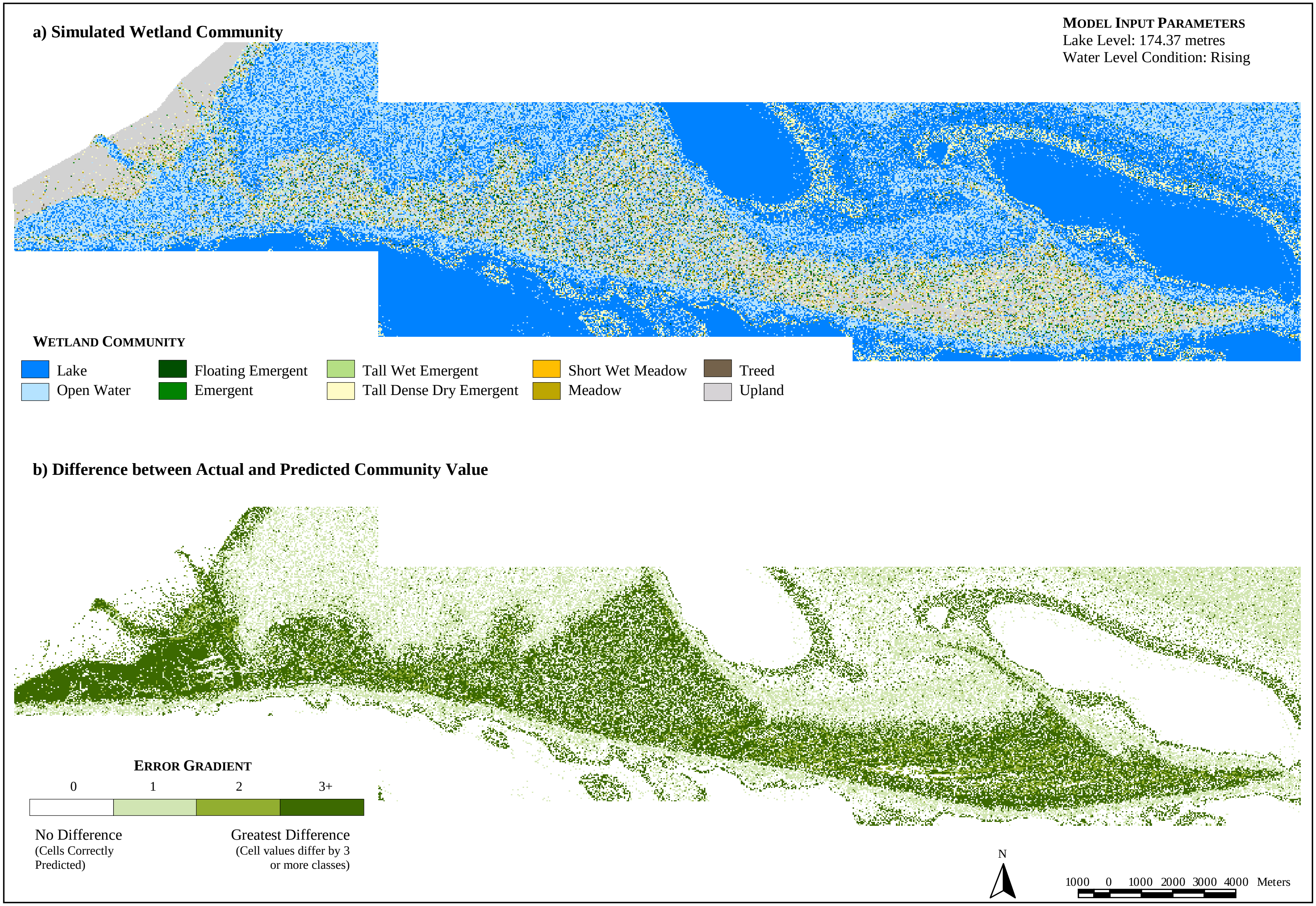


FIGURE I-16: RESULTS OF THE PROBABILITY MODEL V2.2, 1978

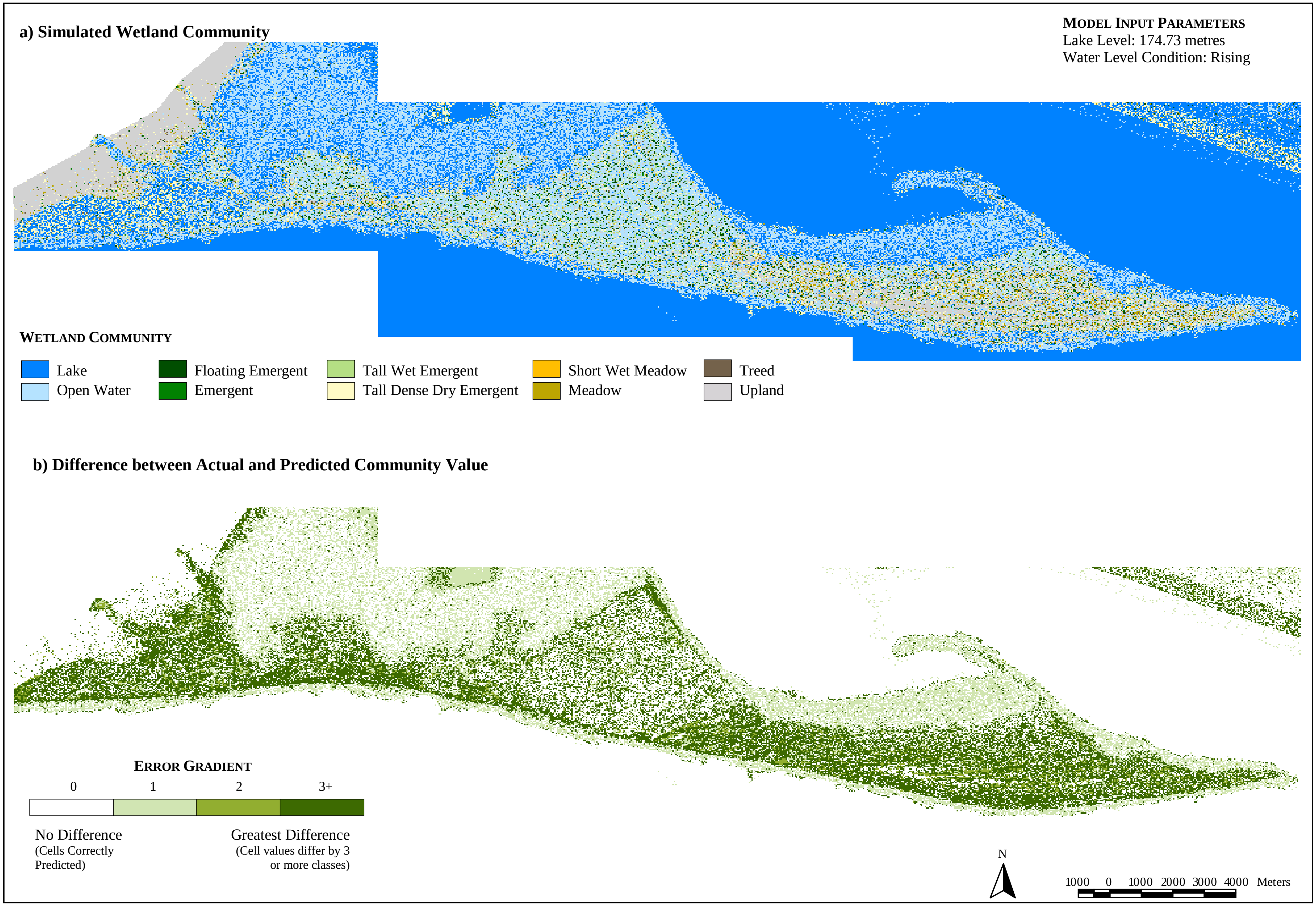


Figure I-17: Results of the Probability Model v2.2, 1985

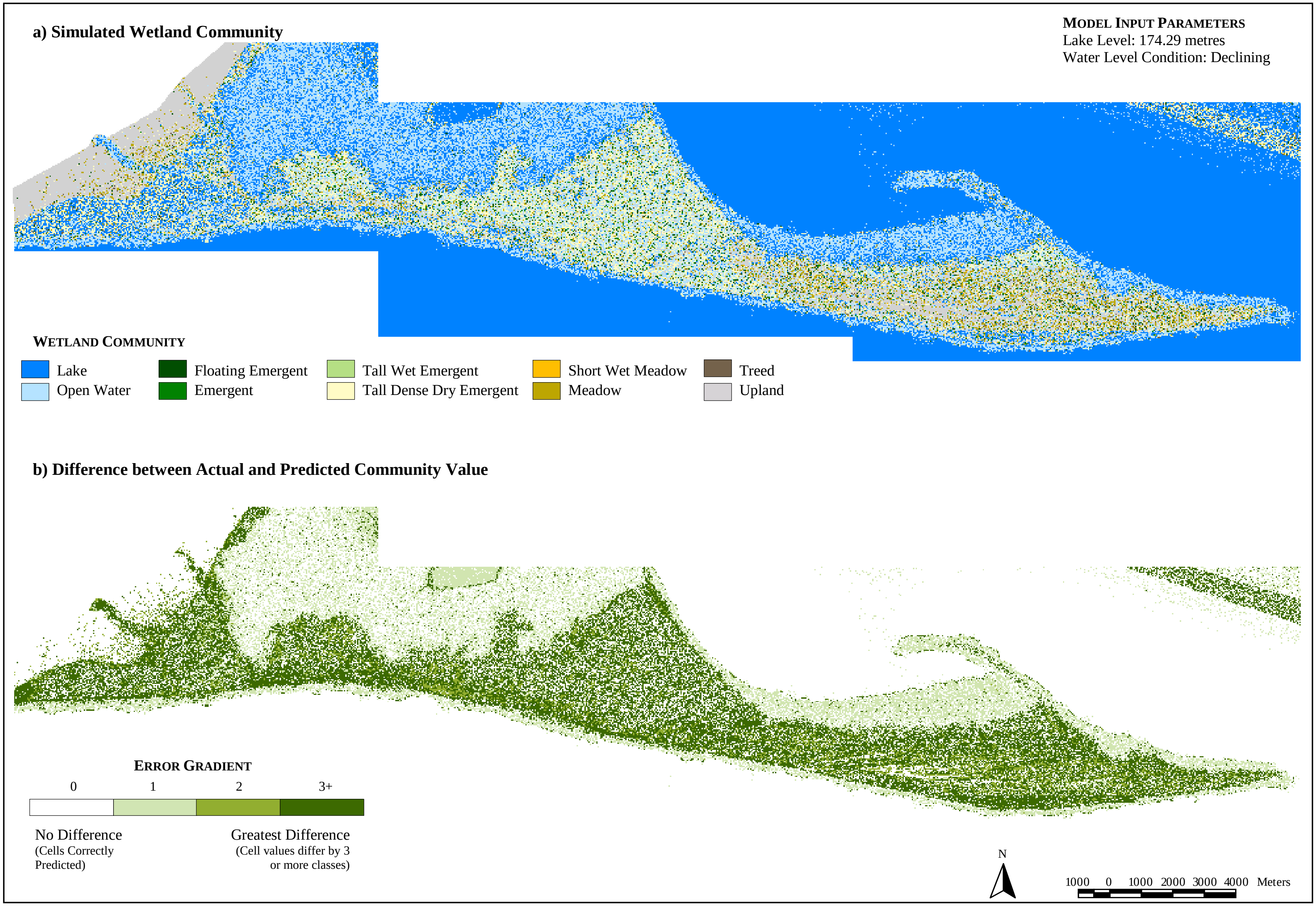


FIGURE I-18: RESULTS OF THE PROBABILITY MODEL V2.2, 1995

APPENDIX J
Vegetation Transition Modelling Results, 1945-1995

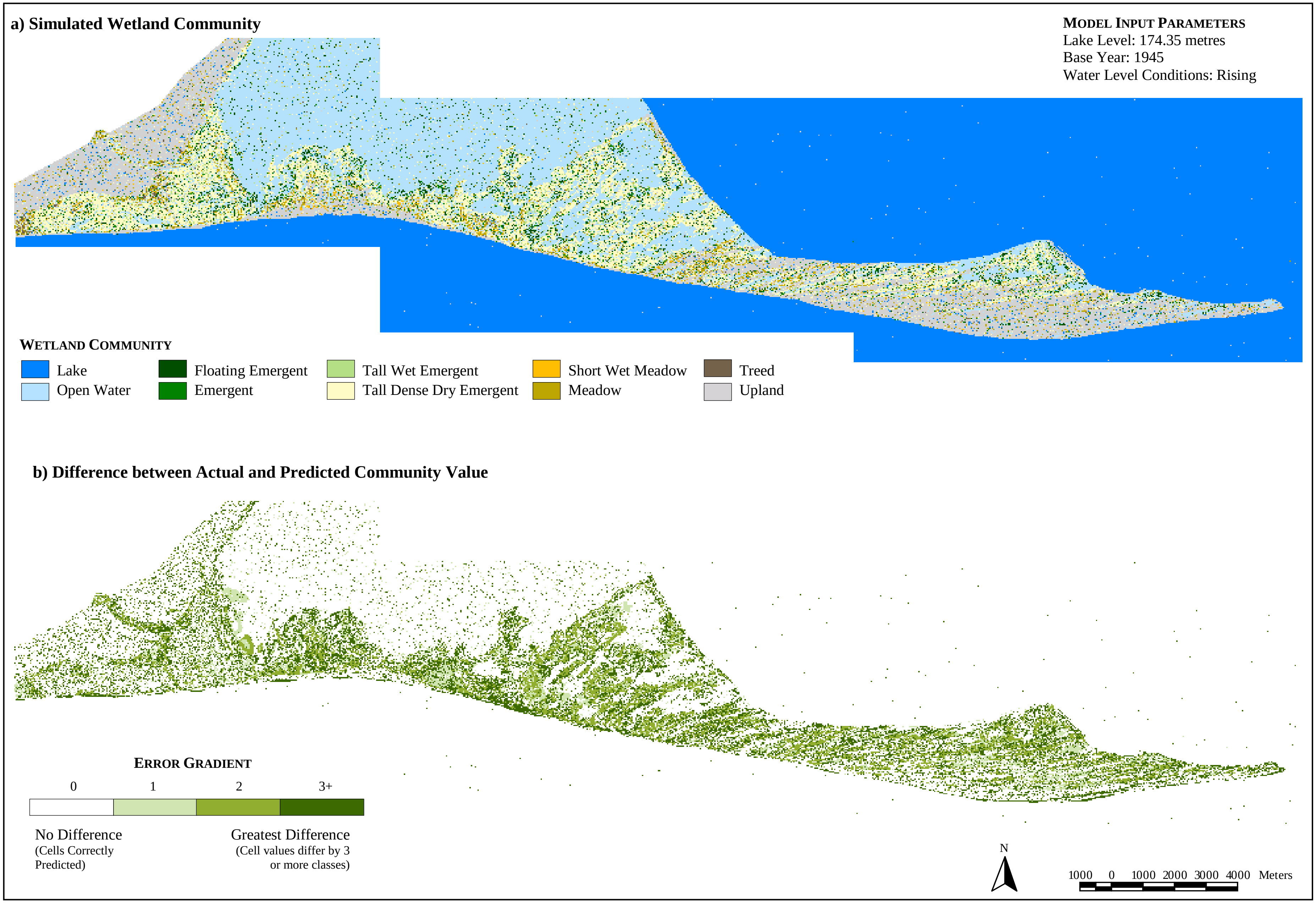


FIGURE J-1: RESULTS OF THE VEGETATION TRANSITION MODEL v3.0, 1955

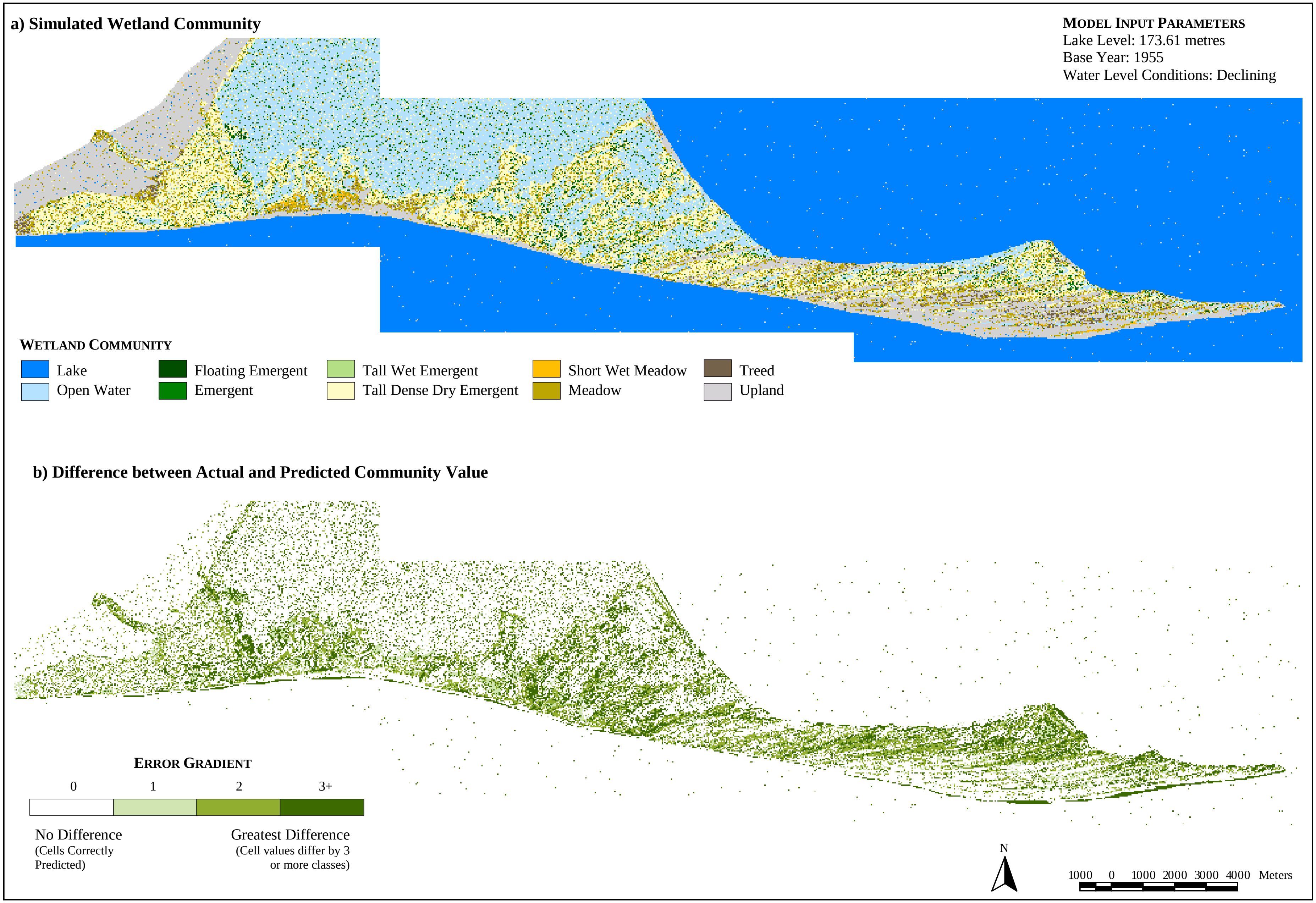


FIGURE J-2: RESULTS OF THE VEGETATION TRANSITION MODEL v3.0, 1964

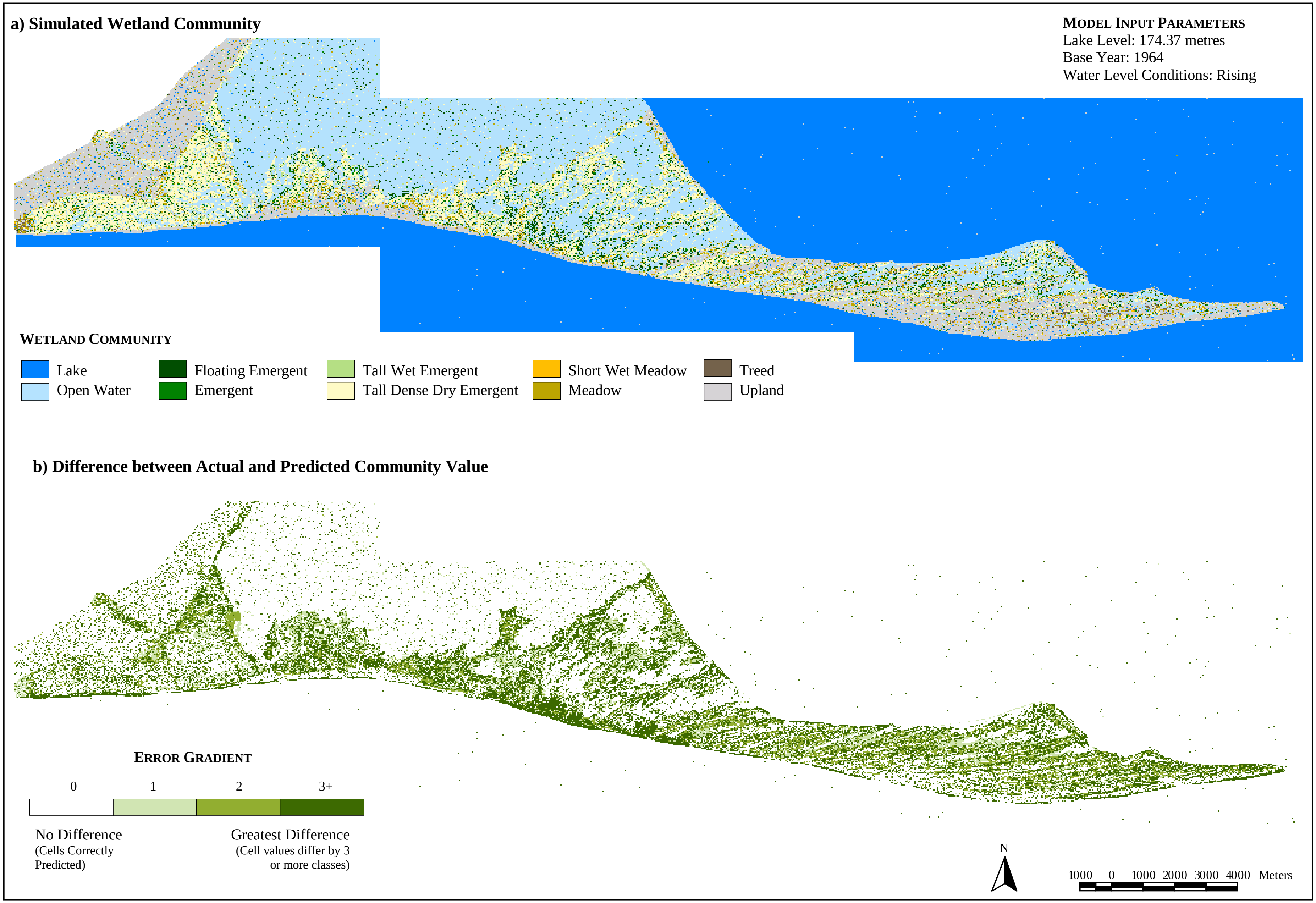


Figure J-3: Results of the Vegetation Transition Model v3.0, 1978

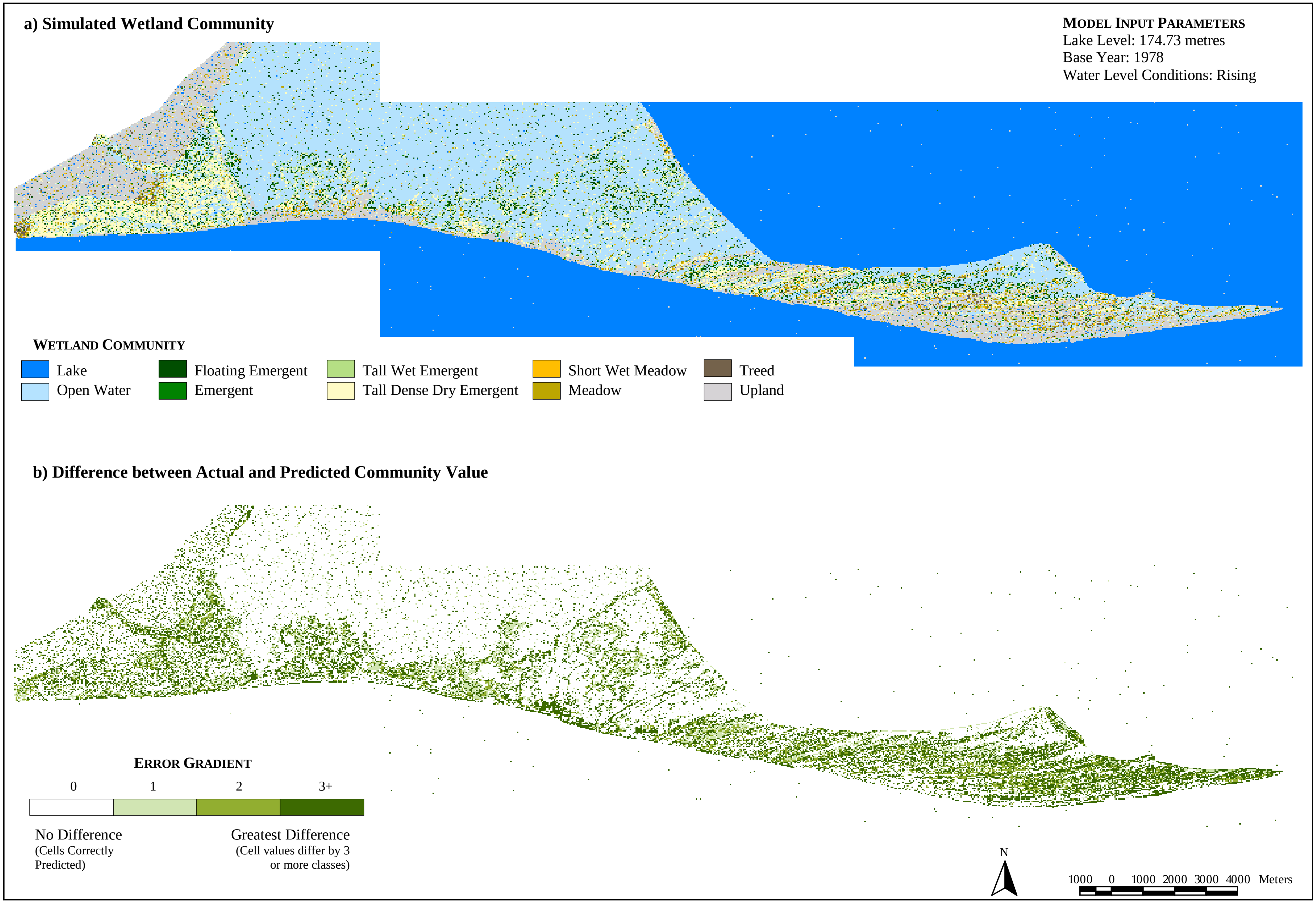


Figure J-4: Results of the Vegetation Transition Model v3.0, 1985

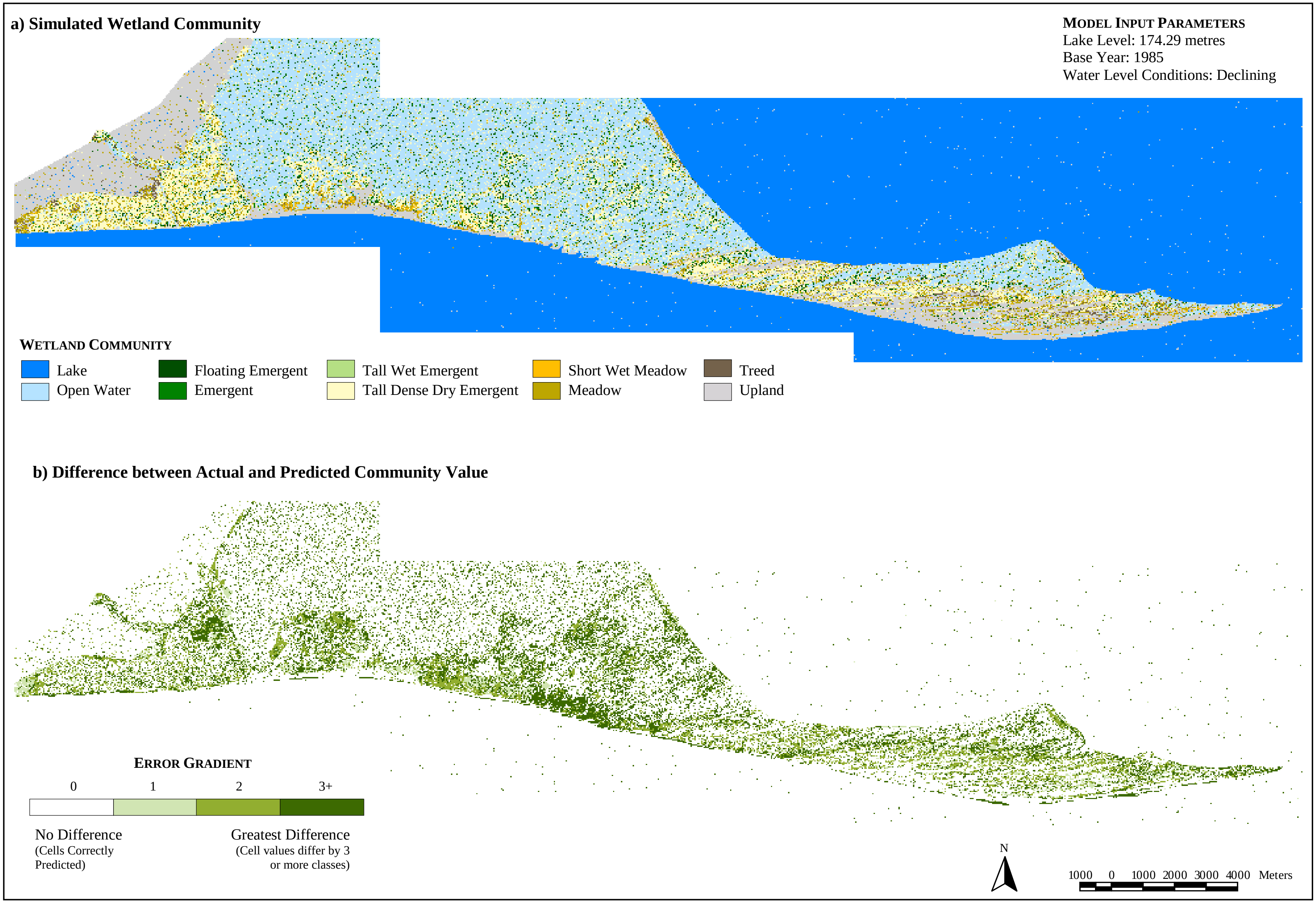


FIGURE J-5: RESULTS OF THE VEGETATION TRANSITION MODEL v3.0, 1995

APPENDIX K
Model Evaluation Tables

TABLE K-1: RULE-BASED MODEL V1.0 ACCURACY ASSESSMENT

Wetland Community	Number of Cells			Area (ha)			Percent (%)	
	Actual	Predicted	Correctly Predicted	Actual	Predicted	Difference in Area (P-A)	Correctly Predicted Cells	Difference in Area
1955 - Rising								
L	1183846	1128375	1030042	17047.382	16248.600	-798.782	87.01	-4.69
OW	391535	342360	182115	5638.104	4929.984	-708.120	46.51	-12.56
E1	45674	124109	13818	657.706	1787.170	1129.464	30.25	171.73
E	62552	154655	26243	900.749	2227.032	1326.283	41.95	147.24
