DISSOLVED INORGANIC NITROGEN REMOVAL EFFICIENCY OF THE REED BEDS SURROUNDING LAKE MOGAN USING MODELING APPROACHES

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MUSTAFA GÖKMEN

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Approval of the Graduate School of Natural and Applied Sciences

Prof. Dr. Canan Özgen
Director

I certify that this thesis satisfies all requirements as a thesis for the degree of Master of Science

Prof. Dr. Semra Kocabıyık
Head of Department

This is to certify that we have read this thesis and that in our opinion it is fully adequate, in scope and quality, as a thesis for the degree of Master of Science.

Assoc. Prof. Dr. Meryem Beklioğlu
Supervisor

Examining Committee Members

Prof. Dr. Musa Doğan

Prof. Dr. Zeki Kaya

Assoc. Prof. Dr. Mustafa Oğuz

Assoc. Prof. Dr. C. Can Bilgin

Assoc. Prof. Dr. Meryem Beklioğlu

ABSTRACT

DISSOLVED INORGANIC NITROGEN REMOVAL EFFICIENCY OF THE REED BEDS SURROUNDING LAKE MOGAN USING MODELING APPROACH

Gökmen, Mustafa

M.Sc., Department of Biology

Supervisor: Assoc. Prof. Dr. Meryem Beklioğlu

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In this study, yearly and seasonally nitrogen retention dynamics of reed beds surrounding Lake Mogan were investigated by comparing surface aerial nitrogen load and in-lake concentrations. The analyses were performed separately for nitrate-N, ammonium-N and dissolved inorganic nitrogen (sum of nitrate-N and ammonium-N) to reveal differences between them in terms of retention dynamics. 1998, 1999 and 2002 were relatively high-load years in terms of DIN-input to reed beds surrounding Lake Mogan, compared with the DIN-loadings of 1997, 2000 and 2001. A significant difference was observed between NO_3-N input and output for the relatively high-load years to Lake Mogan reed beds indicating significantly high NO_3-N retention rates for that periods, while no significant difference was observed in the relatively low-load years. Also, a clear linear relationship (R^2 = 0.975) was found between amount of NO_3-N retention and amount of NO_3-N input to the system. NH_4-N input and output were not significantly different in none of the study years.

Then, a dynamic “Wetland Nitrogen Model” was utilized to model dissolved inorganic nitrogen removal capacity of the reed beds surrounding Lake Mogan. The model was firstly calibrated and validated using data sets of different study years and then used for prediction under wet and dry year scenarios. The model predictions revealed that NO_3-N retention efficiency was distinctively higher in wet rather than the dry year conditions since the reed beds might have limited denitrification capacity in dry years due to unavailability of enough NO_3-N load.

Finally, the land-use changes occurred in the closer catchment of Lake Mogan and the potential risk areas for non-point nitrogen input to Lake Mogan were determined using aerial photos of the region and Geographical Information Systems (GIS). It was observed that highest potential risk area for non-point nitrogen input around the lake was north-east of the lake whereas, north end of the lake was least potential risk area.

Key words: Lake Mogan, reed bed, nitrate, ammonium, dissolved inorganic nitrogen, model, GIS

ÖZ

MOGAN GÖLÜNÜ ÇEVRELEYEN SAZLIK ALANLARININ ÇÖZÜNMÜŞ İNORGANİK AZOT BERTARAFINDAKİ ETKİNLİĞİNİN MODELLEME YAKLAŞIMLARI KULLANILARAK BELİRLENMESİ

Gökmen, Mustafa

Yüksek Lisans, Biyoloji Bölümü

Tez Danışmanı: Doç. Dr. Meryem Beklioğlu

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Bu çalışmada, Mogan Gölü'nü çevreleyen sazlık alanların yıllık ve mevsimsel azot tutma (retention) dinamikleri, sözkonusu sazlık alana gelen yüzeysel azot yükü ile göl-içi azot konsantrasyonları karşılaştırılarak araştırılmıştır. Mogan Gölü sazlık alanlarında, nitrat-N, amonyum-N ve çözünmüş inorganik azot (DIN) gibi farklı azot bileşiklerininin tutulmasındaki farklı dinamikleri ortaya koymak amacıyla analizler herbir azot bileşiği için farklı olarak gerçekleştirilmiştir. Çalışma 1997-2002 yılları arasındaki verileri kapsarken, Mogan Gölü sazlık alanlarına 1998, 1999 ve 2002 yıllarında diğer çalışma yılları ile karşılaştırıldığında görece daha yüksek DIN girdisi olmuştur.

Mogan Gölü sazlık alanlarına görece daha yüksek azot yükü gözlenen yıllarda NO_3-N girdisi ve çıktısı arasında önemli bir farklılık görülürken görece düşük azot yüklü yıllarda böyle önemli farklılık görülmemiştir. Ayrıca, sazlık alana nitrat girdisi miktarı ile sazlık alanın nitrat tutma kapasitesi arasında doğrusal bir ilişki gözlenmiştir (R^2 = 0.975). Diğer taraftan, çalışma yıllarının hiçbirinde amonyum girdisi ve çıktısı arasında önemli bir farklılık görülmemiştir.

Çalışma yıllarına ait verilerin sazlık alandaki azot bertarafı dinamikleri açısından analiz edilmesinin ardından dinamik bir “Sulakalan Azot Modeli”, Mogan Gölü sazlık alanlarının çözünmüş inorganik azot tutma kapasitesini modellemek amacıyla uygulanmıştır. Sözkonusu dinamik sulakalan modeli çalışma yıllarına ait veriler kullanılarak kalibre edilip doğrulanarak (validation), ıslak ve kurak yıl senaryolarında sazlık alanın azot tutma kapasitesindeki değişimi tahmin etmek amacıyla kullanılmıştır.

Son olarak, Mogan Gölü ve yakın havzasına ait hava fotoğrafları ve Coğrafi Bilgi Sistemleri (CBS) kullanılarak, Mogan Gölü çevresindeki geniş tarım arazilerinden kaynaklanan yayılı azot girdisinin muhtemel olarak gölün hangi bölgelerinden gerçekleştiği belirlenmiştir. Gölün kuzey doğu kıyısı, yayılı azot girdisi bakımından en riskli bölge olarak bulunurken, en az riskli bölge olarak gölün kuzey ucu bulunmuştur.

Anahtar kelimeler: Mogan Gölü, sazlı alan, nitrat, amonyum, çözünmüş inorganik azot, model, CBS

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF ABBREVIATIONS

EU	European Union
DSI	State Hydraulic Works
EIE	General directorate of Electrical Power, Resource Survey & Development Administration
SIS	State Institute of Statistics
SPA	Specially protected area
IBA	Important bird area
GIS	Geographical information systems
N	Nitrogen
N_2-N	Nitrite-nitrogen
N_3O-N	Nitrate-nitrogen
NH_4-N	Ammonium-nitrogen
DIN	Dissolved inorganic nitrogen
TN	Total nitrogen
P	Phosphorus
TP	Total phosphorus
SRP	Soluble reactive phosphate
Eh	Soil redox potential
Hyd. cond.	Hydraulic conductivity

CHAPTER 1

INTRODUCTION

Non-point nutrients include nitrogen (N), phosphorus (P), and together with the non-point pollutants of heavy metals, and other chemicals from fertilizers, pesticides, herbicides, animal wastes, overland flow wastewater treatment systems, urban stormwater, and other sources (Muscutt *et al.*, 1993). Given the negative impacts of increasing nitrogen loads, the role of freshwater systems, especially wetlands, for reducing local and downstream nitrogen concentrations are becoming increasingly important.

Nitrogen retention in wetlands is known to be a function of denitrification, sedimentation and plant uptake. Of these, nitrification and subsequent denitrification at the aerobic-anaerobic interface of the wetland is thought to be the primary means of nitrogen removal in wetlands (Neely and Baker, 1989; Brix and Schierup, 1989; Reddy and D'Angelo, 1994). Denitrification is the process whereby facultative anaerobic bacteria produce N_2-N or N_2O-N gas by using nitrate (NO_3-N) or nitrite (NO_2-N) as terminal electron acceptors (Knowles et al., 1982). Nitrification and denitrification processes are mainly regulated by presence of a usable carbon source, presence of dissolve oxygen and temperature. Hydrology

(Cooke, 1994), the form, concentration and timing of nitrogen inputs (Phipps and Crumpton, 1994) and physicochemical characteristics of the wetland (Spieles and Mitsch, 2000) can influence the rate of denitrification in wetlands. Efficiency of denitrification is enhanced in wetlands by occurrence of anoxic and oxic conditions with inundation at high lake levels (Saunders & Kalff, 2001; Sanchez-Carrillo & Alvarez-Cobeleas, 2001). Diffusive transport of N species from the aerobic water and substrate to the anaerobic substrate layer is also an important regulator of denitrification (Martin and Reddy, 1997). Different wetland substrates can have remarkably different characteristics (Ambus and Lowrance, 1991) and together with water chemistry can dictate the system nitrogen removal capacity (Spieles and Mitsch, 2000). The hydraulic loading and hydroperiod of the wetland (wet-period to dry-period) can also influence its nitrogen removal effectiveness.

When considering the seasonality of nitrogen retention in wetlands, temperature, one of the main regulating factors becomes important in terms of seasonal denitrification potential of temperate wetlands, particularly in seasonally cold climates (Spieles and Mitch, 2000). While biological removal of N is most efficient at 20-25 ^{o}C (Sutton et al., 1975), water temperatures lower than 15 ^{o}C or greater than 30 ^{o}C have been shown to drastically reduce the growth rate of nitrifying bacteria, thus limiting the rate of the denitrification (Reddy and Patrick, 1984). During the winter months in cold climate wetlands, there is little nitrogen assimilation, a decrease in the release and mineralization of organic nitrogen, and a decrease in the rate of nitrification and denitrification; thus a decrease in nitrogen removal capacity in the winter months is to be expected (Kadlec and Knight, 1996). However, nitrate removal can occur all year long even in cold climates (Brodrick *et al.*, 1988; Phipss and Crumpton, 1994; Hosomi *et al.*, 1994) and significant denitrification has been shown to occur at temperatures as low as 5^{o}C in the laboratory (Brodrick *et al.*, 1988).

Cooke (1994) also noted that nitrate release from wetlands can occur during periods of high flow and suggested that changes in soil redox potential (Eh) result in desorption of ammonium, which is subsequently flushed from the system.

It is suggested that the combined effects of seasonal fluctuations of temperature, nitrogen and hydraulic loading rates cause significant changes in the nitrogen-removal potential of wetlands.

1.1. Sources of nitrogen

During the last century, human activities have dramatically changed the global nitrogen cycle. Practices such as agricultural fertilization, fossil fuel combustion and clearing and conversion of land have dramatically increased the supply of nitrogen to freshwaters (Jansson *et al.*, 1994, Vitousek *et al.*, 1997 and Moffat *et al.*, 1998).

From a land use perspective, agricultural activities have been identified as major sources of non-point source of nitrogen & sediment, animal wastes, plant nutrients, crop residues, inorganic salts and minerals, pesticides (Viessman and Hammer, 1993) and are known to impact water quality. Residential/urban/built-up areas are another dominant factor in generating large amounts of point source nutrients from storm water discharge. Since nitrogen is more active, mobile and soluble element than phosphorus, a considerable amount of N is usually delivered into the drainage and surface runoff (Tumas *et al.*, 2000).

1.2. Effects of nitrogen load to freshwater systems

Nutrient enrichment, or cultural eutrophication (Moss et al., 1998), has been a serious threat to most of the world's lakes as eutrophication leads to an increase in the phytoplankton biomass, especially potentially toxic cyanobacteria, and changes in the community structure and ecosystem functioning. Nutrient limitation plays an important role in the eutrophication of aquatic systems especially through affecting the phytoplankton dynamics (Sommer, 1989). Under pristine state, though phosphorus is the limiting nutrient in most lakes (Wetzel, 1993), nitrogen becomes limiting when there is excessive phosphorus input. Increasing nitrate (NO_3-N) concentrations is of particular concern because of associated human health risks. Finally, nitrate is known to contribute to lake acidification.

In Europe, increasing public concerns about steadily increasing nitrate concentrations in drinking water resources, and disturbance of aquatic ecosystems by eutrophication. The best known examples were the south-eastern North Sea, the Adriatic Sea and Venice lagoons, where algae blooms were recorded more frequently from the 70's. The EU Commission Report (COM 2002-407) have triggered the European Union for taking an action to improve water quality. In this context, the Nitrates Directive (Council Directive 91/676/EEC) concerning the protection of waters against pollution caused by nitrates from agricultural sources was adopted on 12th December 1991. The Report states the possible impacts of non-point nitrogen pollution as follows;

- A part of nitrogen loss to the environment (50-80%) is recycled to water and soils, causing groundwaters enrichment, eutrophication of surface waters, in synergy with phosphorus, and contributing to "acid rain" damages on terrestrial flora and soils; another part 20-50% is "denitrified" into inert nitrogen gas (and some N_2O with a greenhouse gas effect), by soil and sediment bacteria, or by natural chemical reduction in certain types of soils and groundwaters.
- Mineral fertilizers directly introduce ammonium and nitrates into groundwaters by leaching, and into surface waters by run-off and subsoil "drainage". The extent of this depends on ground conditions at time of spreading.
- Organic N (in manure) uses the same "pathways", plus additional losses to the atmosphere in the form of ammonia (volatilization) and N_2O (incomplete denitrification). These range from 10% to 30% of the initial N excreted by animals, and are re-deposited on the soil and waterbodies in rain (wet deposition) or directly (dry atmospheric deposition).

1.3. The role of wetlands in nitrogen retention

Major pathways for nitrogen removal in wetlands include mineralization of organic N, ammonia volatilization, assimilation into plant biomass, absorption of ammonium onto the substrate and nitrification followed by denitrification (Reddy and Patrick, 1984).

Nitrogen removal by assimilation into plant or microbial biomass or by dissimilatory reduction to ammonium has been shown to account for 1-34% of the total N loss, with denitrification accounting for 60-95% of the total N removed (Bartlett *et al.*, 1979; Stengel *et al.*, 1987; Cooke., 1994).

1.4. Modeling of nitrogen retention in wetlands

The use of a wetland as a sink of nutrients raises the question of how we can utilize wetland as a nutrient trap? Use of a wetland model, which considers all the factors that determine the removal of nutrients from the water, can provide reliable information on the wetland capacity as a N sink.

Models can be defined as formal expressions of the essential elements of a problem in either physical or mathematical terms (Jorgensen, 1994). Classification of models is given in Table 1.1. Of these, a stochastic model contains stochastic input disturbances and random measurement errors. While, a deterministic model implies that the future response of the system is completely determined by a knowledge of the present state and future measured inputs. The classification of reductionistic and holistic models is based upon a difference in the scientific ideas behind the model. The reductionistic approach attempts to incorporate as many details of the system as possible while the holistic one try to include in the model properties of the whole system by use of general system principles.

Dynamic systems might have four classes of state. The initial state changes through the transient states to a state where the system oscillates around a steady state. Dynamic models use differential or difference equations to describe the system response to external factors.

The causal models characterize how the inputs are connected to the states and how the states are connected to each other and to the outputs of the system, whereas the black box model reflects only what changes the input will effect in the output responses. A model that relates the input of nutrient with the phytoplankton concentration in a reservoir directly is an example of a black box model.

An ecological model consists, in its mathematical formulation, of five components; forcing functions or external variables, state variables, mathematical equations, parameters and universal constants (Jorgensen *et al.*, 1994).

Forcing functions are functions or variables of an external nature that influence the state of the ecosystem. The model is used to predict what will change in the ecosystem when forcing functions are varied with time (i.e., forcing functions are the input of pollutants to the ecosystem, temperature, precipitation, etc.).

State variables describe, as the term indicates, the state of the ecosystem. The selection of variables is crucial for the model structure, but in most cases the choice is obvious. For instance, while modeling the eutrophication of a lake it is natural to include the phytoplankton density and the concentrations of nutrients.