EW	13338	49158	4856	192.067	707.875	515.808	36.41	268.56
E2	181855	57523	26775	2618.712	828.331	-1790.381	14.72	-68.37
M1	6176	13938	740	88.934	200.707	111.773	11.98	125.68
M	57883	40416	6977	833.515	581.990	-251.525	12.05	-30.18
T	15234	56785	2425	219.370	817.704	598.334	15.92	272.75
U	186097	176871	135532	2679.797	2546.942	-132.854	72.83	-4.96
Total	*2144190*	*2144190*	*1429523*	*30876.336*	*30876.336*		*66.67*	
1964 - Declining								
L	1173489	891632	884262	16898.242	12839.501	-4058.741	75.35	-24.02
OW	415199	520440	222346	5978.866	7494.336	1515.470	53.55	25.35
E1	15692	173507	6171	225.965	2498.501	2272.536	39.33	1005.70
E	48299	94425	7120	695.506	1359.720	664.214	14.74	95.50
EW	10311	22207	1878	148.478	319.781	171.302	18.21	115.37
E2	171004	140956	75191	2462.458	2029.766	-432.691	43.97	-17.57
M1	1197	9424	356	17.237	135.706	118.469	29.74	687.30
M	73014	63523	18861	1051.402	914.731	-136.670	25.83	-13.00
T	8734	41944	1485	125.770	603.994	478.224	17.00	380.24
U	226137	185018	170593	3256.373	2664.259	-592.114	75.44	-18.18
Total	*2143076*	*2143076*	*1388263*	*30860.294*	*30860.294*		*64.78*	
1978 – Rising								
L	1193108	1140852	1038171	17180.755	16428.269	-752.486	87.01	-4.38
OW	485135	333239	185702	6985.944	4798.642	-2187.302	38.28	-31.31
E1	41662	166525	13172	599.933	2397.960	1798.027	31.62	299.70
E	28078	160246	9019	404.323	2307.542	1903.219	32.12	470.72
EW	32130	11087	1266	462.672	159.653	-303.019	3.94	-65.49