Table 1.1 Classification of Models (Pairs of Model Types) (Jorgensen, 1989)

Type of Models	Characterization
Research models	Used as a research tool
Management models	Used as a management tool
Deterministic models	Predicted values are computed exactly
Stochastic models	Predicted values depend on probability distribution
Compartment models	Variables defining the system are quantified by means of time-dependent differential equations
Matrix models	Matrices are used in the mathematical formulation
Reductionistic models	As many relevant details are included as possible
Holistic models	General principles are used
Static models	Variables defining the system do not depend on time
Dynamic models	Variables defining the system are a function of time
Distributed models	Parameters are considered function of time and space
Lumped models	Parameters are within certain prescribed spatial locations and time, considered as constants
Linear models	First-degree equations are used consecutively
Nonlinear models	One or more of the equations are not first degree
Causal models	Inputs, states, and outputs are interrelated by use of causal relations
Black box models	Input disturbances affect only the output responses. No causality is required.
Autonomous models	Derivatives do not explicitly depend on the independent variable (time)

In the case of Lake Mogan Wetland Model, a dynamic model approach was followed to be able to describe the dynamic nature of nitrogen processes in the wetland.

Most models contain more state variables than are directly required for purpose of management, because the relationships are so complex that they require the introduction of additional state variables. For instance, it would be sufficient in many eutrophication models to relate the input of one nutrient with the phytoplankton density, since this variable is influenced by many other factors, (i.e., temperature, hydrology of the water body, zooplankton density, solar radiation, transparency of the water, etc.) (Jorgensen, 1989).

The biological, chemical, and physical processes in the ecosystem are represented in the model by means of mathematical equations. They are the relations between forcing functions and state variables. The same type of processes can be found in many ecosystems, which implies that the same equations can be used in different models. (Jorgensen, 1989)

The mathematical representations of processes in the ecosystem contains coefficients or parameters which can be considered as a constant for a specific ecosystem or part of ecosystem. Only a few parameters are known exactly and so it is necessary to calibrate the others.

Most models also contain universal constants such as the gas constant or molecular weights. Such constants are of course not subject to calibration. (Jorgensen, 1989)

1.4.1. Modeling procedure

A tentative modeling procedure is presented in the following Figure 1.1.

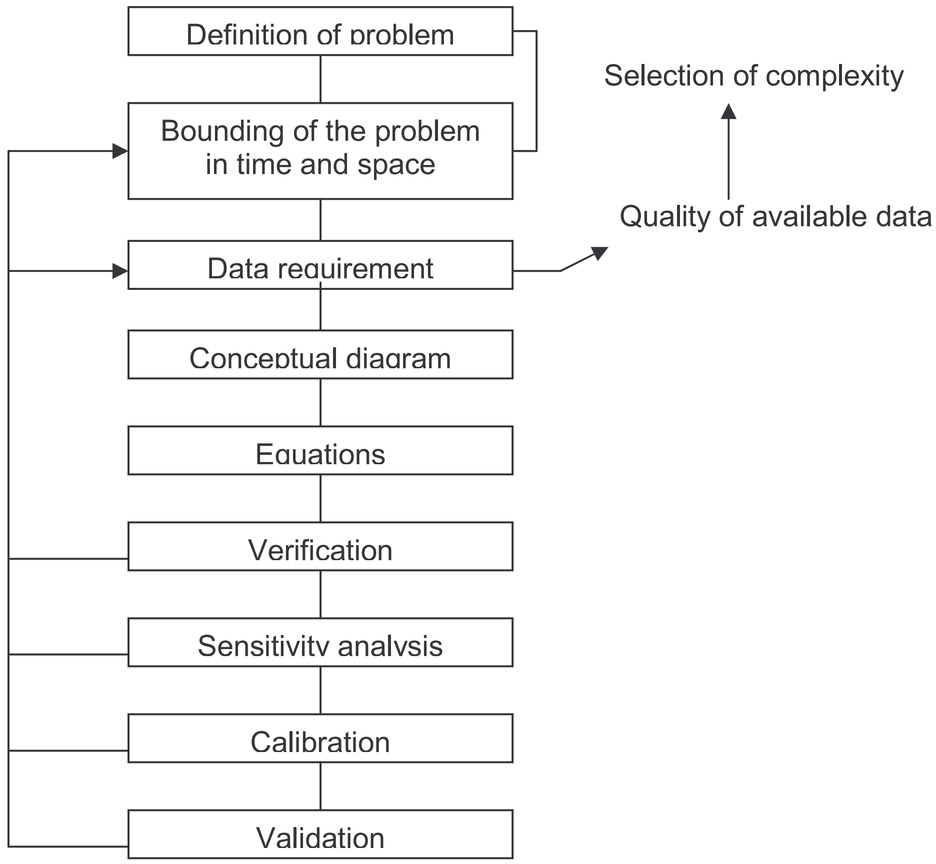


Figure 1.1. General modeling approach.

The primary focus of all research at all times is to define the problem (Jorgensen, 1994). Then, the definition of the problem must be bound by the constituents of space, time and subsystems. It has been argued that a more complex should be able to account more accurately for the complexity of the real system, but it is not true (Jorgensen, 1989). As increasing numbers of parameters are added to the model there will be an increase in the uncertainty. Once the model complexity has been selected, at least for the first attempt, it is then possible to conceptualize the model deciding on which state variables and processes are required in the model. For most processes a mathematical description is available, and most of the parameters have, at least within limits, known values from the literature. Biological parameters can most often not be determined with the same accuracy as chemical or physical parameters owing to changing and uncontrolled experimental conditions (Jorgensen, 1989). Consequently, calibration by the application of a set of measured data is always required. However, the calibration of several parameters is not realistic. Therefore, it is recommended that sound values from the literature be used for all parameters, and that a sensitivity analysis of the parameters be made before the calibration. Then the most sensitive parameters should be selected for an acceptable calibration.

By calibration it is attempted to find the best accordance between computed and observed state variables by variation of a number of parameters. The calibration may be carried out by trial and error or by use of software developed to find the parameters that give the best fit. The influence of the ecological processes that are of minor importance to the state variables under consideration are not included in the model, while they can, to a certain extent, be considered by the calibration where the results of the model are compared with observations in the ecosystems.

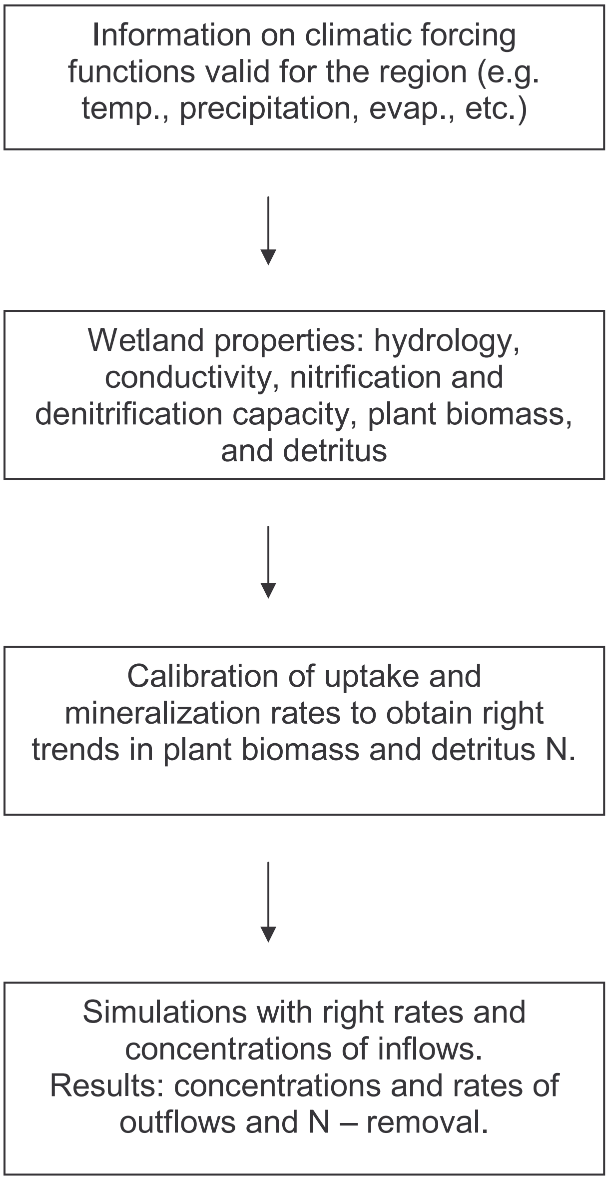


Figure 1.2. Procedure for development of a wetland model.

This might also explain why the parameters have different values in the same model when used for different ecosystems. If a reasonable fit cannot be achieved with realistic parameters the entire model should be questioned instead of forcing the model to fit observations.

After the calibration it is important to validate the model, preferably against a series of measurements from a period with changed conditions, for example, with changed external loading or climatic conditions. Calibration and thereby sensitivity analysis might sometimes be redundant, when the parameters already are known with sufficient accuracy (Jorgensen, 1989).

Modeling should, however, be considered as an iterative process as it is always open for improvement with new ideas.

1.5. Use of Geographical Information Systems (GIS)

Land cover change is regarded as the single most important variable of global change affecting ecological systems (Vitousek, 1997) with an impact on the environment. This is at least as large as that associated with climate change (Skole, 1994). Changes in the land use result from the complex interaction of many factors including policy, management, economics, culture, human behaviour and the environment (Houghton, 1994). It is well established that land cover change has significant effects on basic processes including biogeochemical cycling and thereby on global warming (Penner, 1994), the erosion of soils and thereby on sustainable land use (Douglas, 1999), and for at least the next 100 years is likely to be the most significant variable impacting on biodiversity (Chapin et al., 2000).

A basin is the up slope area contributing flow to a given location. Such a feature is also referred to as a catchment, or watershed, and comprise part of a hierarchy in that a given basin is generally part of a larger basin (Basnyat et al., 1999). An effective catchment management planning strategy needs to be developed to obtain a sustainable development and protecting natural resources. Remote sensing (aerial photography or satellite images) provides a synoptic view of the terrestrial landscape and is used for inventorying, monitoring, and change detection analysis of environmental and natural resources (Narumalani et al., 1996). Then, Geographical Information Systems (GIS) applications serve an important tool for this kind of studies. GIS are tools for collecting, storing, retrieving at will, transforming, and displaying spatial data for a particular set of purposes (Burrough and McDonnell, 1998). GIS can be used to perform a number of fundamental spatial analysis operations. Its major advantages is that it allows the user to identify the spatial relationships between various map features. More precisely, overlay techniques allows the synthesis of different map layers, based on a database where the information is stored as a whole. Comparisons, as well as further analysis, among and between both variables and layers can then be easily performed (GIS, 1994).

Interpretation and analysis of remote sensing imagery involves the identification and/or measurement of various features (point, line, or area features) in an image in order to extract useful information about them. Much interpretation and analysis of the features in remote sensing imagery is performed manually or visually, i.e. by a human interpreter. The human mind uses the following elements when performing visual analysis: shape, size, pattern, tone (or color), texture, shadow, and association (Türker et al., 2000):

Shape: refers to the general form, structure, or outline of individual objects,

Size: of objects is a function of scale. It is important to assess the size of a target relative to the other objects in a scene,

Pattern: refers to the spatial arrangements of visibly discernable objects,

Tone: refers to the relative brightness or color of objects in an image,

Texture: refers to the arrangements and frequency of tonal variation in particular areas of an image,

Shadow: may provide an idea of the profile and relative height of a target or targets, which may make identification easier,

Association: takes into account the relationship between other recognizable objects or features in proximity to the target of interest (Türker *et al*., 2000)

As Sivertun *et al.* states, GIS can also be used as a tool to pinpoint critical non-point pollution areas. The results of such GIS analysis can be used to make decisions for better water protection as well as for erosion protection in these critical areas (Sivertun & Prange, 2003). After identification of the risky areas by the GIS analysis, it can be given a preference for which agricultural land should be removed from production from time to time. They could be changed to wetland or, in critical areas, the thresholds for the allowed amount of fertilizers can be lowered, while they can stay at their current level in non-risk areas (Sivertun *et al.*, 1988).

Schauble (1999) mentions that erosion includes not only the transport of sediment particles but also the transport of nutrients and pollutants. Both mechanisms depend on the amount of surface runoff and are therefore linked together (Sivertun *et al.*, 2003). For modeling erosion, many models have been developed. De Roo (1993) gives an overview of some important models: universal soil loss equation (USLE; Wischmeier and Smith, 1978), revised USLE (RUSLE; Renard et al., 1991), modified USLE (MUSLE87; Hensel and Bork, 1988), areal non-point source watershed environment response system (ANSWERS; Beasley and Huggins, 1982) and agricultural non-point source pollution model (AGNPS; Young et al., 1987).

As a result of empirical long-term runoff studies on test fields, an equation has been reached which estimates the long-term annual soil loss in (tons/ha) and consists of multiplied factors:

$$A = R*K*L*S*C*P$$

A: result: mean long-term annual soil loss in (tons/ha)

R: rain and surface runoff factor

K: soil erodibility factor

L: slope length factor

S: slope steepness factor

C: vegetation cover factor

P: erosion protection factor

However, use of such erosion models requires capture and processing of large amounts of data. In deed, an exact computation of the load amount is not always necessary, especially in the case of critical areas identification (Sivertun *et al.*, 2003). Such simplified GIS models, which just shows possible risk areas without computing the amount of sediment or nutrient load, is enough for the first step of erosion or pollution analysis. This simple first step GIS models can be used for a huge area, require only a limited amount of data, are relatively easy to implement in GIS software (Sivertun *et al.*, 2003). Only where the rough GIS model really finds critical areas is a second step analysis with a more sophisticated expert model or an on-site exploration necessary and it is not in the scope of this study.

1.6. Scope of the study:

One of the scopes of this study was, to investigate yearly and seasonal nitrogen retention dynamics of reed beds surrounding Lake Mogan by comparing surface nitrogen aerial load and in-lake concentrations.

Secondly, a dynamic “Wetland Nitrogen Model” was utilized to model dissolved inorganic nitrogen removal efficiency of the reed beds surrounding Lake Mogan.

Finally, it was aimed to determine the potential risk areas in the near catchment of Lake Mogan for non-point N input using Geographical Information Systems.

CHAPTER 2

STUDY SITE

2.1. Lake Mogan Wetland

Lake Mogan, located 20 km south of Ankara (39°47'N 32°47E), is a shallow (mean depth 2.1 m, max depth 3.5 m), large (5.4-6 km^2) lake with a total of 925 km^2 drainage area. The lake is an alluvial dam lake formed by damming of Imrahor River at the beginning of last century. The lake is mainly fed by four main inflows, the Sukesen brook in the north, Gölcük and Yavrucak brooks in the west and the Çölovası brook in the east (Figure 2.1). These brooks, excluding Sukesen, first run through agricultural lands and then through the reed beds before reaching the lake. The outflow of the lake empties into downstream Lake Eymir through a canal and a wetland in the north.

Sukesen brook which runs through the town before reaching the lake, used to recieve sewage effluent discharge of Gölbaşı town until 1999. Since then, the effluent has been connected to a collector. However, the west catchment of the lake has been recently opened to settlement and the sewage effluent of the houses discharged into Gölcük and Yavrucak brooks. About 18 km^2 of area within the 34 km^2 vicinity of the lake were cultivated, mainly for wheat and barley.

Besides the pressure of settlement areas, the lake is affected by agricultural practices, recreation, as well as small-scale industries in the catchment.

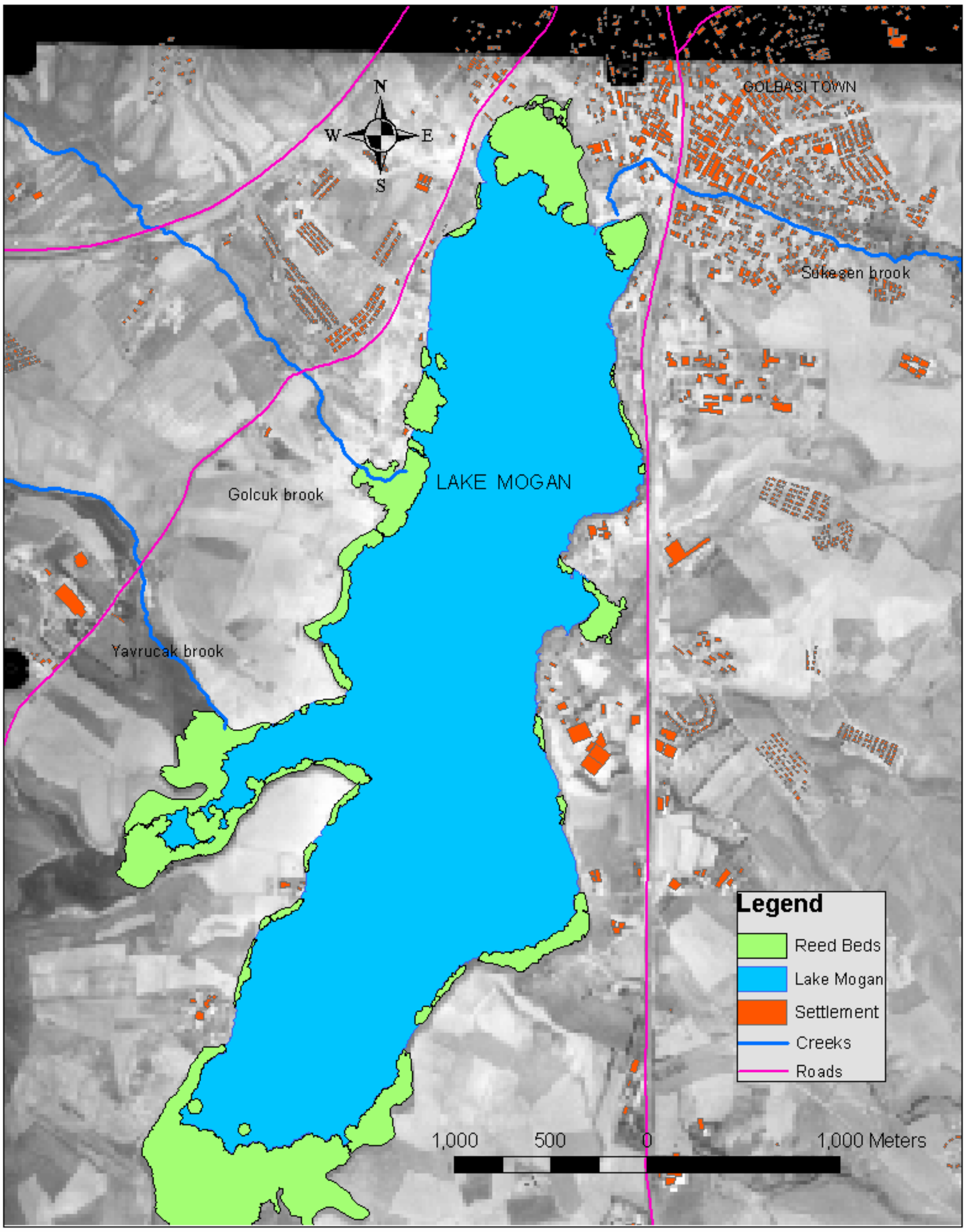


Figure 2.1. View of Lake Mogan reed beds and surrounding area (Aerial photo, 1999)

Lake Mogan is situated in the borders of Gölbaşı Municipality, which has a total (urban and rural) population size of 62602 (census data of 2000, State Institute of Statistics). While Gölbaşı town is located in the north of Lake Mogan, several villages and other settlement areas exist around the lake (Figure 2.1). The lake also serves as an important recreational site for Ankara.

The region has the Central Anatolian climatic conditions (semi-arid), cold in winter and hot in summer, with an average annual precipitation of 373 mm (max: 506 mm in May and December; min:85 mm in August) and an annual evaporation of 163 mm (DSI, 1993).

In this study, Lake Mogan reed beds surrounding the lake was taken as the study area. *Phragmites australis* dominanted the emergent plant found in Lake Mogan reed beds. The area of the reed beds was calculated using the aerial photos of the catchment taken in 1991 and 1999.

Lake Mogan and the surrounding wetlands have been famous for the high density of waterfowl and mostly host more than 20,000 waterfowl in winter (Özesmi, 1999; Kıraç et al., 1995). In 1990 Lake Mogan was given the status of “Specially Protected Area” (SPA) by the Ministry of Environment and Forest. Furthermore, the lake also obtained the status of the “Important Bird Area” (IBA) and internationally important wetland due to the rich and diverse community of waterfowl.

2.2. Hydrology of the lake

In Lake Mogan, the hydraulic residence time differs significantly between the years. Beklioglu & Tan (submitted) recorded that the longest hydraulic residence time occurred in 2001 (79 yr^{-1}) which differed significantly from the previous years, while the shortest hydraulic residence time 2 yr^{-1} was recorded in 1998. Furthermore, during 2001 the lake outflow dried out and there were no inflows to the lake from June to November (Beklioglu & Tan submitted). The same study revealed that fluctuations of the lake level occurred on different time scales. A typical annual fluctuation was observed every year with the increasing water level during the rainy periods which occurring in late winter and spring, and the decreasing water level during the dry period in summer but also in autumn and winter in some years.

CHAPTER 3

MATERIALS AND METHOD

3.1. Analysis of DIN input vs output dynamics

The lake level in meters above sea level (m a.s.l.) was recorded daily from a fixed gauge positioned in the southwest corner of the lake, and flow rates of the inflows and outflow were measured with a Gurrly current meter by the General directorate of Electrical Power, Resource Survey & Development Administration (EİE) (EİE, 2001). The mean depth was estimated by dividing monthly mean lake volume by monthly mean lake area. The hydraulic residence time was also estimated from dividing the lake volume (V_{lake}) by the volume of water flowing into the lake (V_{in}) per unit of time (Vollenweider, 1975).

Burnak & Beklioglu, 2000; Beklioglu & Tan (2000) conducted the sampling at monthly intervals during 1997 and 1998, and fortnightly from 1999 onwards. Water samples were taken from the lake, Sukesen, Gölcük, Yavrucak, Çölovası brooks and also outflow of the lake. Water for chemical analyses was stored in acid-washed 1-l Pyrex bottles. Conductivity and salinity were measured using an Orion conductivity meter to a precision of ± 1%. SRP, total phosphate (TP) and total oxidised nitrogen (nitrite and nitrate: NO_2-N+NO_3-N) were analysed using the methods described by Mackereth *et al.*, (1978) to precisions of ± 3%, ± 8% and ± 8%, respectively. Ammonium-nitrogen (NH_4-N) was determined according to Chaney & Morbach (1962) to precisions of ± 4%.

Five year of data set of dissolved inorganic nitrogen (DIN) (NO_3-N+NO_2-N and NH_4-N) and hydrological data including the years between 1997 and 2002 were used for estimating the nitrogen loads for Lake Mogan reed beds. Winter months were excluded in this study because of missing data due to ice cover caused by harsh winter conditions. Unit of g DIN m^{-2} $month^{-1}$ was selected for representing seasonal DIN input and output calculations and g DIN m^{-2} yr^{-1} for yearly nitrogen retention calculations. The DIN input-output estimations were performed separately for total oxidized nitrogen which includes nitrite and nitrate (NO_2-N+NO_3-N) and for ammonium nitrogen (NH_4-N). For ease of use, only NO_3-N will be used throughout the chapter to represent the total oxidized nitrogen. In the estimations, DIN input was taken as sum of the surface NO_3-N and NH_4-N loads coming through with surface run-off water from the catchment. The DIN output was the in-lake concentrations of NO_3-N and NH_4-N multiplied by the average lake depth.

DIN input-output estimations have been performed using the following equations;

NO_3-N input = Mean [NO_3-N] in the inflows (g/m^3) $\times$ total surface run-off water to the reed bed area, (m/month)

NH_4-N input = Mean [NH_4-N] in the inflows, (g/m^3) × total surface run-off water to the reed bed area, (m/month)

NO_3-N output = Mean [NO_3-N] in Lake Mogan, (g/m^3) × mean monthly depth of the lake, (m/month)

NH_4-N output = Mean [NH_4-N] in Lake Mogan, (g/m^3) × mean monthly depth of the lake, (m/month)

Single ANOVA and Tukey's HSD tests were utilized using SPSS Software for evaluating DIN input-output data.

3.2. Modeling

3.2.1. Modeling approach

The model utilized in this study was based upon previous model approaches by Jorgensen *et al.*, (1994). It was both dynamic in hydrological as well as in the biological part.

STELLA Software (High Performance Systems, HPS Inc.) was used to construct the model. STELLA II was an object oriented programming language that uses an iconographic interface to facilitate construction of dynamic systems structures.

In Stella Software, the essential features of the system are defined in terms of stocks, flows and auxiliary parameters. The user places the icons for each of these features in the modeling area and makes the appropriate connections between features. The functional relationships between the features are then defined by the user. These relationships can be mathematical, logical, or graphical function.

Stocks represent a reservoir of material such as population, biomass, nutrients, or money. Material flows between stocks or into and out of undefined sources and sinks (represented by ‘clouds’ at the ends of flow structures). Flows are affected by auxiliary variables, stocks, and other flows through the use of information arrows. Auxiliary variables can take the form of constants, mathematical or graphical functions, and data sets.

The model construction took into account 1 m^2 of reed bed and calculates the conversion of nitrogen in this area. The result of the model will therefore be the amount of nitrogen that can be removed, accumulated and/or released per unit of area. As it is explained by Jorgensen et al., (1994) two hydrological state variables were applied, one representing the surface layer, where nitrification can take place, and the the active zone, where a pronounced denitrification and accumulation take place. The depth of the latter layer is not very important because in most cases the limiting factor is the hydraulic conductivity. The amount of organic matter and the presence of denitrifying microorganisms in this zone are under no circumstances limiting.

Cycling of nitrogen takes place in the active layer: ammonium and nitrate were taken up by plants. Plant-N forms the detritus-N by decay and after mineralization ammonium was formed. Nitrification and denitrification are described using Michaelis-Menten equations, while the uptake of nitrate and ammonium by the plants are formulated by first-order kinetics and proportional to light. There is no differences between the uptake rates for ammonium and nitrate. The uptake was therefore proportional to the concentrations of these two nitrogen sources of the plants. The mineralization also follows a first-order kinetics. All biological rates dependent on temperature with a more pronounced dependence for the nitrification and denitrification. A conceptual diagram of hydrological part of the model is presented in Figure 3.1.

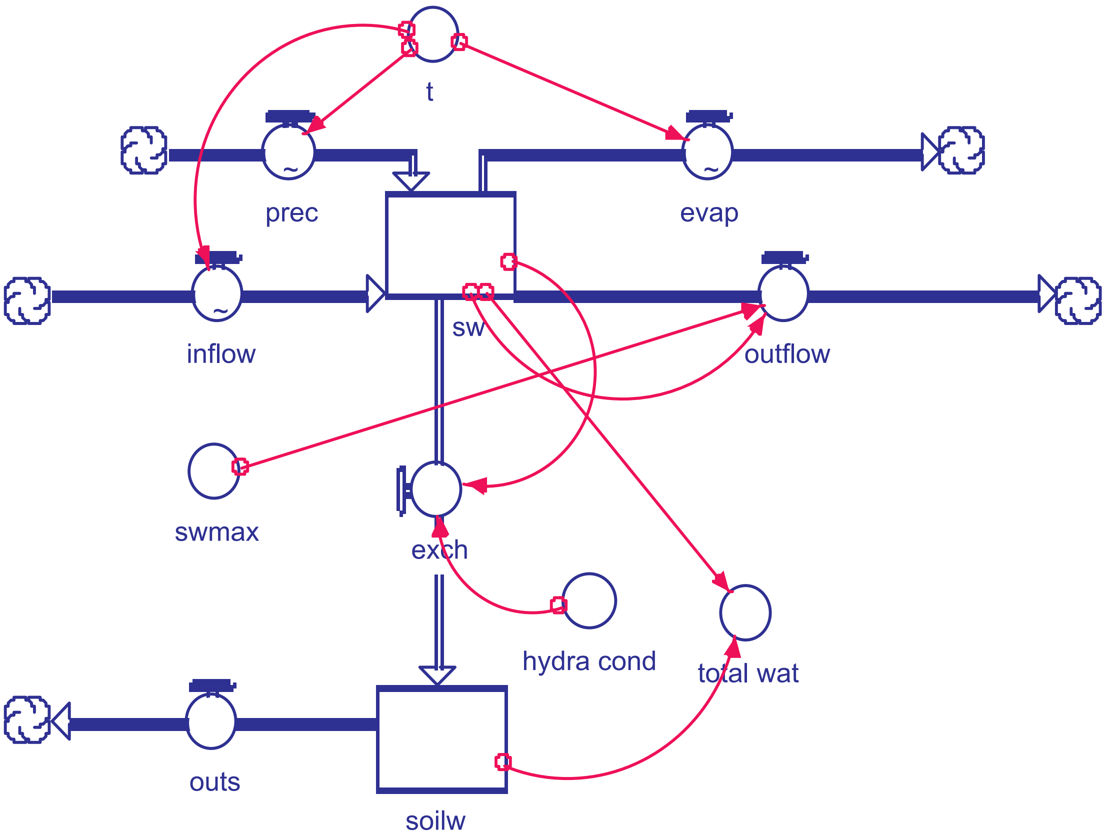


Figure 3.1. Hydrology sub-division of the conceptual diagram of the wetland model (prec: precipitation; evap: evaporation; inflow: surface water inflow; outflow: surface water outflow; sw: surface water; soilw: soil water; hydra cond: hydraulic conductivity) (Jorgensen, 1994)

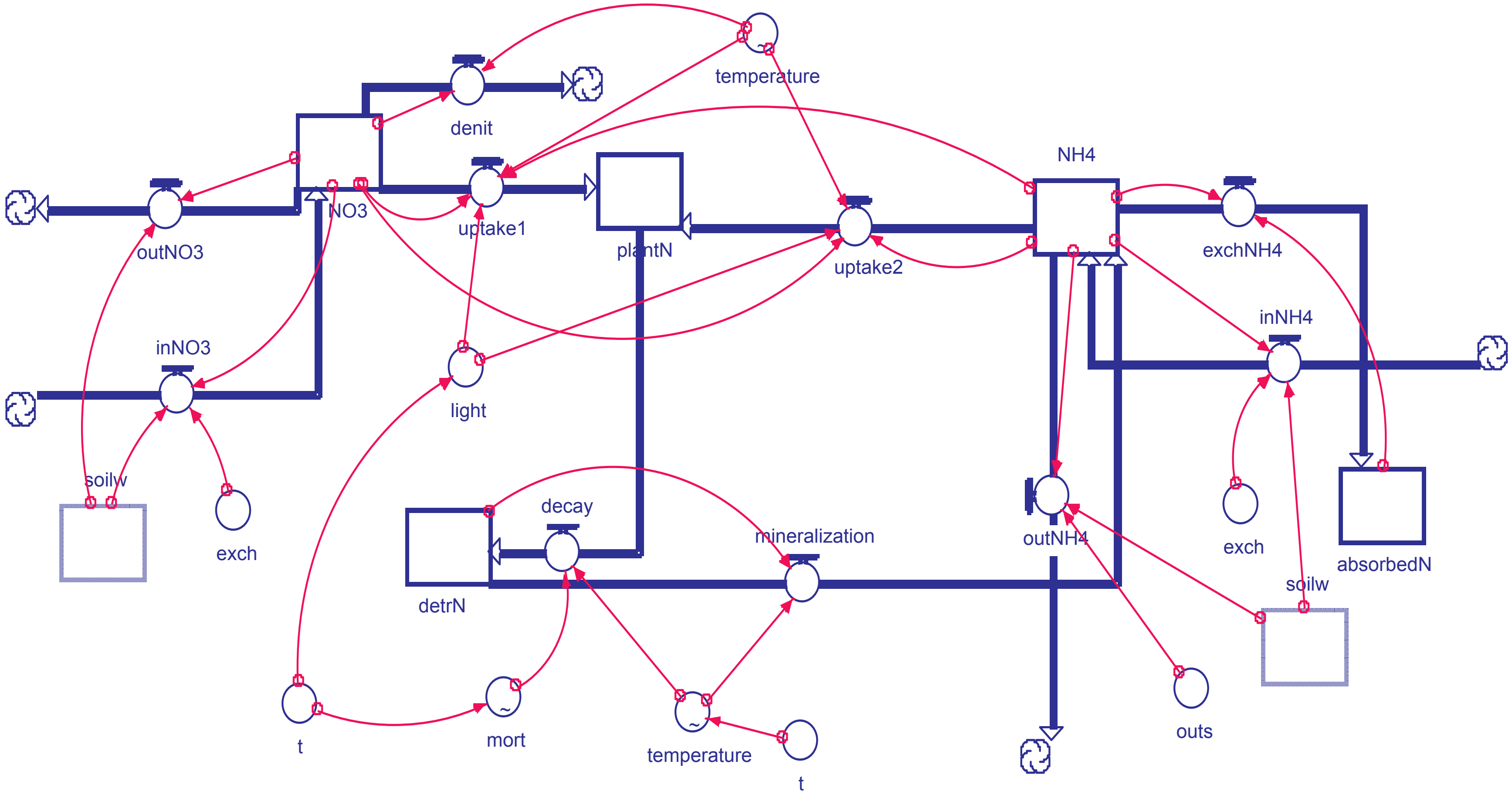


Figure 3.2. Biology sub-division of the conceptual diagram of the wetland model (denit: denitrification; exch: exchange; detr: detritus; t: time; mort: mortality rate; soilw: soil water) (Jorgensen, 1994)