E2	99243	56449	11281	1429.099	812.866	-616.234	11.37	-43.12
M1	3279	8068	239	47.218	116.179	68.962	7.29	146.05
M	58320	42578	11417	839.808	613.123	-226.685	19.58	-26.99
T	12741	63455	2296	183.470	913.752	730.282	18.02	398.04
U	190494	161691	127377	2743.114	2328.350	-414.763	66.87	-15.12
Total	*2144190*	*2144190*	*1399940*	*30876.336*	*30876.336*		*65.29*	
1985 - Rising								
L	1201068	1249252	1070238	17295.379	17989.229	693.850	89.11	4.01
OW	525670	429004	268848	7569.648	6177.658	-1391.990	51.14	-18.39
E1	42646	54693	5172	614.102	787.579	173.477	12.13	28.25
E	16015	133808	4039	230.616	1926.835	1696.219	25.22	735.52
EW	23343	10357	627	336.139	149.141	-186.998	2.69	-55.63
E2	108803	32557	8316	1566.763	468.821	-1097.942	7.64	-70.08
M1	6337	22154	829	91.253	319.018	227.765	13.08	249.60
M	31758	24280	5078	457.315	349.632	-107.683	15.99	-23.55
T	9483	56803	1124	136.555	817.963	681.408	11.85	499.00
U	179069	131284	112825	2578.594	1890.490	-688.104	63.01	-26.69
Total	*2144192*	*2144192*	*1477096*	*30876.365*	*30876.365*		*68.89*	

Wetland Community	Number of Cells: Actual	Predicted	Correctly Predicted	Area (ha): Actual	Predicted	Difference in Area (P-A)	Percent (%): Correctly Predicted Cells	Difference in Area
1995 - Declining								
L	1199370	1113064	1022788	17270.928	16028.122	-1242.806	85.28	-7.20
OW	441929	395800	204345	6363.778	5699.520	-664.258	46.24	-10.44
E1	31632	239617	15051	455.501	3450.485	2994.984	47.58	657.51
E	22142	99702	4771	318.845	1435.709	1116.864	21.55	350.28
EW	4404	44933	436	63.418	647.035	583.618	9.90	920.28
E2	173032	18279	4270	2491.661	263.218	-2228.443	2.47	-89.44
M1	7608	6920	184	109.555	99.648	-9.907	2.42	-9.04
M	61678	47147	18773	888.163	678.917	-209.246	30.44	-23.56
T	11650	41054	839	167.760	591.178	423.418	7.20	252.39
U	190747	137676	130516	2746.757	1982.534	-764.222	68.42	-27.82
Total	*2144192*	*2144192*	*1401973*	*30876.365*	*30876.365*		*65.38*	
1999 Declining								
L	1195434	1067860	1006752	17214.250	15377.184	-1837.066	84.22	-10.67
OW	433773	419225	220830	6246.331	6036.840	-209.491	50.91	-3.35
E1	35209	173611	5380	507.010	2499.998	1992.989	15.28	393.09
E	22176	88993	8725	319.334	1281.499	962.165	39.34	301.30
EW	4015	97869	1768	57.816	1409.314	1351.498	44.03	2337.58
E2	181270	19284	3238	2610.288	277.690	-2332.598	1.79	-89.36
M1	5423	21102	1359	78.091	303.869	225.778	25.06	289.12
M	62051	39134	8530	893.534	563.530	-330.005	13.75	-36.93
T	10463	68531	1585	150.667	986.846	836.179	15.15	554.98
U	194249	148454	127997	2797.186	2137.738	-659.448	65.89	-23.58
Total	*2144063*	*2144063*	*1386164*	*30874.507*	*30874.507*		*64.65*	

Table K-2: Overall Accuracy of v1.0 *

Wetland Community	Average	Declining	Rising
Overall	*65.944*	*64.938*	*66.949*
L	84.663	81.616	87.710
OW	47.773	50.233	45.312
E1	29.364	34.063	24.666
E	29.155	25.211	33.098
EW	19.197	24.050	14.345
E2	13.660	16.075	11.244
M1	14.929	19.073	10.784
M	19.606	23.339	15.873
T	14.191	13.118	15.264
U	68.743	69.918	67.567