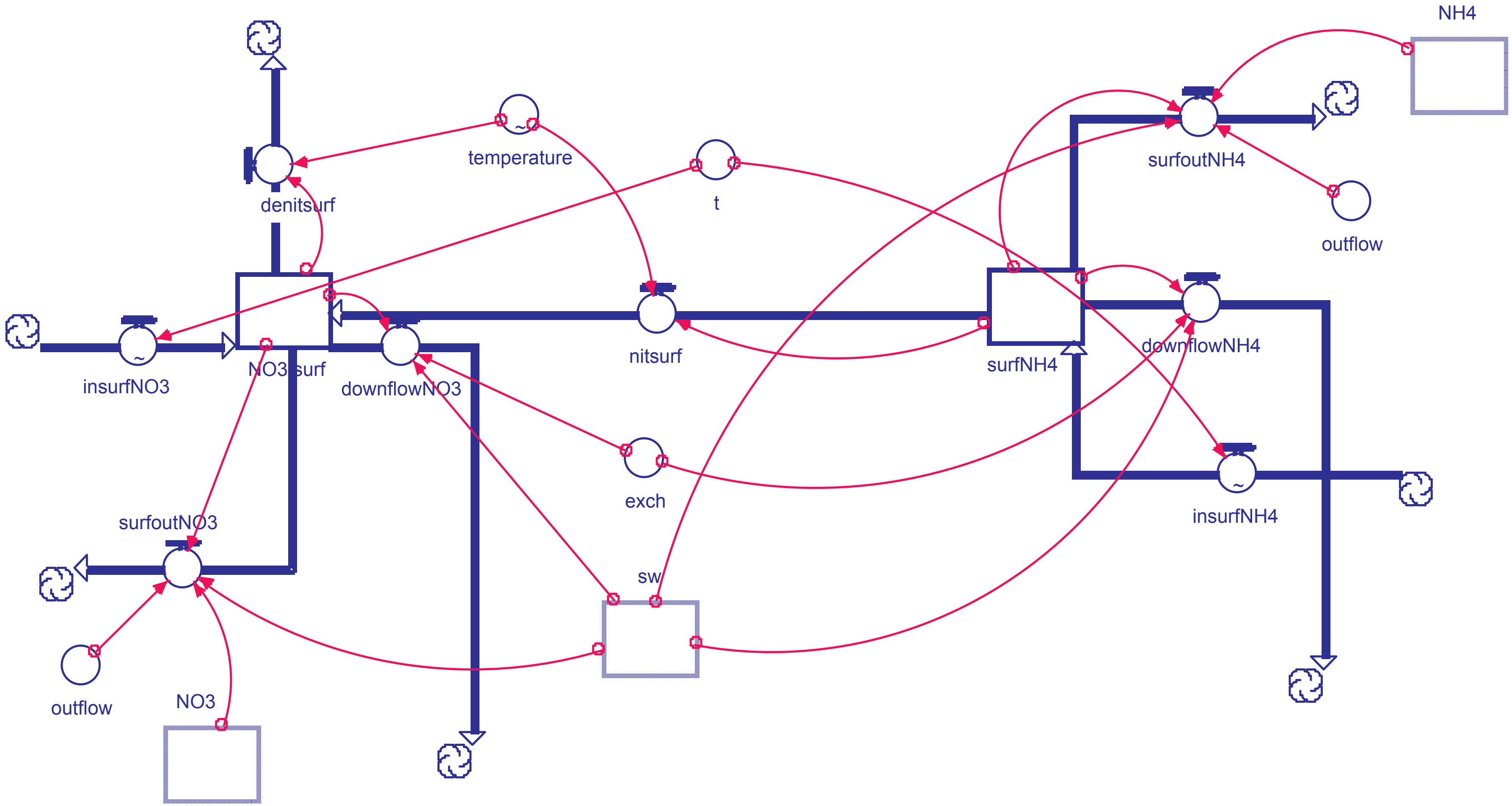


Figure 3.2. (Cont.) Biology sub-division of the conceptual diagram of the wetland model (insurfNO3: surface NO3 input; insurfNH4: surface NH4 input; nitsurf: surface nitrification; denitsurf: surface denitrification) (Jorgensen, 1994)

3.2.2 The model components

The components of the wetland model which was developed for Lake Mogan reed beds were given in the Table 3.1;

Table 3.1. The model components

The climatic forcing functions	- Precipitation - Evaporation - Temperature - solar irradiance
The site specific forcing functions	- nitrate and ammonium concentrations in the inflow water - flow rate
Hydrological state variables	- the surface layer (where the nitrification takes place – part 2) - the active soil zone (where a pronounced denitrification and accumulation take place – part 1)
Nitrogen state variables	- nitrate and ammonium in the surface layer - and nitrate, ammonium, detritus-N, plant-N and absorbed nitrogen in the active soil layer.

3.2.3. Main nitrogen processes included in the model

Major pathways for nitrogen removal in the model included mineralization of organic N, ammonia volatilization, assimilation into plant biomass, absorption of ammonium onto the substrate and nitrification followed by denitrification. Of these, nitrification and subsequent denitrification at the aerobic-anaerobic interface of the wetland substrate was the primary means of nitrogen removal.

Table 3.2. Spectrum of hydraulic conductivity for selected soil (Jorgensen et al., 1994)

Type of soil	**Hydraulic conductivity (permeability of soil)**	
	(m/day)	(m/month)
Clay	0.0005	0.015
Sand	50	1500
Sandy soil	10	300
Medium humic soil	1-5	30-150
Compact peat	0.01-0.05	0.3-1.5*

* taken as the reference for the model runs

The interval of hydraulic conductivity used in the model was decided based on Table 3.2 and compact peat was taken as the representative of the wetland soil of Lake Mogan reed beds. In this context, minimum, maximum and average values of compact peat hydraulic conductivity interval were used in the model calibration and validation as it is given below;

<u>3 Model-scenarios:</u>

Scenario-1. High hydraulic conductivity = 1.2-1.3 m/month

Scenario-2. Average hydraulic conductivity = 0.8-0.9 m/month

Scenario-3. Low hydraulic conductivity = 0.4-0.5 m/month

3.3. GIS Analyses

Geographical Information Systems (GIS) serve two functions in this study: (a) creation of accurate maps and databases based on aerial photographs, (b) Aerial estimations of land-uses (reed beds, agriculture, settlement) in the near catchment of Lake Mogan (b) overlay of the maps to detect changes in the land-uses over time.

Aerial photos of the region taken in 1978, 1991 and 1999 were used for analyses. Following steps were carried out;

- Geographical registration of all the aerial photos,
- Digitizing all the land features in the photos,
- Performing overlay analysis and geo-processing for detecting the changes in the land uses throughout the years

3.3.1. Registration (Georeferencing)

As the first step in the georeferencing of all the available resources, the topological map of the study area was registered with the given control points on the map. Then, the aerial photos that was used in the analysis were georeferenced using this topological map. All these processes have been performed in ESRI’s ArcGIS 8.1 (Arc Map 8.1) Environment.

3.3.2. Digitizing

After georefencing the aerial photos taken in 1978, 1991 and 1999 of the study area, all the distinctive features on the photos were classified into 6 landuse classes which include agriculture, lake, wetland, settlement, meadow and roads and were digitized to corresponding shape files. Also, the same coordinate and projection system (UTM WGS_European_1984 - Zone 36) was assigned to all the shapefiles generated.

3.3.3. Overlay and geoprocessing

After the digitization of the pictures taken in 1978, 1991 and 1999, all the shape files were overlaid two by two. Then, using the intersection function of geoprocessing wizard, intersection areas of the landuse shapefiles were found and saved as new shape files.

3.3.4. Determining the potential risk areas for non-point N input to Lake Mogan

Following steps were taken for this analysis:

Step1: ArcGIS / ArcMap / Spatial Analyst

Spatial Analyst / Options : working directory and cell size chosen

Spatial Analyst / Slope

Reclassify / Slope: Clasify 10 classes with quantile method

Reclassify the slope layer to steep slopes are given higher values (indicating higher risk)

Spatial Analyst / Distance / Straight Line:

Distances to rivers and lake layers are classified in to 1 to 10 values with equal interval method, giving a 10 value to areas closest to the layers.

Step 2: Weighting and combining the datasets

Slope 40%

Distance to rivers	30%
Distance to lakes	30%

Spatial Analyst / Raster Calculator

(0.4* Slope) + (0.3* Distance to rivers) + (0.3* Distance to lakes)

Result of the raster calculation

Evaluation Grid = (0.4 * slope factor) + (0.3 * distance to river) + (0.3 * distance to the lake)

CHAPTER 4

RESULTS

4.1. DIN input vs output estimations

It is observed that there was a significant difference in both NO_3-N surface loads (F:2.87, $p<0.05$) and NH_4-N surface loads (F:2.52, $p<0.05$) in the study years (Table 4.1.1). Specifically, Tukey's HSD test revealed that there were significant difference between NO_3-N surface inputs of 1997 and 1998; 1998 and 2000 and finally 1998 and 2001 (Table 4.1.1). Outputs of both NO_3-N and NH_4-N did not show a statistically significant difference during the period of 1997 to 2001. Water level and water temperature, were also tested for the same period because of their possible effect on the dynamics of nitrogen retention in Lake Mogan reed beds. While a strong difference was observed for water level (F:5.15, $p<0.005$) in the study years, the same was not true for temperature (F:0.51, $p>0.05$) (Table 4.1.1). The water levels recorded in 1998, 1999 and 2002 were significantly different from water level of 2001.

Table 4.1.1 Summary for annual cumulative surface N-loads and outputs, water level and temperature for DIN-retention in Lake Mogan reed beds. Significance of differences was tested by one way ANOVA for the study years and then which years are different from each other were found using Tukey's HSD test.

		1997	1998	1999	2000	2001	2002	ANOVA	Tukey's HSD Test
Input	NO_3-N (g/m^2*yr^1)	6.94	25.71	13.69	6.69	4.64	13.85	F:2.87, p:0.02	98-01 (p<0.05) 97-98; 98-00 (p<0.1)
	NH_4-N (g/m^2*yr^1)	0.59	1.40	1.57	2.14	1.07	3.07	F:2.52, p:0.04	97-02 (p<0.05)
	DIN (g/m^2*yr^1)	7.54	27.11	15.26	8.84	5.71	16.92	F:2.96, p:0.02	98-01 (p<0.05) 97-98 (p<0.1)
Output	NO_3-N (g/m^2*yr^1)	2.15	1.45	1.58	1.18	2.50	2.46	F:1.81, p:0.126	ns
	NH_4-N (g/m^2*yr^1)	2.11	2.28	1.20	1.47	3.07	1.41	F:0.51, p:0.768	ns
	DIN (g/m^2*yr^1)	4.26	3.73	2.79	2.64	5.57	3.87	F:0.82, p:0.54	ns
	Water level (m)	972.8± 0.24	972.9± 0.20	972.8± 0.17	972.7± 0.42	972.4± 0.30	972.9± 0.31	F:5.15, p:0.0005	98-01 (p<0.05); 99-01(p<0.05); 01-02 (p<0.05)
	Water temperature (^oC)	14.1±7.77	16.9±7.79	17.3±7.74	18.0±5.9	17.9±7.02	15.3±6.3	F:0.49, p:0.78	ns

NO_3-N and NH_4-N showed different trends throughout the study years. Surface inflow of NO_3-N was considerably higher in 1998, 1999 and 2002 while, surface inflow of NH_4-N showed variability (Figure 4.1.1 and Figure 4.1.2). For outputs, NO_3-N stayed almost constant, while NH_4-N showed a distinctive increase in the study period (Figure 4.1.1 and Figure 4.1.2).

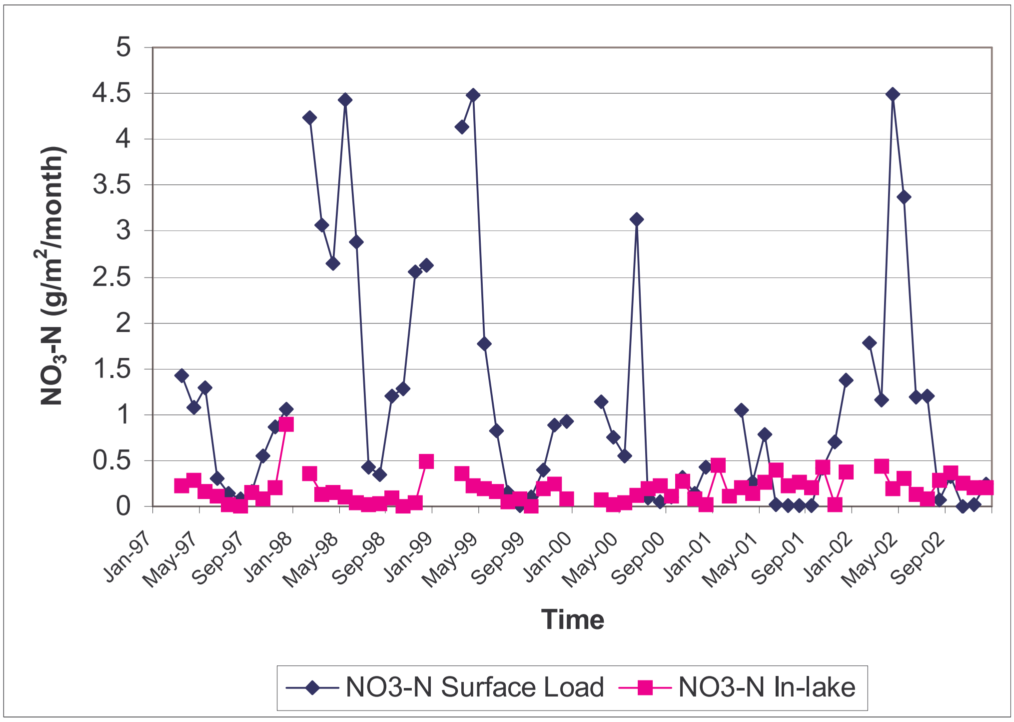


Figure 4.1.1. Changes in NO_3-N surface load to Lake Mogan reed beds and NO_3-N in the lake throughout the study years.