* Based on percentage of correctly predicted cells

TABLE K-3: RULE-BASED MODEL V1.1 ACCURACY ASSESSMENT

WETLAND COMMUNITY	NUMBER OF CELLS			AREA (HA)			PERCENT (%)	
	ACTUAL	PREDICTED	CORRECTLY PREDICTED	ACTUAL	PREDICTED	DIFFERENCE IN AREA (P-A)	CORRECTLY PREDICTED CELLS	DIFFERENCE IN AREA
1955 - RISING								
L	1183846	1198976	1174274	17047.382	17265.254	217.872	99.19	1.28
OW	390726	381860	346490	5626.454	5498.784	-127.670	88.68	-2.27
E1	45674	34948	7368	657.706	503.251	-154.454	16.13	-23.48
E	62552	73036	4706	900.749	1051.718	150.970	7.52	16.76
EW	13338	78517	4796	192.067	1130.645	938.578	35.96	488.67
E2	181855	81611	34807	2618.712	1175.198	-1443.514	19.14	-55.12
M1	6176	24117	869	88.934	347.285	258.350	14.07	290.50
M	57883	54415	9864	833.515	783.576	-49.939	17.04	-5.99
T	15234	49978	2797	219.370	719.683	500.314	18.36	228.07
U	186097	165923	129260	2679.797	2389.291	-290.506	69.46	-10.84
Total	*2143381*	*2143381*	*1715231*	*30864.686*	*30864.686*		*80.02*	
1964 - DECLINING								
L	1174604	1135259	1129971	16914.298	16347.730	-566.568	96.20	-3.35
OW	414390	342252	285898	5967.216	4928.429	-1038.787	68.99	-17.41
E1	15692	113163	5295	225.965	1629.547	1403.582	33.74	621.15
E	48299	128793	18140	695.506	1854.619	1159.114	37.56	166.66
EW	10311	1810	227	148.478	26.064	-122.414	2.20	-82.45
E2	171004	122135	73643	2462.458	1758.744	-703.714	43.07	-28.58
M1	1197	409	0	17.237	5.890	-11.347	0.00	-65.83
M	73014	73578	23846	1051.402	1059.523	8.122	32.66	0.77
T	8734	55132	1401	125.770	793.901	668.131	16.04	531.23
U	226137	170851	157443	3256.373	2460.254	-796.118	69.62	-24.45
Total	*2143382*	*2143382*	*1695864*	*30864.701*	*30864.701*		*79.12*	
1978 – RISING								
L	1193108	1201590	1181400	17180.755	17302.896	122.141	99.02	0.71
OW	484326	408747	379969	6974.294	5885.957	-1088.338	78.45	-15.60
E1	41662	51059	6221	599.933	735.250	135.317	14.93	22.56
E	28078	68717	4849	404.323	989.525	585.202	17.27	144.74
EW	32130	92860	20923	462.672	1337.184	874.512	65.12	189.01
E2	99243	39785	8129	1429.099	572.904	-856.195	8.19	-59.91
M1	3279	25507	497	47.218	367.301	320.083	15.16	677.89
M	58320	50841	13983	839.808	732.110	-107.698	23.98	-12.82
T	12741	53767	1277	183.470	774.245	590.774	10.02	322.00
U	190494	150508	120635	2743.114	2167.315	-575.798	63.33	-20.99
Total	*2143381*	*2143381*	*1737883*	*30864.686*	*30864.686*		*81.08*	
1985 - RISING								
L	1201068	1238160	1193327	17295.379	17829.504	534.125	99.36	3.09
OW	524861	505895	457692	7557.998	7284.888	-273.110	87.20	-3.61
E1	42646	36378	5702	614.102	523.843	-90.259	13.37	-14.70
E	16015	124169	3795	230.616	1788.034	1557.418	23.70	675.33
EW	23343	3414	516	336.139	49.162	-286.978	2.21	-85.37
E2	108803	38080	13289	1566.763	548.352	-1018.411	12.21	-65.00
M1	6337	2644	72	91.253	38.074	-53.179	1.14	-58.28
M	31758	46548	10052	457.315	670.291	212.976	31.65	46.57
T	9483	47534	890	136.555	684.490	547.934	9.39	401.25
U	179069	100561	94172	2578.594	1448.078	-1130.515	52.59	-43.84
Total	*2143383*	*2143383*	*1779507*	*30864.715*	*30864.715*		*83.02*	

Wetland Community	Number of Cells			Area (ha)			Percent (%)	
	Actual	Predicted	Correctly Predicted	Actual	Predicted	Difference in Area (P-A)	Correctly Predicted Cells	Difference in Area
1995 - Declining								
L	1199370	1205211	1185657	17270.928	17355.038	84.110	98.86	0.49
OW	441120	502933	403032	6352.128	7242.235	890.107	91.37	14.01
E1	31632	71153	7414	455.501	1024.603	569.102	23.44	124.94
E	22142	66386	2237	318.845	955.958	637.114	10.10	199.82
EW	4404	11829	434	63.418	170.338	106.920	9.85	168.60
E2	173032	60071	38359	2491.661	865.022	-1626.638	22.17	-65.28
M1	7608	2585	753	109.555	37.224	-72.331	9.90	-66.02
M	61678	54411	22925	888.163	783.518	-104.645	37.17	-11.78
T	11650	38797	702	167.760	558.677	390.917	6.03	233.02
U	190747	130007	123722	2746.757	1872.101	-874.656	64.86	-31.84
Total	*2143383*	*2143383*	*1785235*	*30864.715*	*30864.715*		*83.29*	
1999 Declining								
L	1195434	1188209	1173779	17214.250	17110.210	-104.040	98.19	-0.60
OW	432968	435961	386431	6234.739	6277.838	43.099	89.25	0.69
E1	35209	54991	10759	507.010	791.870	284.861	30.56	56.18
E	22176	66719	8384	319.334	960.754	641.419	37.81	200.86
EW	4015	6867	264	57.816	98.885	41.069	6.58	71.03
E2	181270	111462	84713	2610.288	1605.053	-1005.235	46.73	-38.51
M1	5423	5307	1462	78.091	76.421	-1.670	26.96	-2.14
M	62051	62988	21678	893.534	907.027	13.493	34.94	1.51
T	10463	69806	839	150.667	1005.206	854.539	8.02	567.17
U	194249	140948	121819	2797.186	2029.651	-767.534	62.71	-27.44
Total	*2143258*	*2143258*	*1810128*	*30862.915*	*30862.915*		*84.46*	

Table K-4: Overall Accuracy of v1.1 *

Wetland Community	Average	Declining	Rising
Overall	*70.065*	*82.289*	*81.376*
L	98.468	97.748	99.189
OW	83.991	83.203	84.778
E1	22.029	29.246	14.811
E	22.326	28.489	16.163
EW	20.320	6.211	34.429
E2	25.252	37.322	13.182
M1	11.203	12.286	10.121
M	29.572	34.921	24.223
T	11.309	10.028	12.589
U	63.762	65.732	61.792

* Based on percentage of correctly predicted cells

Table K-5: Vegetation Transition Model v2.0 Accuracy Assessment

Wetland Community	Number of Cells: Actual	Number of Cells: Predicted	Number of Cells: Correctly Predicted	Area (ha): Actual	Area (ha): Predicted	Area (ha): Difference in Area (P-A)	Percent (%): Correctly Predicted Cells	Percent (%): Difference in Area
1945 - Rising								
L	1178839	1052334	868088	16975.282	15153.610	-1821.672	73.64	-10.73
OW	376361	488526	146666	5419.598	7034.774	1615.176	38.97	29.80
E1	22817	50640	1026	328.565	729.216	400.651	4.50	121.94
E	37004	23968	1063	532.858	345.139	-187.718	2.87	-35.23
EW	7005	22330	70	100.872	321.552	220.680	1.00	218.77
E2	203411	126346	18728	2929.118	1819.382	-1109.736	9.21	-37.89
M1	4775	5379	45	68.760	77.458	8.698	0.94	12.65
M	75450	48796	4186	1086.480	702.662	-383.818	5.55	-35.33
T	3135	13886	27	45.144	199.958	154.814	0.86	342.93
U	235400	311992	136470	3389.760	4492.685	1102.925	57.97	32.54
Total	*2144197*	*2144197*	*1176369*	*30876.437*	*30876.437*		*54.86*	
1955 - Rising								
L	1183846	1090378	877712	17047.382	15701.443	-1345.939	74.14	-7.90
OW	391535	494278	142092	5638.104	7117.603	1479.499	36.29	26.24
E1	45674	47661	2261	657.706	686.318	28.613	4.95	4.35
E	62552	22882	1737	900.749	329.501	-571.248	2.78	-63.42
EW	13338	24305	274	192.067	349.992	157.925	2.05	82.22
E2	181857	126277	16104	2618.741	1818.389	-800.352	8.86	-30.56
M1	6176	5199	53	88.934	74.866	-14.069	0.86	-15.82
M	57884	43899	3307	833.530	632.146	-201.384	5.71	-24.16
T	15234	12732	236	219.370	183.341	-36.029	1.55	-16.42
U	186097	276582	105315	2679.797	3982.781	1302.984	56.59	48.62
Total	*2144193*	*2144193*	*1149091*	*30876.379*	*30876.379*		*53.59*	
1964 - Declining								
L	1174604	1020875	872185	16914.298	14700.600	-2213.698	74.25	-13.09
OW	415199	464454	161956	5978.866	6688.138	709.272	39.01	11.86
E1	15692	28693	482	225.965	413.179	187.214	3.07	82.85
E	48299	18625	686	695.506	268.200	-427.306	1.42	-61.44
EW	10311	3945	47	148.478	56.808	-91.670	0.46	-61.74
E2	171006	145362	20444	2462.486	2093.213	-369.274	11.96	-15.00
M1	1197	10232	14	17.237	147.341	130.104	1.17	754.80
M	73014	80857	6974	1051.402	1164.341	112.939	9.55	10.74
T	8734	16779	87	125.770	241.618	115.848	1.00	92.11
U	226140	354374	143360	3256.416	5102.986	1846.570	63.39	56.71
Total	*2144196*	*2144196*	*1206235*	*30876.422*	*30876.422*		*56.26*	
1978 - Rising								
L	1193108	1096449	880488	17180.755	15788.866	-1391.890	73.80	-8.10
OW	485135	495272	170064	6985.944	7131.917	145.973	35.05	2.09
E1	41662	46649	1973	599.933	671.746	71.813	4.74	11.97
E	28078	22512	783	404.323	324.173	-80.150	2.79	-19.82
EW	32130	25523	851	462.672	367.531	-95.141	2.65	-20.56
E2	99243	126784	7599	1429.099	1825.690	396.590	7.66	27.75
M1	3279	5118	29	47.218	73.699	26.482	0.88	56.08
M	58320	43045	3255	839.808	619.848	-219.960	5.58	-26.19
T	12741	12322	163	183.470	177.437	-6.034	1.28	-3.29
U	190498	270520	107280	2743.171	3895.488	1152.317	56.32	42.01
Total	*2144194*	*2144194*	*1172485*	*30876.394*	*30876.394*		*54.68*	

Wetland Community	Number of Cells			Area (ha)			Percent (%)	
	Actual	Predicted	Correctly Predicted	Actual	Predicted	Difference in Area (P-A)	Correctly Predicted Cells	Difference in Area
1985 - Rising								
L	1201068	1201402	906846	17295.379	17300.189	4.810	75.50	0.03
OW	525670	525570	194160	7569.648	7568.208	-1.440	36.94	-0.02
E1	42648	40727	1840	614.131	586.469	-27.662	4.31	-4.50
E	16015	15886	304	230.616	228.758	-1.858	1.90	-0.81
EW	23343	23564	621	336.139	339.322	3.182	2.66	0.95
E2	108803	108868	7461	1566.763	1567.699	0.936	6.86	0.06
M1	6337	5578	82	91.253	80.323	-10.930	1.29	-11.98
M	31758	31732	1855	457.315	456.941	-0.374	5.84	-0.08
T	9483	10282	131	136.555	148.061	11.506	1.38	8.43
U	179071	180587	92613	2578.622	2600.453	21.830	51.72	0.85
Total	*2144196*	*2144196*	*1205913*	*30876.422*	*30876.422*		*56.24*	
1995 - Declining								
L	1199417	1199220	963102	17271.605	17268.768	-2.837	80.30	-0.02
OW	441929	441875	164272	6363.778	6363.000	-0.778	37.17	-0.01
E1	31632	31331	1284	455.501	451.166	-4.334	4.06	-0.95
E	22143	22186	734	318.859	319.478	0.619	3.31	0.19
EW	4404	4460	27	63.418	64.224	0.806	0.61	1.27
E2	173032	173102	35113	2491.661	2492.669	1.008	20.29	0.04
M1	7608	7450	98	109.555	107.280	-2.275	1.29	-2.08
M	61678	61645	6448	888.163	887.688	-0.475	10.45	-0.05
T	11650	11692	139	167.760	168.365	0.605	1.19	0.36
U	190747	191279	95124	2746.757	2754.418	7.661	49.87	0.28
Total	*2144240*	*2144240*	*1266341*	*30877.056*	*30877.056*		*59.06*	
1999 - Declining								
L	1195434	1157535	950533	17214.250	16668.504	-545.746	79.51	-3.17
OW	433775	458341	172885	6246.360	6600.110	353.750	39.86	5.66
E1	35209	31824	1264	507.010	458.266	-48.744	3.59	-9.61
E	22176	23073	693	319.334	332.251	12.917	3.13	4.04
EW	4015	5188	33	57.816	74.707	16.891	0.82	29.22
E2	181271	185682	39777	2610.302	2673.821	63.518	21.94	2.43
M1	5423	6786	65	78.091	97.718	19.627	1.20	25.13
M	62051	60130	5773	893.534	865.872	-27.662	9.30	-3.10
T	10463	10321	74	150.667	148.622	-2.045	0.71	-1.36
U	194252	205189	97970	2797.229	2954.722	157.493	50.43	5.63
Total	*2144069*	*2144069*	*1269067*	*30874.594*	*30874.594*		*59.19*	

Table K-6: Overall Accuracy of v2.0 *

Wetland Community	Average	Declining	Rising
Overall	*55.716*	*57.723*	*54.379*
L	75.069	76.884	73.859
OW	37.836	39.431	36.772
E1	4.169	3.331	4.728
E	2.597	2.273	2.813
EW	1.396	0.639	1.901
E2	11.924	16.949	8.573
M1	1.011	1.184	0.895
M	7.140	9.428	5.614
T	1.079	0.852	1.230
U	56.942	56.914	56.960

* Based on percentage of correctly predicted cells

* Results for 1985 and 1995 were excluded from the averages

Table K-7: Vegetation Transition Model v2.1 Accuracy Assessment

Wetland Community	Number of Cells: Actual	Number of Cells: Predicted	Number of Cells: Correctly Predicted	Area (ha): Actual	Area (ha): Predicted	Area (ha): Difference in Area (P-A)	Percent (%): Correctly Predicted Cells	Percent (%): Difference in Area
1945 - Rising								
L	1178839	1108074	887641	16975.282	15956.266	-1019.016	75.30	-6.00
OW	376361	464144	134627	5419.598	6683.674	1264.075	35.77	23.32
E1	22817	41100	1001	328.565	591.840	263.275	4.39	80.13
E	37004	22488	1151	532.858	323.827	-209.030	3.11	-39.23
EW	7005	13667	95	100.872	196.805	95.933	1.36	95.10
E2	203411	117383	18583	2929.118	1690.315	-1238.803	9.14	-42.29
M1	4775	6539	27	68.760	94.162	25.402	0.57	36.94
M	75450	48263	4262	1086.480	694.987	-391.493	5.65	-36.03
T	3135	12805	16	45.144	184.392	139.248	0.51	308.45
U	235400	309734	136449	3389.760	4460.170	1070.410	57.96	31.58
Total	*2144197*	*2144197*	*1183852*	*30876.437*	*30876.437*		*55.21*	
1955 - Rising								
L	1183846	1143383	898444	17047.382	16464.715	-582.667	75.89	-3.42
OW	391535	468687	133388	5638.104	6749.093	1110.989	34.07	19.71
E1	45674	42767	2472	657.706	615.845	-41.861	5.41	-6.36
E	62552	20904	1709	900.749	301.018	-599.731	2.73	-66.58
EW	13338	13758	200	192.067	198.115	6.048	1.50	3.15
E2	181857	118127	16591	2618.741	1701.029	-917.712	9.12	-35.04
M1	6176	6539	46	88.934	94.162	5.227	0.74	5.88
M	57884	41531	2937	833.530	598.046	-235.483	5.07	-28.25
T	15234	11979	197	219.370	172.498	-46.872	1.29	-21.37
U	186097	276518	107109	2679.797	3981.859	1302.062	57.56	48.59
Total	*2144193*	*2144193*	*1163093*	*30876.379*	*30876.379*		*54.24*	
1964 - Declining								
L	1174604	1027600	870152	16914.298	14797.440	-2116.858	74.08	-12.52
OW	415199	453878	146561	5978.866	6535.843	556.978	35.30	9.32
E1	15692	27635	513	225.965	397.944	171.979	3.27	76.11
E	48299	17990	735	695.506	259.056	-436.450	1.52	-62.75
EW	10311	3668	44	148.478	52.819	-95.659	0.43	-64.43
E2	171006	137805	19143	2462.486	1984.392	-478.094	11.19	-19.42
M1	1197	10931	11	17.237	157.406	140.170	0.92	813.20
M	73014	84345	6868	1051.402	1214.568	163.166	9.41	15.52
T	8734	17659	81	125.770	254.290	128.520	0.93	102.19
U	226140	362685	145676	3256.416	5222.664	1966.248	64.42	60.38
Total	*2144196*	*2144196*	*1189784*	*30876.422*	*30876.422*		*55.49*	
1978 - Rising								
L	1193108	1148498	901878	17180.755	16538.371	-642.384	75.59	-3.74
OW	485135	470457	160923	6985.944	6774.581	-211.363	33.17	-3.03
E1	41662	41942	2203	599.933	603.965	4.032	5.29	0.67
E	28078	20594	831	404.323	296.554	-107.770	2.96	-26.65
EW	32130	13972	508	462.672	201.197	-261.475	1.58	-56.51
E2	99243	118624	7551	1429.099	1708.186	279.086	7.61	19.53
M1	3279	6237	16	47.218	89.813	42.595	0.49	90.21
M	58320	42048	3139	839.808	605.491	-234.317	5.38	-27.90
T	12741	12313	146	183.470	177.307	-6.163	1.15	-3.36
U	190498	269509	108588	2743.171	3880.930	1137.758	57.00	41.48
Total	*2144194*	*2144194*	*1185783*	*30876.394*	*30876.394*		*55.30*	

Wetland Community	Number of Cells			Area (ha)			Percent (%)	
	Actual	Predicted	Correctly Predicted	Actual	Predicted	Difference in Area (P-A)	Correctly Predicted Cells	Difference in Area
1985 - Rising								
L	1201068	1201087	926692	17295.379	17295.653	0.274	77.16	0.00
OW	525670	525984	202345	7569.648	7574.170	4.522	38.49	0.06
E1	42648	42719	2623	614.131	615.154	1.022	6.15	0.17
E	16015	15852	365	230.616	228.269	-2.347	2.28	-1.02
EW	23343	23346	1241	336.139	336.182	0.043	5.32	0.01
E2	108803	109093	8870	1566.763	1570.939	4.176	8.15	0.27
M1	6337	6288	127	91.253	90.547	-0.706	2.00	-0.77
M	31758	31653	1889	457.315	455.803	-1.512	5.95	-0.33
T	9483	9484	151	136.555	136.570	0.014	1.59	0.01
U	179071	178690	94565	2578.622	2573.136	-5.486	52.81	-0.21
Total	*2144196*	*2144196*	*1238868*	*30876.422*	*30876.422*		*57.78*	
1995 - Declining								
L	1199417	1199422	965234	17271.605	17271.677	0.072	80.48	0.00
OW	441929	441736	172642	6363.778	6360.998	-2.779	39.07	-0.04
E1	31632	31497	1312	455.501	453.557	-1.944	4.15	-0.43
E	22143	21931	790	318.859	315.806	-3.053	3.57	-0.96
EW	4404	4434	30	63.418	63.850	0.432	0.68	0.68
E2	173032	173487	35808	2491.661	2498.213	6.552	20.69	0.26
M1	7608	7383	178	109.555	106.315	-3.240	2.34	-2.96
M	61678	61914	6607	888.163	891.562	3.398	10.71	0.38
T	11650	11530	167	167.760	166.032	-1.728	1.43	-1.03
U	190747	190906	97060	2746.757	2749.046	2.290	50.88	0.08
Total	*2144240*	*2144240*	*1279828*	*30877.056*	*30877.056*		*59.69*	
1999 - Declining								
L	1195434	1184904	954657	17214.250	17062.618	-151.632	79.86	-0.88
OW	433775	416649	163364	6246.360	5999.746	-246.614	37.66	-3.95
E1	35209	26621	1016	507.010	383.342	-123.667	2.89	-24.39
E	22176	19423	559	319.334	279.691	-39.643	2.52	-12.41
EW	4015	4801	33	57.816	69.134	11.318	0.82	19.58
E2	181271	140010	23864	2610.302	2016.144	-594.158	13.16	-22.76
M1	5423	9492	93	78.091	136.685	58.594	1.71	75.03
M	62051	70202	6297	893.534	1010.909	117.374	10.15	13.14
T	10463	13351	79	150.667	192.254	41.587	0.76	27.60
U	194252	258616	102973	2797.229	3724.070	926.842	53.01	33.13
Total	*2144069*	*2144069*	*1252935*	*30874.594*	*30874.594*		*58.44*	