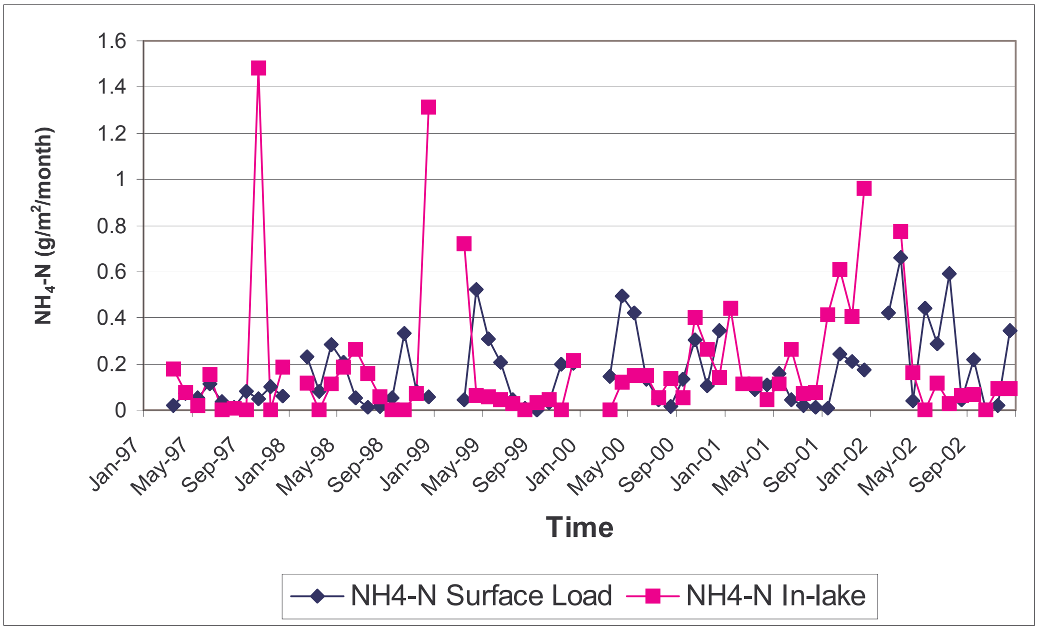


Figure 4.1.2. Changes in NH_4-N surface load to Lake Mogan reed beds and NH_4-N in the lake throughout the study years

4.1.1. Nitrogen retention in Lake Mogan reed beds:

It was observed that retention of NO_3-N in Lake Mogan reed beds was higher when the inflow of NO_3-N to the reed beds was higher (Table 4.1.2). However, this was not the case in retention of NH_4-N in Lake Mogan reed beds. Actually, retention of NH_4-N in Lake Mogan reed beds showed a great variability and significant amount of NH_4-N export occurred from the reed beds to the lake for some years (Table 4.1.3). Tukey's HSD test revealed that there was significant difference between the input and output of NO_3-N in 1998 and 1999 whereas, no significant difference was observed for the years 1997, 2000 and 2001 (Table 4.1.2).

Table 4.1.2. Comparison of cumulative NO_3-N load and NO_3-N out for the study years. Significance of differences was tested using Tukey's HSD Test. (Monthly values of NO_3-N load and NO_3-N out were used in the calculations of statistical analysis).

Years	NO_3-N load ($g/m^2/yr$)	NO_3-N out ($g/m^2/yr$)	NO_3-N retention ($g/m^2/yr$)	NO_3-N retention rate (%)	Tukey's HSD Test
1997	6.94	2.15	4.79	69.0	n.s.
1998	25.71	1.45	24.26	94.4	0.000* ($p<0.05$)
1999	13.69	1.58	12.11	88.4	0.021* ($p<0.05$)
2000	6.69	1.18	5.52	82.4	n.s.
2001	4.64	2.50	2.13	46.0	n.s.
2002	13.85	2.46	11.39	82.2	($p<0.05$)

Table 4.1.3. Comparison of cumulative NH_4-N load and NH_4-N out for the study years. Significance of differences was tested using Tukey's HSD Test. (Monthly values of NH4 load and NH4 out were used in the calculations of statistical analysis).

Years	NH_4-N load ($g/m^2/yr$)	NH_4-N out ($g/m^2/yr$)	NH_4-N retention ($g/m^2/yr$)	NH_4-N retention rate (%)	Tukey's HSD Test
1997	0.592	2.109	-1.517	-256.3	n.s.
1998	1.404	2.282	-0.878	-62.5	n.s.
1999	1.569	1.204	0.365	23.3	n.s.
2000	2.144	1.466	0.678	31.6	n.s.
2001	1.072	3.065	-1.993	-185.9	n.s.
2002	3.074	1.41	1.66	54.1	n.s.

A decreasing trend was observed for NO_3-N retention from 1998 to 2001 in Lake Mogan reed beds. The decrease was more distinctive in terms of the amount retained than the percentage (Figure 4.1.3). In contrast to the decline in NO_3-N load (Table 4.1.2), the outflow of NO_3-N from the reed beds did not show much variability, which in turn resulted in the decline of NO_3-N retention rate.

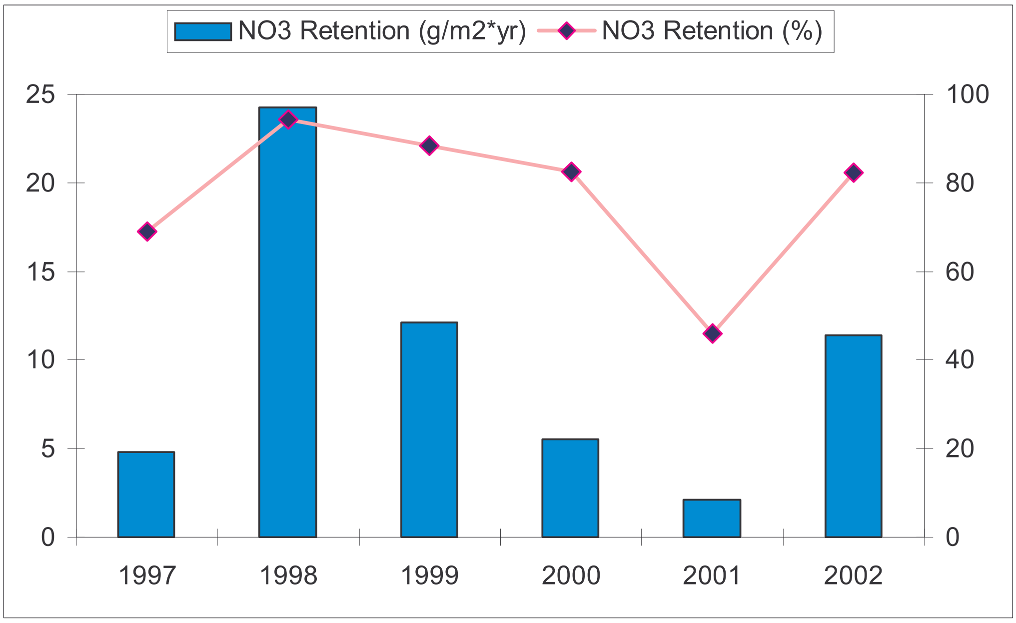


Figure 4.1.3. Cumulative retention of NO_3-N in Lake Mogan reed beds

On the other hand, NH_4-N retention showed a great variability throughout the study years and, there was NH_4-N release from the reed beds in years 1997, 1998 and 2001 (Figure 4.1.4). Tukey's HSD test resulted that no significant difference between NH_4-N in and NH_4-N out for any one of the study years. The DIN retention in Lake Mogan reed beds showed a pattern as observed for the retention trend of NO_3-N (Figure 4.1.3 & Figure 4.1.5).

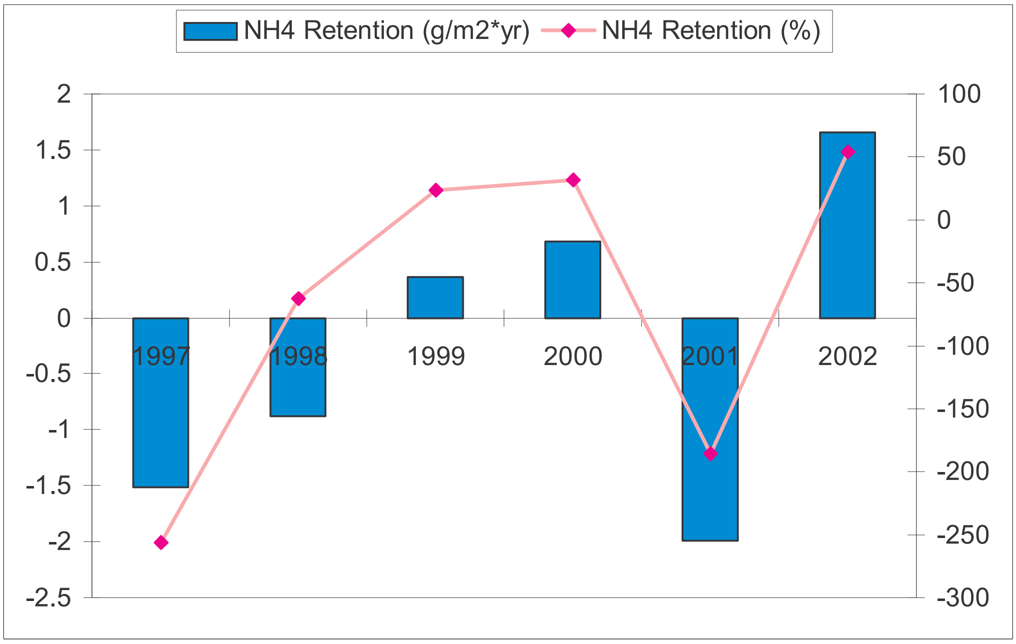


Figure 4.1.4. Cumulative NH_4-N Retention in Lake Mogan reed beds

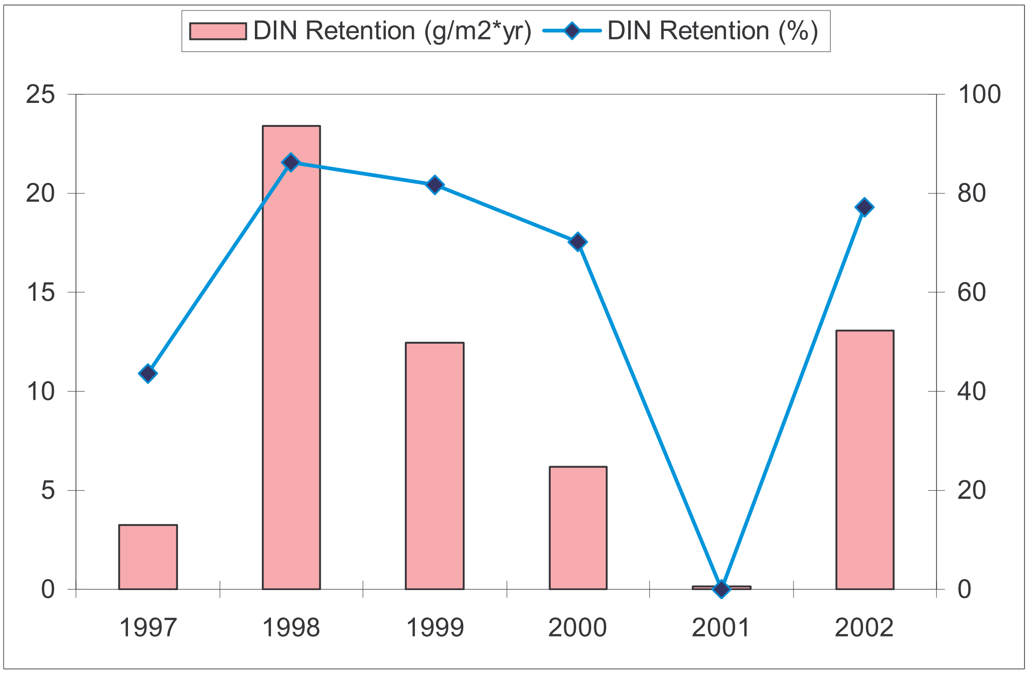


Figure 4.1.5. Cumulative DIN retention in Lake Mogan reed beds

Monthly surface loads, outputs & retentions for NO_3-N, NH4-N and DIN in Lake Mogan reed beds are presented in the following figures (Figure 4.1.6a-c, Figure 4.1.7a-c, Figure 4.1.8a-c). In terms of seasonality, NO_3-N and DIN retentions were higher in spring months when the surface load was also high (Figure 4.1.6c and Figure 4.1.8c). Retention of NO_3-N and DIN was also high in autumn specifically in 1998. In summer, almost no retention or export of NO_3-N and DIN occurred in the study period. Comparing between the years, the NO_3-N surface load was distinctively higher in years 1998, 1999, 2002 in springtime, while only 1998 surface load was distinctively higher in fall months (Figure 4.1.6a). NO_3-N outputs oscillated around the same level for all the study years with a slight increasing trend towards the end of year (Figure 4.1.6b). DIN surface loads and outputs followed same trend with NO_3-N, higher input in spring time for the high load years, 1998, 1999 and 2002 and no difference for the output trends between the years (Figure 4.1.8a and Figure 4.1.8c).

For NH4-N, some amount of retention observed in spring months for most of study years whereas, NH4-N export from the reed beds was considerably high in autumn for some years (Figure 4.1.7c).

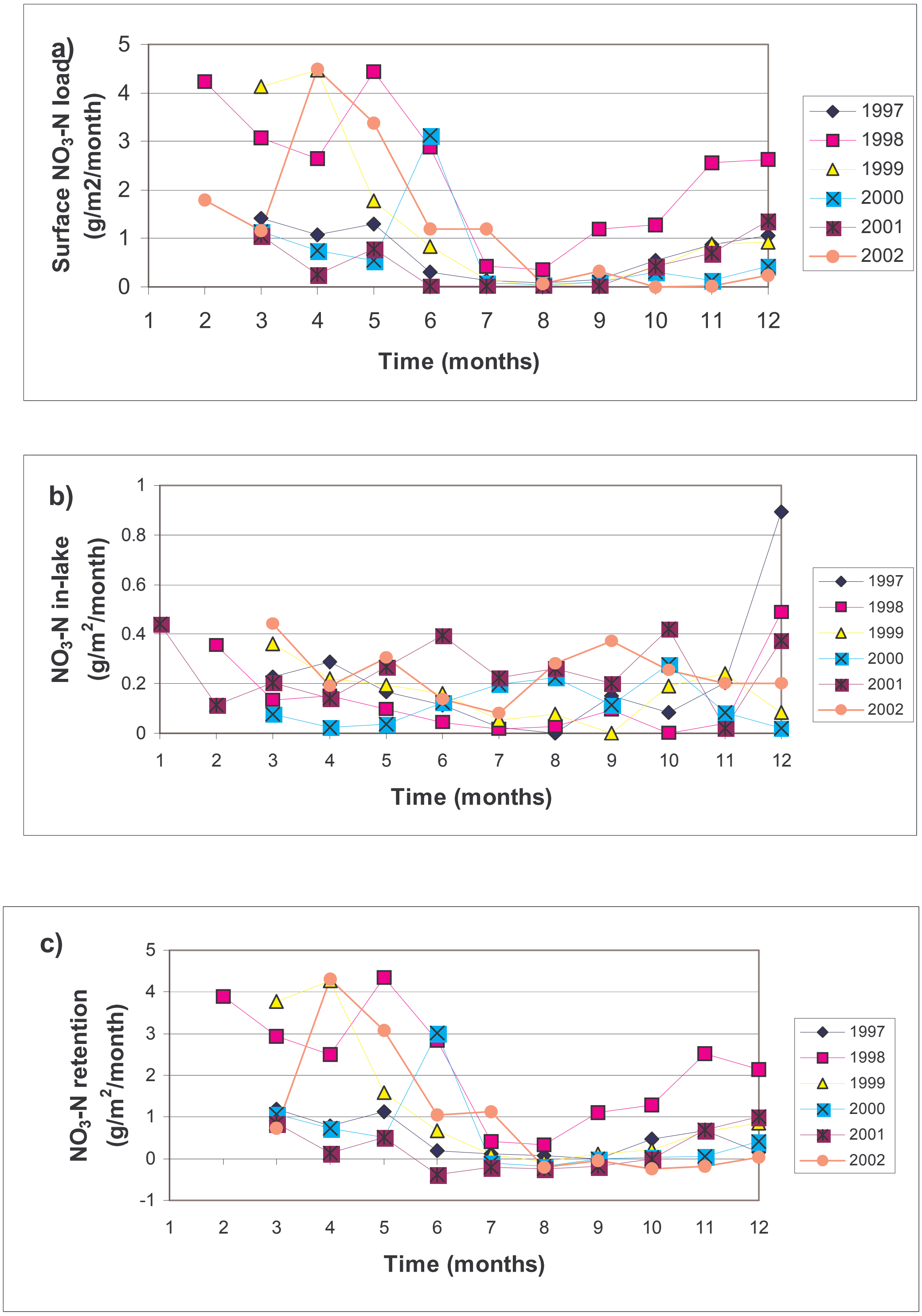


Figure 4.1.6 Yearly changes in a) surface NO_3-N load; b) NO_3-N in lake & c) NO_3-N retention

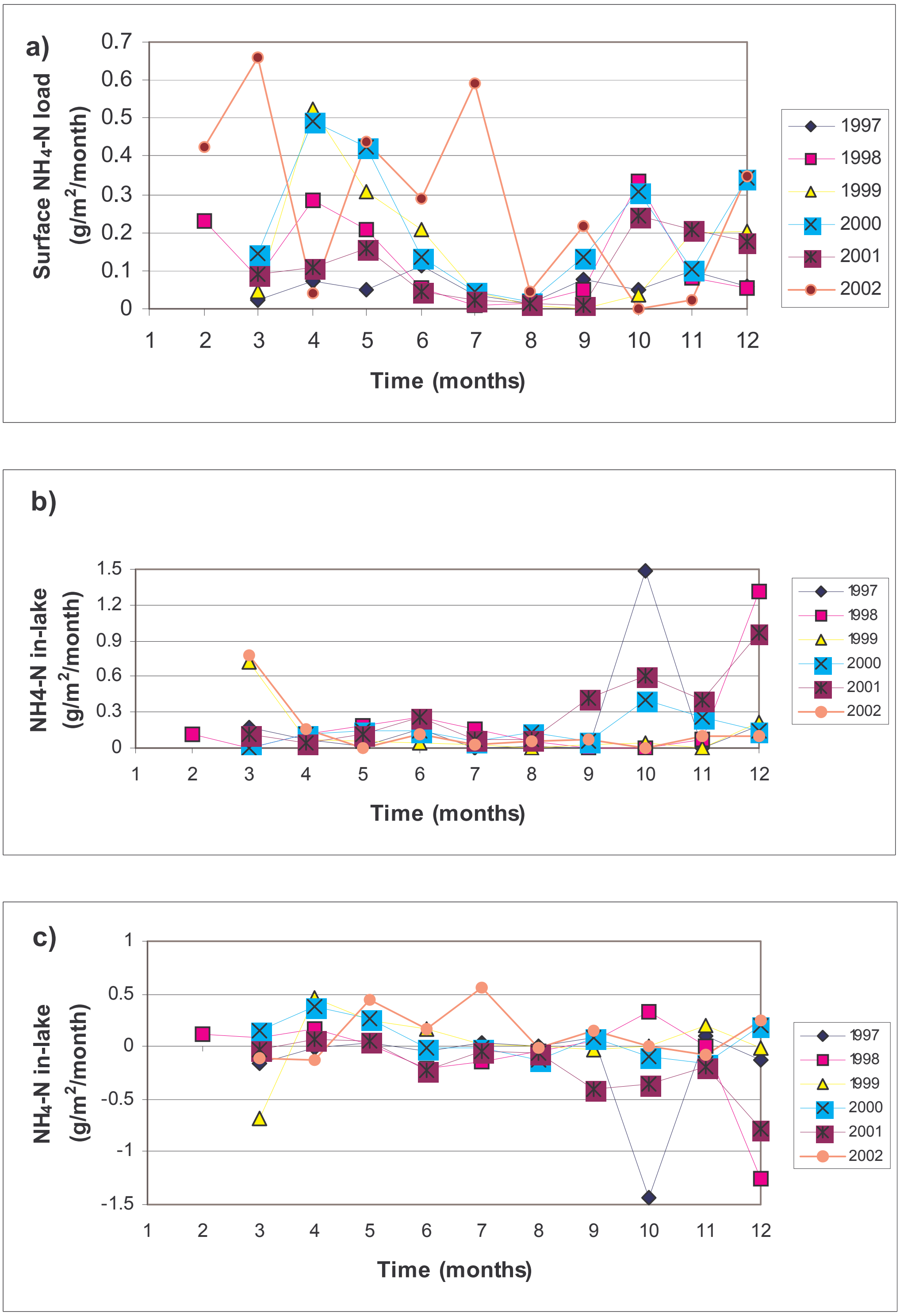
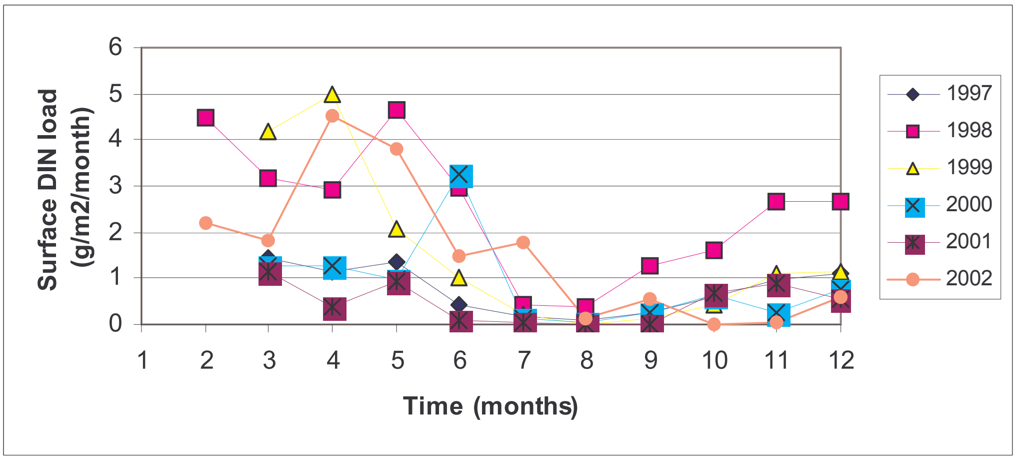
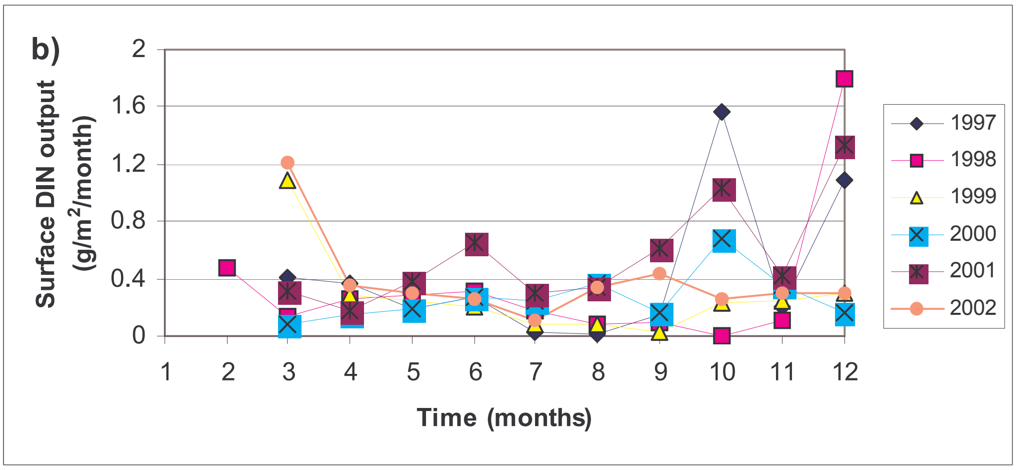


Figure 4.1.7. Yearly changes in a) surface NH_4-N load; b) NH_4-N in lake & c) NH_4-N retention





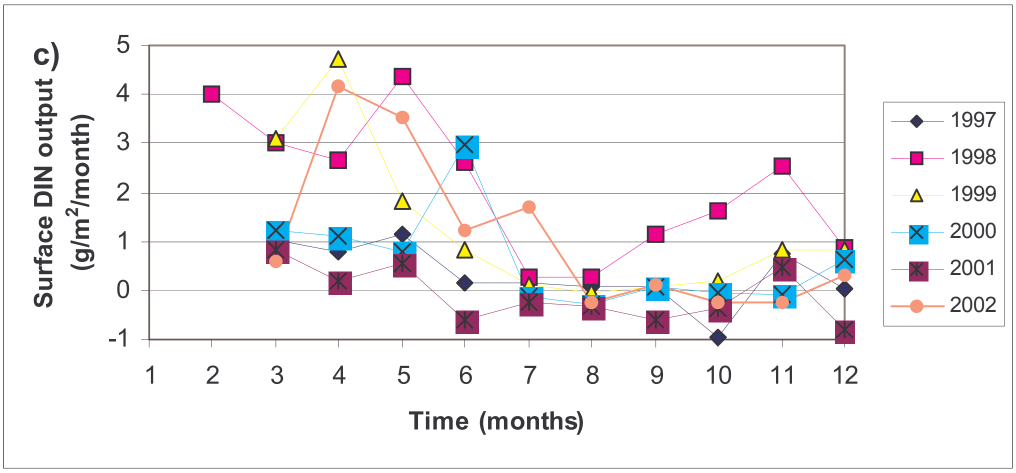


Figure 4.1.8. Yearly changes in a) surface DIN load; b) DIN in lake & c) DIN retention

4.1.2. Effect of NO_3-N load on NO_3-N retention

A significant linear relationship was observed between load and retention of NO_3-N (r^2 = 0.97) and DIN (r^2 = 0.90) (Figure 4.1.9 & Figure 4.1.10).

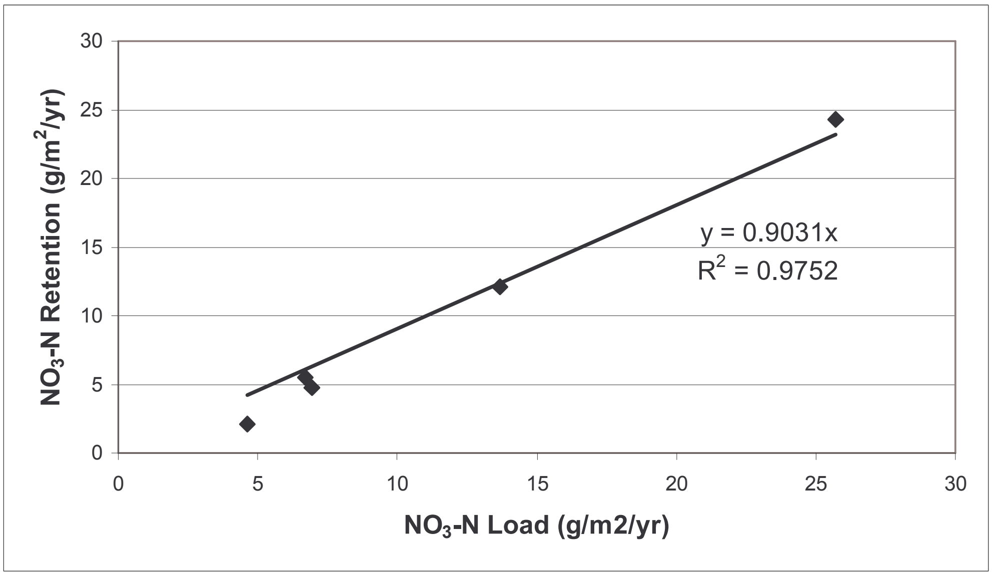


Figure 4.1.9. Relation between NO_3-N surface load and NO_3-N retention for the study years.

4.1.3. Effect of water level on NO_3-N retention

It was observed that the retention of NO_3-N and DIN was higher in the high water level years (Figure 4.1.11 and Figure 4.1.12), Retention rates increased considerable especially when the water level was above 972.40 m for both NO_3-N and DIN.

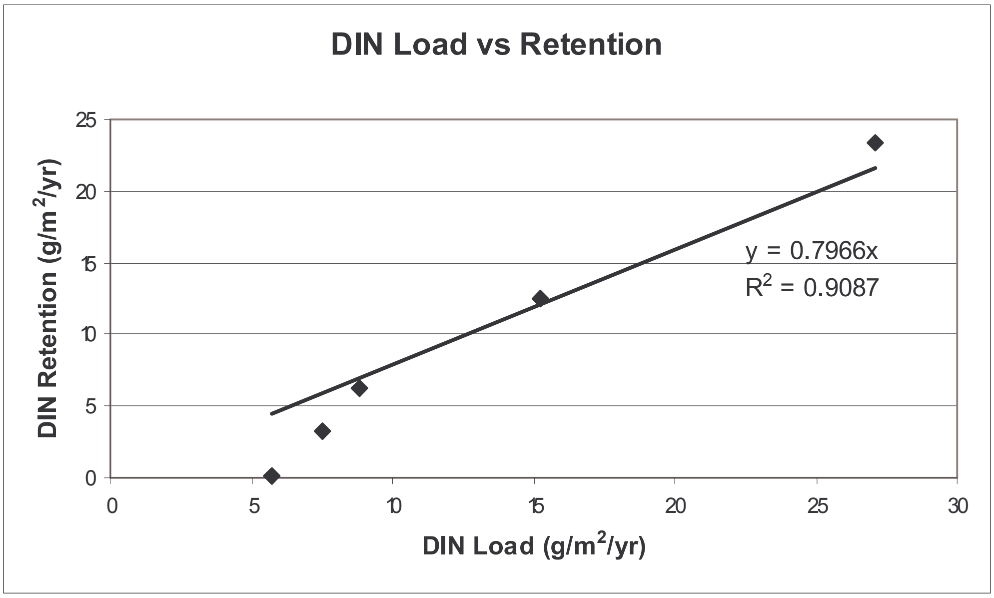


Figure 4.1.10. Relation between DIN surface load and DIN retention for the study years.