Table K-8: Overall Accuracy of v2.1 *

Wetland Community	Average	Declining	Rising
Overall	*55.737*	*56.963*	*54.919*
L	76.144	76.970	75.594
OW	35.194	36.480	34.336
E1	4.248	3.077	5.029
E	2.569	2.021	2.934
EW	1.137	0.624	1.479
E2	10.045	12.180	8.622
M1	0.886	1.317	0.599
M	7.132	9.777	5.368
T	0.926	0.841	0.983
U	57.990	58.714	57.507

* Based on percentage of correctly predicted cells

* Results for 1985 and 1995 were excluded from the averages

Table K-9: Vegetation Transition Model v2.2 Accuracy Assessment

Wetland Community	Number of Cells: Actual	Number of Cells: Predicted	Number of Cells: Correctly Predicted	Area (ha): Actual	Area (ha): Predicted	Area (ha): Difference in Area (P-A)	Percent (%): Correctly Predicted Cells	Percent (%): Difference in Area
1945 - Rising								
L	1178839	790835	629527	16975.282	11388.024	-5587.258	53.40	-32.91
OW	376361	729180	166163	5419.598	10500.192	5080.594	44.15	93.74
E1	22817	48241	1088	328.565	694.670	366.106	4.77	111.43
E	37004	24031	1113	532.858	346.046	-186.811	3.01	-35.06
EW	7005	15348	91	100.872	221.011	120.139	1.30	119.10
E2	203411	155512	18193	2929.118	2239.373	-689.746	8.94	-23.55
M1	4775	6765	43	68.760	97.416	28.656	0.90	41.68
M	75450	51842	4402	1086.480	746.525	-339.955	5.83	-31.29
T	3135	14875	17	45.144	214.200	169.056	0.54	374.48
U	235400	307568	136469	3389.760	4428.979	1039.219	57.97	30.66
Total	*2144197*	*2144197*	*957106*	*30876.437*	*30876.437*		*44.64*	
1955 - Rising								
L	1183846	951523	780583	17047.382	13701.931	-3345.451	65.94	-19.62
OW	391535	609737	180677	5638.104	8780.213	3142.109	46.15	55.73
E1	45674	48271	2527	657.706	695.102	37.397	5.53	5.69
E	62552	22176	1764	900.749	319.334	-581.414	2.82	-64.55
EW	13338	15626	195	192.067	225.014	32.947	1.46	17.15
E2	181857	155638	16472	2618.741	2241.187	-377.554	9.06	-14.42
M1	6176	6618	47	88.934	95.299	6.365	0.76	7.16
M	57884	44685	3119	833.530	643.464	-190.066	5.39	-22.80
T	15234	13918	192	219.370	200.419	-18.950	1.26	-8.64
U	186097	276001	108707	2679.797	3974.414	1294.618	58.41	48.31
Total	*2144193*	*2144193*	*1094283*	*30876.379*	*30876.379*		*51.03*	
1964 - Declining								
L	1174604	651276	495504	16914.298	9378.374	-7535.923	42.18	-44.55
OW	415199	783073	149756	5978.866	11276.251	5297.386	36.07	88.60
E1	15692	37746	508	225.965	543.542	317.578	3.24	140.54
E	48299	19132	828	695.506	275.501	-420.005	1.71	-60.39
EW	10311	3863	38	148.478	55.627	-92.851	0.37	-62.54
E2	171006	181154	19797	2462.486	2608.618	146.131	11.58	5.93
M1	1197	11038	17	17.237	158.947	141.710	1.42	822.14
M	73014	99553	9196	1051.402	1433.563	382.162	12.59	36.35
T	8734	22872	160	125.770	329.357	203.587	1.83	161.87
U	226140	334489	136958	3256.416	4816.642	1560.226	60.56	47.91
Total	*2144196*	*2144196*	*812762*	*30876.422*	*30876.422*		*37.91*	
1978 - Rising								
L	1193108	969443	798445	17180.755	13959.979	-3220.776	66.92	-18.75
OW	485135	598653	211709	6985.944	8620.603	1634.659	43.64	23.40
E1	41662	47220	2190	599.933	679.968	80.035	5.26	13.34
E	28078	21822	861	404.323	314.237	-90.086	3.07	-22.28
EW	32130	15804	529	462.672	227.578	-235.094	1.65	-50.81
E2	99243	156173	7475	1429.099	2248.891	819.792	7.53	57.36
M1	3279	6450	23	47.218	92.880	45.662	0.70	96.71
M	58320	45548	3261	839.808	655.891	-183.917	5.59	-21.90
T	12741	14224	185	183.470	204.826	21.355	1.45	11.64
U	190498	268857	110043	2743.171	3871.541	1128.370	57.77	41.13
Total	*2144194*	*2144194*	*1134721*	*30876.394*	*30876.394*		*52.92*	

Wetland Community	Number of Cells			Area (ha)			Percent (%)	
	Actual	Predicted	Correctly Predicted	Actual	Predicted	Difference in Area (P-A)	Correctly Predicted Cells	Difference in Area
1985 - Rising								
L	1201068	1200941	1023672	17295.379	17293.550	-1.829	85.23	-0.01
OW	525670	525550	278801	7569.648	7567.920	-1.728	53.04	-0.02
E1	42648	42458	2647	614.131	611.395	-2.736	6.21	-0.45
E	16015	15925	377	230.616	229.320	-1.296	2.35	-0.56
EW	23343	23394	1222	336.139	336.874	0.734	5.23	0.22
E2	108803	108971	19408	1566.763	1569.182	2.419	17.84	0.15
M1	6337	6333	125	91.253	91.195	-0.058	1.97	-0.06
M	31758	31707	2072	457.315	456.581	-0.734	6.52	-0.16
T	9483	9574	165	136.555	137.866	1.310	1.74	0.96
U	179071	179343	96399	2578.622	2582.539	3.917	53.83	0.15
Total	*2144196*	*2144196*	*1424888*	*30876.422*	*30876.422*		*66.45*	
1995 - Declining								
L	1199417	1199570	1026504	17271.605	17273.808	2.203	85.58	0.01
OW	441929	442331	208169	6363.778	6369.566	5.789	47.10	0.09
E1	31632	31247	1377	455.501	449.957	-5.544	4.35	-1.22
E	22143	21986	765	318.859	316.598	-2.261	3.45	-0.71
EW	4404	4462	42	63.418	64.253	0.835	0.95	1.32
E2	173032	172375	48767	2491.661	2482.200	-9.461	28.18	-0.38
M1	7608	7343	161	109.555	105.739	-3.816	2.12	-3.48
M	61678	62247	7267	888.163	896.357	8.194	11.78	0.92
T	11650	11714	288	167.760	168.682	0.922	2.47	0.55
U	190747	190965	100425	2746.757	2749.896	3.139	52.65	0.11
Total	*2144240*	*2144240*	*1393765*	*30877.056*	*30877.056*		*65.00*	
1999 - Declining								
L	1195434	1112930	931831	17214.250	16026.192	-1188.058	77.95	-6.90
OW	433775	469448	188427	6246.360	6760.051	513.691	43.44	8.22
E1	35209	28881	1043	507.010	415.886	-91.123	2.96	-17.97
E	22176	19417	640	319.334	279.605	-39.730	2.89	-12.44
EW	4015	4817	24	57.816	69.365	11.549	0.60	19.98
E2	181271	157166	24328	2610.302	2263.190	-347.112	13.42	-13.30
M1	5423	9669	88	78.091	139.234	61.142	1.62	78.30
M	62051	73251	6765	893.534	1054.814	161.280	10.90	18.05
T	10463	15023	105	150.667	216.331	65.664	1.00	43.58
U	194252	253467	105313	2797.229	3649.925	852.696	54.21	30.48
Total	*2144069*	*2144069*	*1258564*	*30874.594*	*30874.594*		*58.70*	

Table K-10: Overall Accuracy of v2.2 *

Wetland Community	Average	Declining	Rising
Overall	*49.039*	*48.303*	*49.531*
L	61.279	60.067	62.087
OW	42.688	39.754	44.645
E1	4.351	3.100	5.186
E	2.699	2.300	2.965
EW	1.075	0.483	1.469
E2	10.106	12.499	8.511
M1	1.081	1.521	0.788
M	8.062	11.749	5.605
T	1.218	1.418	1.085
U	57.786	57.389	58.051