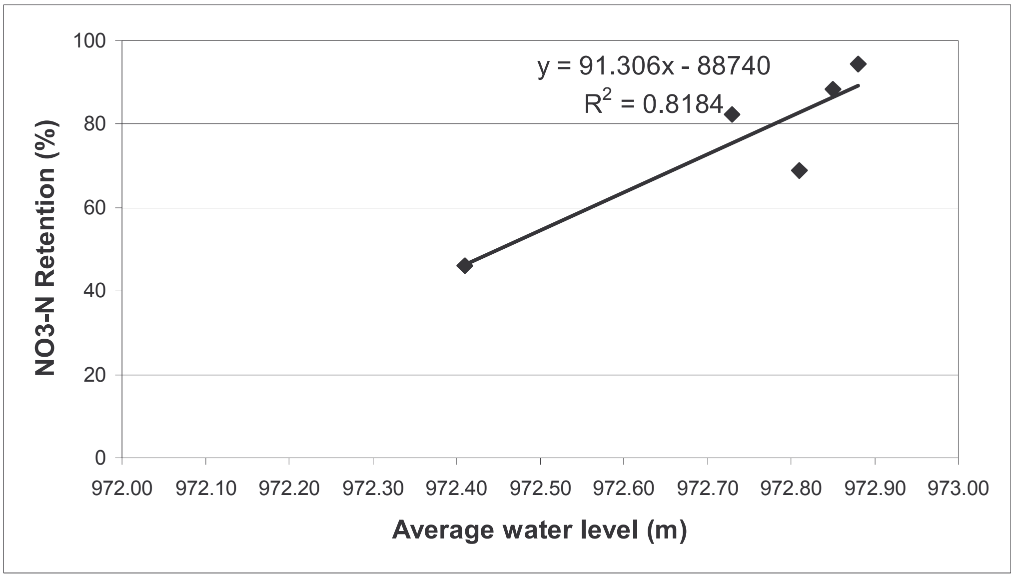


Figure 4.1.11. Relation between average water level of Lake Mogan and NO_3-N retention in the reed beds for the study period

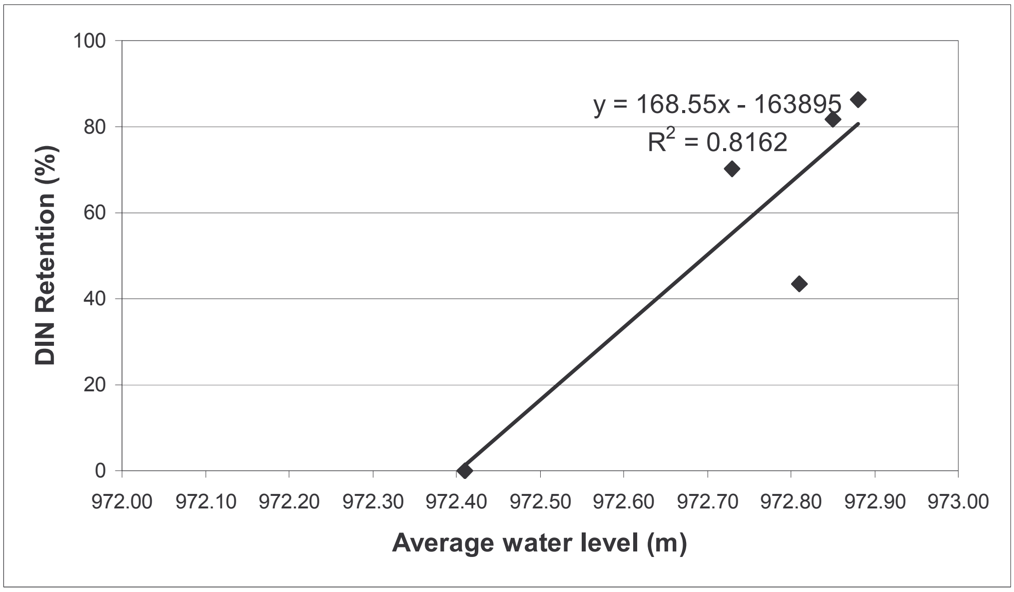


Figure 4.1.12. Relation between average water level of Lake Mogan and DIN retention in the reed beds for the study period

4.1.4. Effect of temperature on NO_3-N retention

The retention of NO_3-N increased with increasing temperature up to the certain value then it leveled off & started to decrease with the further increase in the temperature. Cooler temperature was experienced in relatively wet years with the higher water level, whereas, higher temperature was recorded in drier years with lower water level. As it was recorded for water level, NO_3-N retention rate relationship, an abrupt transition occurs for NO_3-N retention rates in between the temperatures of 17.5 to 19 ^oC (Figure 4.1.13).

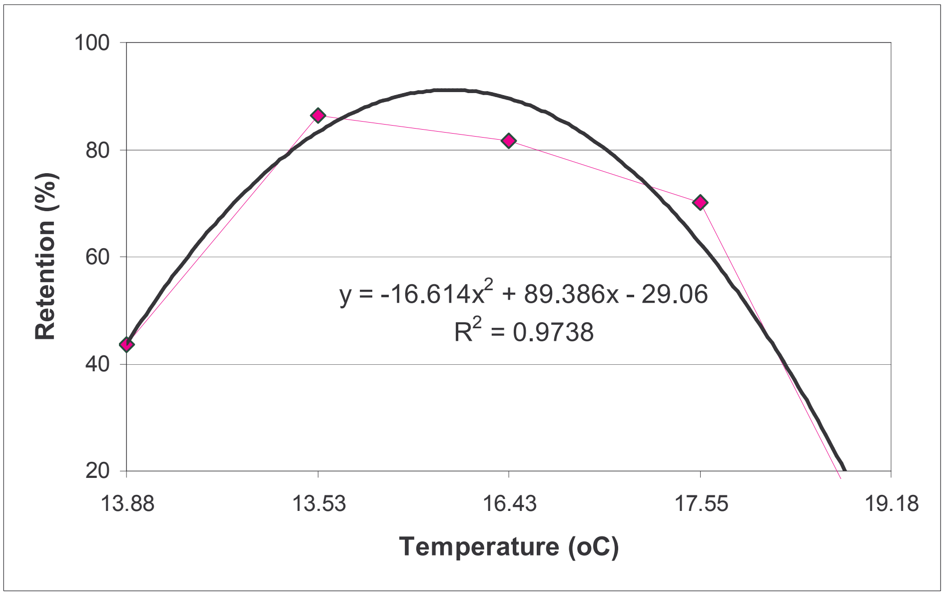


Figure 4.1.13. Relation between average temperature and NO_3-N retention for the study years.

4.2. Modeling

4.2.1. Calibration

a) Uptake rates for nitrate-N and ammonium-N

Data sets of year 1999, 2000 and 2002 were used to calibrate the model parameters. Firstly, uptake rates for nitrate-N and ammonium-N and the mineralization rate were calibrated to give the observed trends in maximum value of plant nitrogen content and also seasonal dynamics of the mentioned parameters (Figure 4.2.1). Although the uptake rate constants for nitrate and ammonium were taken the same, the rates differed since they were dependent to input nutrient concentrations.

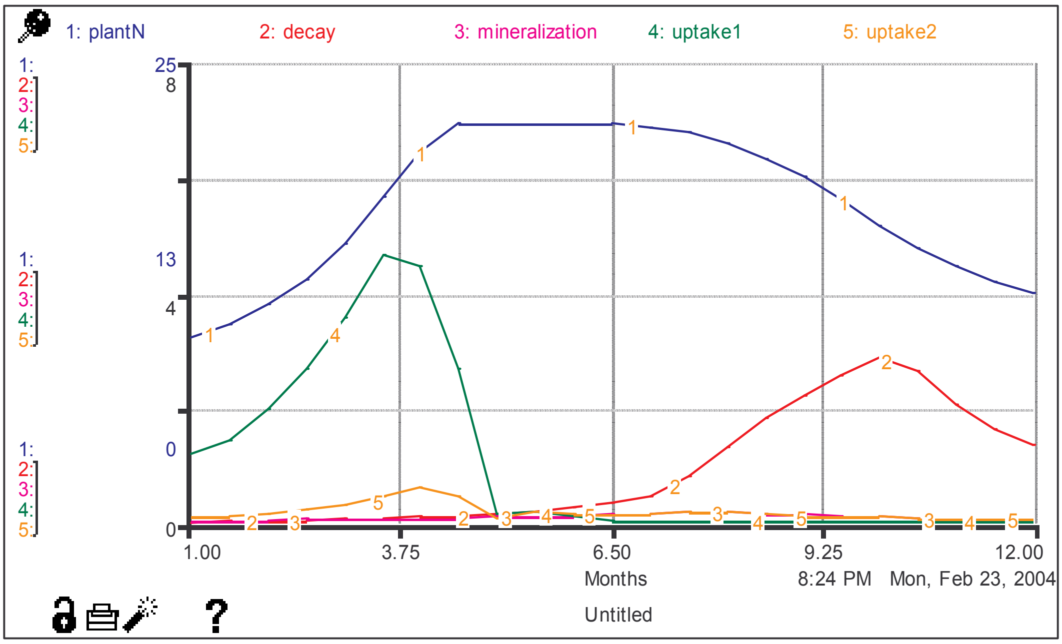


Figure 4.2.1. Calibration for nitrate, ammonium uptake rates, mineralization rate and plant nitrogen content (plantN: plant nitrogen; uptake1: uptake rate for nitrate-N; uptake2: uptake rate for ammonium-N).

After calibration of uptake rates, the model was simulated with different values of hydraulic conductivity to find out best-suited one.

b) Hydraulic conductivity

Similar to calibration of uptake rates, year 1999, 2000 and 2002 data sets were used to calibrate the hydraulic conductivity.

1999:

1999 was a relatively high surface load year for NO_3-N (Table 4.1.1). For 1999, high hydraulic conductivity set of the model best predicted total NO_3-N retention (Table 4.2.1). In terms of seasonal NO_3-N output dynamics, high hydraulic conductivity could also satisfactorily simulated NO_3-N output (Figure 4.2.2).

In contrast to NO_3-N, closest model result to the observed NH_4-N outputs was given by the low hydraulic conductivity (Table 4.2.1). In terms of seasonality of NH_4-N outputs, none of the three high hydraulic conductivity sets could predict could predict the spring pick in the observed NH_4-N output (Figure 4.2.3). Except for spring pick, high hydraulic conductivity best simulated the seasonal NH_4-N output dynamics.

Table 4.2.1. Summary of model simulations and measured values of 1999 for NO_3-N & NH_4-N

	Measured Values	**Model Simulations**		
		High hyd. cond.	Avg. hyd. cond.	Low hyd. cond.
NO_3-N input (g/m^2*year)	12.77	-	-	-
NO_3-N output (g/m^2*year)	1.49	1.45	2.93	5.12
Rate of reduction (%)	88.3	88.6	77.06	59.9
NH_4-N input (g/m^2*year)	1.369	-	-	-
NH_4-N output (g/m^2*year)	1.02	0.205	0.328	0.548
Rate of reduction (%)	25.5	85.03	76.04	59.9

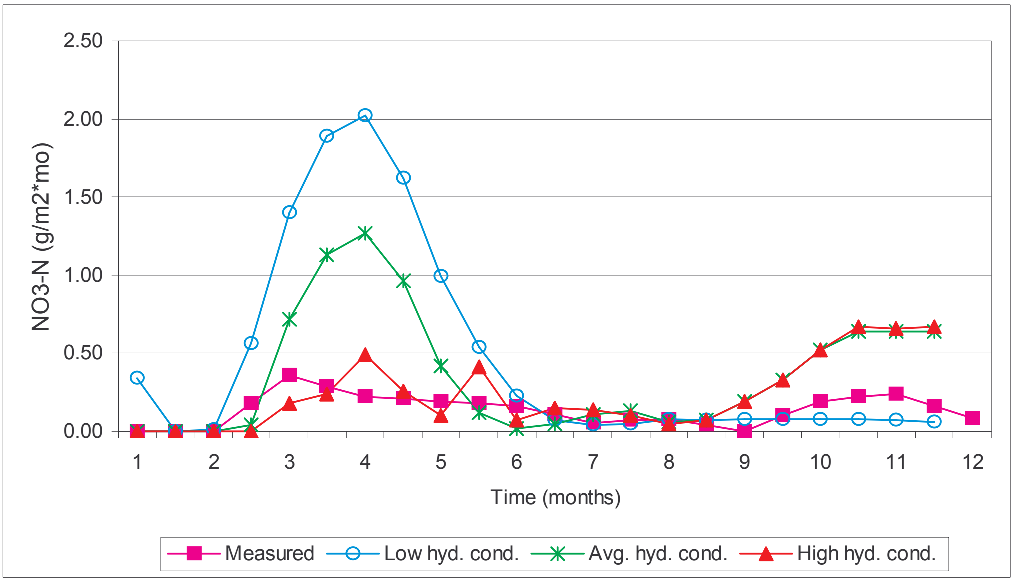


Figure 4.2.2. Model simulation for NO_3-N in 1999

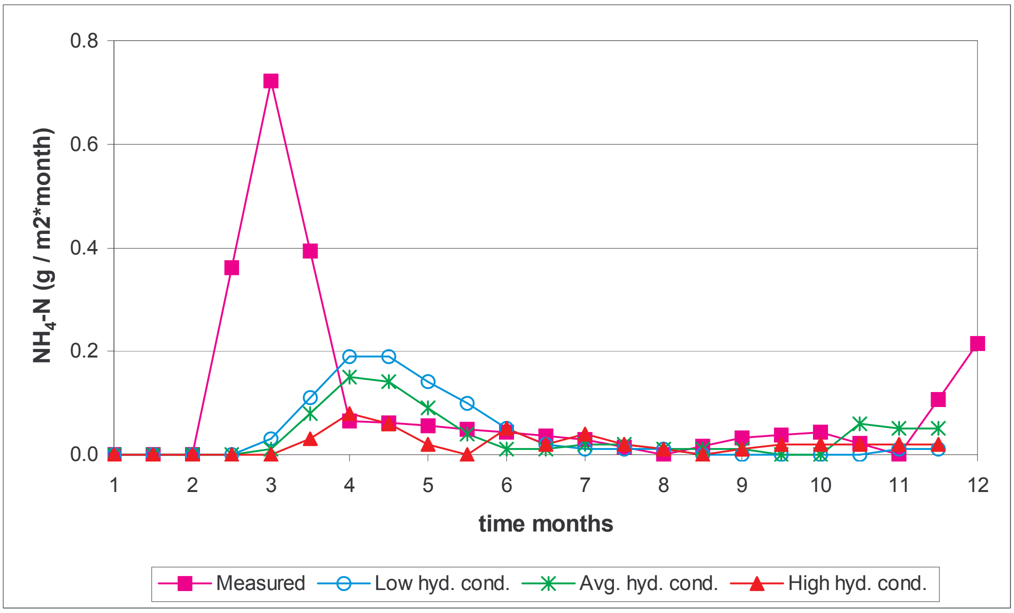


Figure 4.2.3. Model simulation for NH_4-N in 1999

2000:

Afterwards, the model was tested for 2000 data set, which was a relatively low NO_3-N surface load year compared with the 1998, 1999 and 2002 (Table 4.1.1). In this low surface load year, still high hydraulic conductivity set was the closest model result in terms rate of reduction NO_3-N (Table 4.2.2). For NH_4-N, average hydraulic conductivity predicted slightly better than the high hydraulic conductivity for cumulative retention of NH_4-N in 2000 (Table 4.2.2).

In terms of seasonal NO_3-N output dynamics, the three model sets slightly overestimated the NO_3-N outputs in the springtime, though they simulated quite well in overall (Figure 4.2.4). At the same time, high and average hydraulic conductivity model sets simulated almost perfectly the observed seasonal dynamics of NH_4-N output (Figure 4.2.5).