* Based on percentage of correctly predicted cells

* Results for 1985 and 1995 were excluded from the averages

TABLE K-11: VEGETATION TRANSITION MODEL v3.0 ACCURACY ASSESSMENT

WETLAND COMMUNITY	NUMBER OF CELLS			AREA (HA)			PERCENT (%)	
	ACTUAL	PREDICTED	CORRECTLY PREDICTED	ACTUAL	PREDICTED	DIFFERENCE IN AREA (P-A)	CORRECTLY PREDICTED CELLS	DIFFERENCE IN AREA
1955 - RISING								
L	1202585	1207333	1194468	17317.224	17385.595	68.371	99.33	0.39
OW	391535	425727	340459	5638.104	6130.469	492.365	86.95	8.73
E1	45674	37810	5065	657.706	544.464	-113.242	11.09	-17.22
E	62552	19895	2637	900.749	286.488	-614.261	4.22	-68.19
EW	13338	23717	1072	192.067	341.525	149.458	8.04	77.82
E2	181855	174759	89916	2618.712	2516.530	-102.182	49.44	-3.90
M1	6176	7748	471	88.934	111.571	22.637	7.63	25.45
M	57886	37606	6459	833.558	541.526	-292.032	11.16	-35.03
T	15255	7693	1109	219.672	110.779	-108.893	7.27	-49.57
U	186169	220737	146112	2680.834	3178.613	497.779	78.48	18.57
Total	*2163025*	*2163025*	*1787768*	*31147.560*	*31147.560*		*82.65*	
1964 - DECLINING								
L	1193343	1201081	1186972	17184.139	17295.566	111.427	99.47	0.65
OW	415199	335913	303153	5978.866	4837.147	-1141.718	73.01	-19.10
E1	15692	32491	1241	225.965	467.870	241.906	7.91	107.05
E	48299	23827	2644	695.506	343.109	-352.397	5.47	-50.67
EW	10311	4913	96	148.478	70.747	-77.731	0.93	-52.35
E2	171004	231585	107029	2462.458	3334.824	872.366	62.59	35.43
M1	1197	8149	56	17.237	117.346	100.109	4.68	580.79
M	73016	98126	22732	1051.430	1413.014	361.584	31.13	34.39
T	8759	19135	3234	126.130	275.544	149.414	36.92	118.46
U	226206	207806	172554	3257.366	2992.406	-264.960	76.28	-8.13
Total	*2163026*	*2163026*	*1799711*	*31147.574*	*31147.574*		*83.20*	
1978 – RISING								
L	1211847	1202806	1190701	17450.597	17320.406	-130.190	98.26	-0.75
OW	485135	459426	383297	6985.944	6615.734	-370.210	79.01	-5.30
E1	41662	37650	3229	599.933	542.160	-57.773	7.75	-9.63
E	28078	19874	1379	404.323	286.186	-118.138	4.91	-29.22
EW	32130	22501	2444	462.672	324.014	-138.658	7.61	-29.97
E2	99246	154673	48598	1429.142	2227.291	798.149	48.97	55.85
M1	3279	7002	124	47.218	100.829	53.611	3.78	113.54
M	58320	37421	7284	839.808	538.862	-300.946	12.49	-35.84
T	12769	9284	1318	183.874	133.690	-50.184	10.32	-27.29
U	190559	212388	138203	2744.050	3058.387	314.338	72.53	11.46
Total	*2163025*	*2163025*	*1776577*	*31147.560*	*31147.560*		*82.13*	
1985 - RISING								
L	1219807	1219873	1208518	17565.221	17566.171	0.950	99.07	0.01
OW	525670	525493	448026	7569.648	7567.099	-2.549	85.23	-0.03
E1	42649	43059	6704	614.146	620.050	5.904	15.72	0.96
E	16015	16099	740	230.616	231.826	1.210	4.62	0.52
EW	23343	23426	2965	336.139	337.334	1.195	12.70	0.36
E2	108803	108449	50607	1566.763	1561.666	-5.098	46.51	-0.33
M1	6337	6382	377	91.253	91.901	0.648	5.95	0.71
M	31771	31933	5117	457.502	459.835	2.333	16.11	0.51
T	9494	9442	1574	136.714	135.965	-0.749	16.58	-0.55
U	179137	178870	131586	2579.573	2575.728	-3.845	73.46	-0.15
Total	*2163026*	*2163026*	*1856214*	*31147.574*	*31147.574*		*85.82*	

Wetland Community	Number of Cells: Actual	Predicted	Correctly Predicted	Area (ha): Actual	Predicted	Difference in Area (P-A)	Percent (%): Correctly Predicted Cells	Difference in Area
1995 - Declining								
L	1218109	1218033	1210672	17540.770	17539.675	-1.094	99.39	-0.01
OW	441929	441585	355392	6363.778	6358.824	-4.954	80.42	-0.08
E1	31632	31682	3015	455.501	456.221	0.720	9.53	0.16
E	22142	22156	1051	318.845	319.046	0.202	4.75	0.06
EW	4404	4374	32	63.418	62.986	-0.432	0.73	-0.68
E2	173032	172950	86743	2491.661	2490.480	-1.181	50.13	-0.05
M1	7608	7705	1150	109.555	110.952	1.397	15.12	1.27
M	61681	61922	15335	888.206	891.677	3.470	24.86	0.39
T	11671	11700	3278	168.062	168.480	0.418	28.09	0.25
U	190818	190919	161391	2747.779	2749.234	1.454	84.58	0.05
Total	*2163026*	*2163026*	*1838059*	*31147.574*	*31147.574*		*84.98*	
1999 Declining								
L	1214173	1216621	1208014	17484.091	17519.342	35.251	99.49	0.20
OW	433773	373387	342017	6246.331	5376.773	-869.558	78.85	-13.92
E1	35209	28670	5119	507.010	412.848	-94.162	14.54	-18.57
E	22176	19837	1531	319.334	285.653	-33.682	6.90	-10.55
EW	4015	4419	47	57.816	63.634	5.818	1.17	10.06
E2	181270	195893	106620	2610.288	2820.859	210.571	58.82	8.07
M1	5423	8167	1730	78.091	117.605	39.514	31.90	50.60
M	62051	88869	22834	893.534	1279.714	386.179	36.80	43.22
T	10485	17340	3766	150.984	249.696	98.712	35.92	65.38
U	194323	209695	159807	2798.251	3019.608	221.357	82.24	7.91
Total	*2162898*	*2162898*	*1851485*	*31145.731*	*31145.731*		*85.60*	

Table K-12: Overall Accuracy of v3.0 *

Wetland Community	Average	Declining	Rising
Overall	*83.398*	*84.403*	*82.393*
L	99.135	99.479	98.790
OW	79.456	75.930	82.982
E1	10.322	11.224	9.420
E	5.376	6.189	4.564
EW	4.436	1.051	7.822
E2	54.954	60.703	49.206
M1	11.997	18.290	5.704
M	22.895	33.966	11.824
T	22.608	36.420	8.796
U	77.382	79.260	75.504

* Based on percentage of correctly predicted cells

* Results for 1985 and 1995 were excluded from the averages

TABLE K-13: MAGNITUDE OF DIFFERENCE BETWEEN ACTUAL AND PREDICTED WETLAND VALUES

DIFFERENCE (A – P)*	PERCENTAGE OF CELLS INCORRECTLY PREDICTED (%)							AVERAGE PERCENTAGE **		
	1945	1955	1964	1978	1985	1995	1999	OVERALL	DECLINING	RISING
RULE-BASED MODEL v1.0										
< -2		6.221	6.245	9.050	6.504	1.079	4.156	5.543	3.827	7.258
-2		9.558	8.612	9.896	6.604	3.323	4.823	7.136	5.586	8.686
-1		41.029	60.998	41.101	28.148	45.939	49.820	44.506	52.252	36.759
0		-	-	-	-	-	-	-	-	-
1		24.251	7.353	23.246	39.074	24.135	26.490	24.091	19.326	21.893
2		14.448	10.847	11.594	12.120	11.817	8.546	11.562	10.403	10.041
> 2		4.493	5.946	5.113	7.550	13.706	6.165	7.162	8.606	5.719
RULE-BASED MODEL v1.1										
< -2		12.928	8.347	24.300	9.050	2.420	10.220	11.211	6.996	15.426
-2		18.363	12.992	11.577	11.233	6.501	11.272	11.990	10.255	0.275
-1		20.214	36.130	16.335	13.425	15.924	25.101	21.188	25.718	0.813
0		-	-	-	-	-	-	-	-	-
1		21.422	10.497	18.780	22.295	19.195	20.315	18.751	16.669	20.832
2		15.877	22.601	15.418	22.689	22.773	17.861	19.536	21.078	17.995
> 2		11.195	9.432	13.591	21.308	33.187	15.231	17.324	19.283	15.365
PROBABILITY MODEL v2.0										
< -2	28.698	27.906	34.572	29.530	21.893	24.908	26.622	29.466	30.597	28.711
-2	5.786	5.245	7.505	4.883	3.145	5.152	5.560	5.796	6.533	5.305
-1	24.405	24.235	23.135	25.594	24.971	19.996	20.169	23.508	21.652	24.745
0	-	-	-	-	-	-	-	-	-	-
1	12.418	15.271	12.264	16.854	25.097	19.980	17.838	14.929	15.051	14.848
2	4.008	4.539	4.164	3.454	3.206	5.144	5.084	4.250	4.624	4.000
> 2	24.685	22.804	18.360	19.684	21.688	24.821	24.727	22.052	21.543	22.391
PROBABILITY MODEL v2.1										
< -2	26.573	25.873	34.546	27.475	21.452	24.816	29.786	28.851	32.166	26.640
-2	5.440	4.826	7.368	4.511	3.046	5.189	5.757	5.580	6.563	4.926
-1	22.913	23.143	23.074	24.485	25.478	20.027	19.112	22.545	21.093	23.514
0	-	-	-	-	-	-	-	-	-	-
1	16.294	18.829	12.966	20.324	25.503	20.012	18.179	17.318	15.573	18.482
2	4.081	4.583	4.072	3.519	3.073	5.204	4.669	4.185	4.371	4.061
> 2	24.701	22.744	17.973	19.687	21.447	24.752	22.496	21.520	20.235	22.377
PROBABILITY MODEL v2.2										
< -2	25.404	28.419	27.672	30.579	24.925	25.902	31.707	28.756	29.689	28.134
-2	4.860	4.945	5.434	4.668	3.775	5.870	5.802	5.142	5.618	4.824
-1	35.693	27.893	40.735	28.348	21.317	18.241	19.768	30.487	30.252	30.645
0	-	-	-	-	-	-	-	-	-	-
1	10.697	13.353	9.268	14.531	21.380	18.183	15.645	12.699	12.456	12.860
2	3.228	4.205	3.329	3.248	3.779	5.924	4.490	3.700	3.910	3.561
> 2	20.118	21.185	13.561	18.625	24.825	25.881	22.589	19.216	18.075	19.976
VEGETATION TRANSITION MODEL v3.0										
< -2	22.834	32.034	33.927	28.364	26.759	37.104	22.834	22.834	32.034	33.927
-2	20.514	21.609	12.164	8.797	14.554	22.916	20.514	20.514	21.609	12.164
-1	10.805	9.927	12.751	12.831	8.788	10.462	10.805	10.805	9.927	12.751
0	-	-	-	-	-	-	-	-	-	-
1	11.649	6.212	9.197	12.703	8.791	5.136	11.649	11.649	6.212	9.197
2	10.628	14.835	8.482	8.745	14.473	9.067	10.628	10.628	14.835	8.482
> 2	23.571	15.383	23.480	28.560	26.634	15.314	23.571	23.571	15.383	23.480

* - indicates the predicted value is drier than (above) the actual value; + indicates the predicted value is wetter than (below) the actual value
** average for probability and transition models excludes the results for 1985 and 1995