Table 4.2.2. Summary of model simulations and measured values of 2000 for NO_3-N & NH_4-N

	Measured Values	**Model Simulations**		
		High hyd. cond.	Avg. hyd. cond.	Low hyd. cond.
NO_3-N input (g/m2*year)	6.3	-	-	-
NO_3-N output (g/m^2*year)	1.15	1.61	2.47	2.49
Rate of reduction (%)	81.8	74.4	60.8	60.5
NH_4-N input (g/m^2*year)	1.831	-	-	-
NH_4-N output (g/m^2*year)	1.311	0.94	1.04	0.74
Rate of reduction (%)	28.4	48.7	43.2	59.6

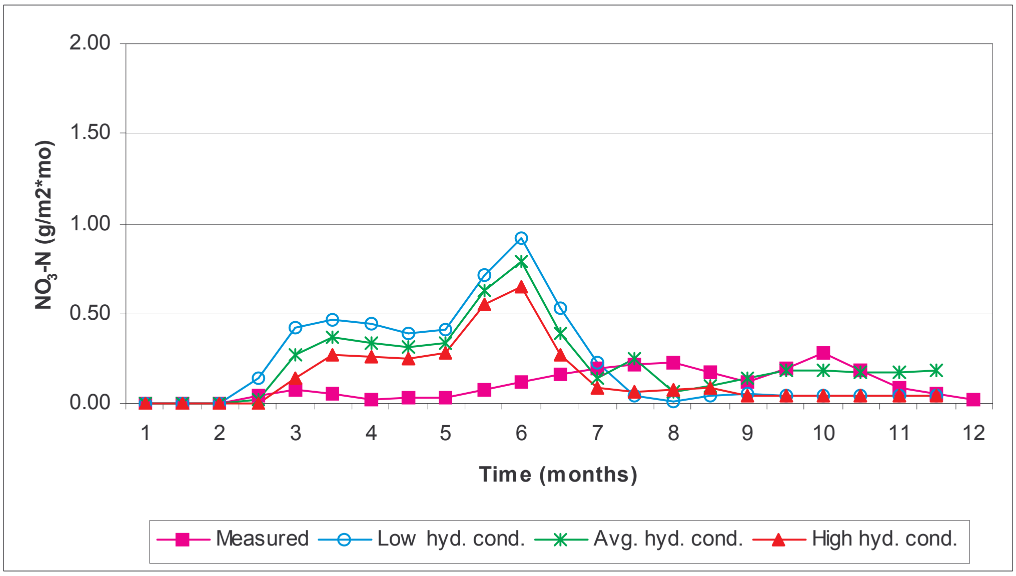


Figure 4.2.4. Model simulation for NO_3-N in 2000

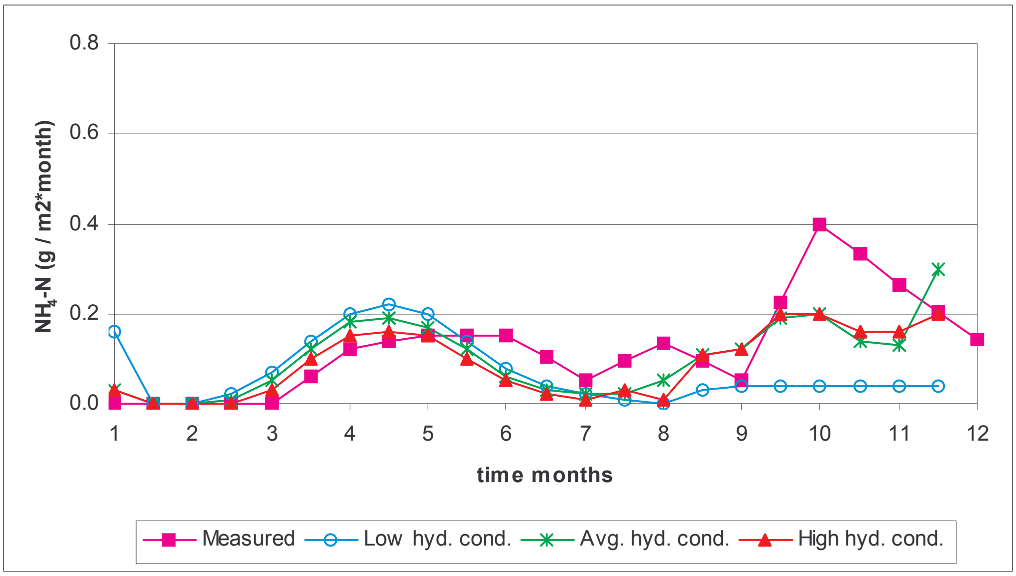


Figure 4.2.5. Model simulation for NH_4-N in 2000

2002:

For calibration, the model was finally tested for 2002 data set, which was another relatively high NO_3-N surface load year (Table 4.1.1). In this high surface load year, average hydraulic conductivity set was the closest model result to the observed cumulative retention of NO_3-N (Table 4.2.3). For NH_4-N, low hydraulic conductivity model set best predicted the cumulative retention of NH_4-N in 2000 (Table 4.2.3).

However, in terms of seasonal NO_3-N output dynamics, high hydraulic conductivity best simulated the NO_3-N output (Figure 4.2.6). Similarly, except for spring pick in NH_4-N output, high hydraulic conductivity model set simulated satisfactorily the observed seasonal dynamics of NH_4-N output (Figure 4.2.7).

Table 4.2.3. Summary of model simulations and measured values of 2002 for NO_3-N & NH_4-N

	Measured Values	**Model Simulations**		
		High hyd. cond.	Avg. hyd. cond.	Low hyd. cond.
NO_3-N input (g/m2*year)	13.19	-	-	-
NO_3-N output (g/m^2*year)	2.26	0.83	2.34	4.67
Rate of reduction (%)	82.9	93.7	82.3	64.6
NH_4-N input (g/m^2*year)	2.707	-	-	-
NH_4-N output (g/m^2*year)	1.324	0.263	0.668	1.265
Rate of reduction (%)	51.1	90.3	75.3	53.3

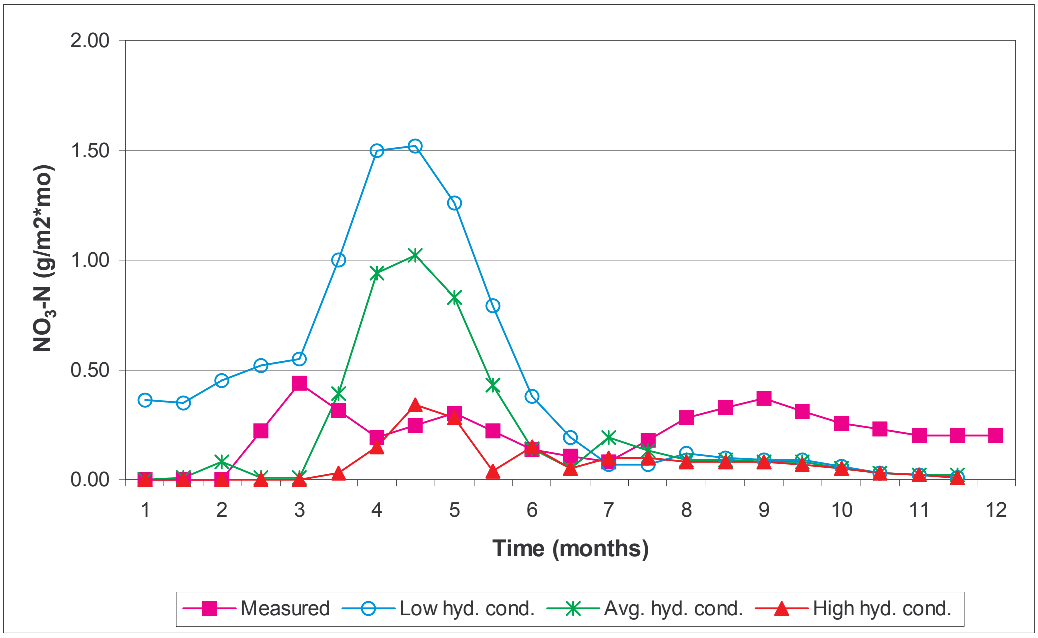


Figure 4.2.6. Model simulation for NO_3-N in 2002

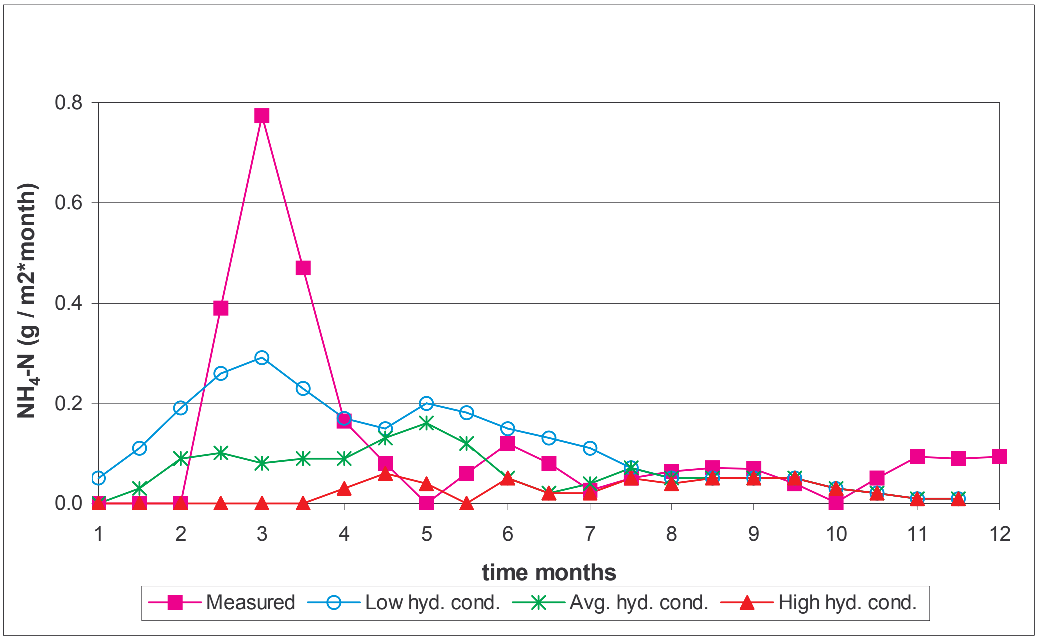


Figure 4.2.7. Model simulation for NH_4-N in 2002

In overall evaluation of model results for the study years used for calibration, high hydraulic conductivity was the best model set simulating observed NO_3-N output from Lake Mogan reed beds. Whereas for NH_4-N output, low hydraulic conductivity model set generally better predicted cumulative NH_4-N output but still high hydraulic conductivity satisfactorily simulated seasonal output dynamics. As a result, high hydraulic conductivity model set was selected for further validation of Lake Mogan Wetland Model.

4.2.2. Validation

After selection of high hydraulic conductivity, data sets of year 1998 and 2001 were used to validate the model parameters.

1998

For validation, the model was firstly tested for 1998 data set, which was highest surface NO_3-N load year among the study years (Table 4.1.1). In this high surface load year, the chosen high hydraulic conductivity model set successfully modeled the observed cumulative retention of NO_3-N (Table 4.2.4). However, this was not the case in terms of modeling the cumulative retention NH_4-N (Table 4.2.4).

In terms of seasonal NO_3-N output dynamics, high hydraulic conductivity still satisfactorily simulated the NO_3-N output dynamics except for slight bias in the timing of spring pick in the output (Figure 4.2.8). For NH_4-N output dynamics, the model generally followed the observed trends but it was insufficient for simulating the dramatic increase of NH_4-N output in the autumn (Figure 4.2.9).

Table 4.2.4. Summary of model results and measured values of 1998 for NO_3-N & NH_4-N

	Measured Values	**Model Simulation** High hyd. cond.
NO_3-N input (g/m^2*year)	22.04	-
NO_3-N output (g/m^2*year)	0.93	1.21
Rate of reduction (%)	95.8	94.5
NH_4-N input (g/m^2*year)	1.289	-
NH_4-N output (g/m^2*year)	1.09	0.32
Rate of reduction (%)	15.1	75.3

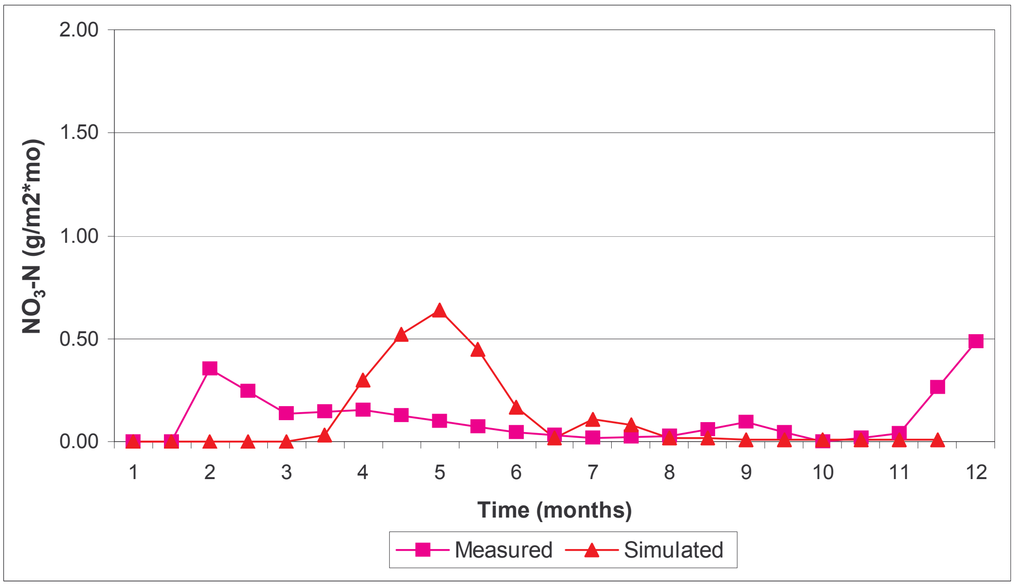


Figure 4.2.8. Model simulation for NO_3-N in 1998

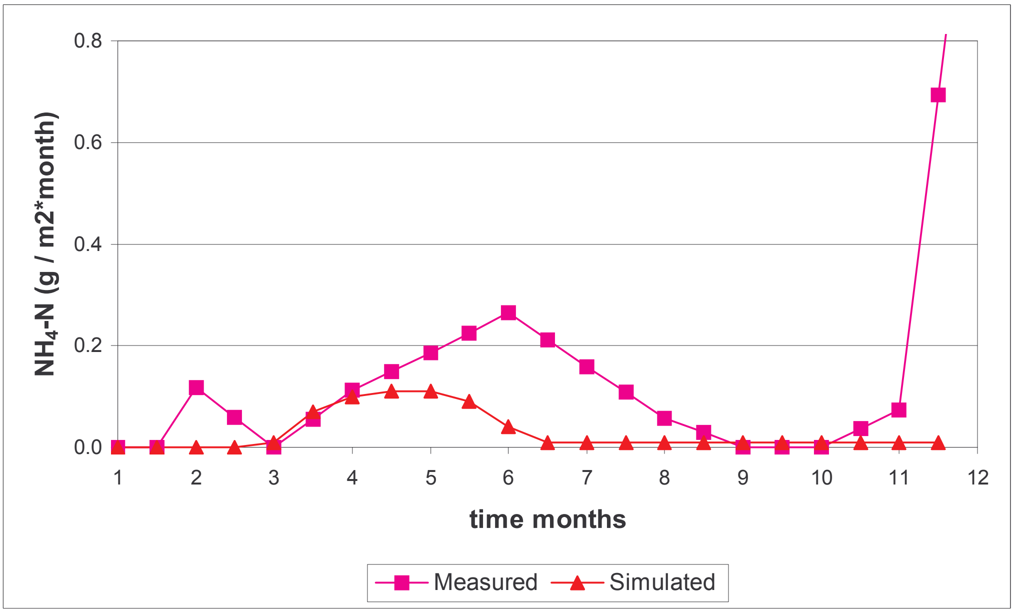


Figure 4.2.9 Model simulation for NH_4-N in 1998

2001:

To be able to validate the model in different and more extreme conditions, the model was finally tested for 2001 data set, which was lowest surface NO_3-N load year among the study years (Table 4.1.1). Also in this low surface load year, the chosen high hydraulic conductivity model set successfully modeled the observed cumulative retention of NO_3-N with in the limits of ±10% (Table 4.2.5). However, again, the model result was not good enough in terms of modeling the cumulative retention NH_4-N (Table 4.2.5).

In terms of seasonal NO_3-N output dynamics, high hydraulic conductivity satisfactorily simulated the NO_3-N output dynamics (Figure

4.2.10). For NH_4-N output dynamics, though the model generally followed the observed trends NH_4-N output dynamics, the model outputs were distinctively lower than the measured outputs throughout the year (Figure 4.2.11).

Table 4.2.5. Summary of model results and measured values of 2001 for NO_3-N & NH_4-N

	Measured Values	**Model Simulation**
		High hyd. cond.
NO_3-N input (g/m^2*year)	3.35	-
NO_3-N output (g/m^2*year)	2.20	2.51
Rate of reduction (%)	34.3	25.1
NH_4-N input (g/m^2*year)	0.896	-
NH_4-N output (g/m^2*year)	2.176	0.625
Rate of reduction (%)	-142.8	30.2

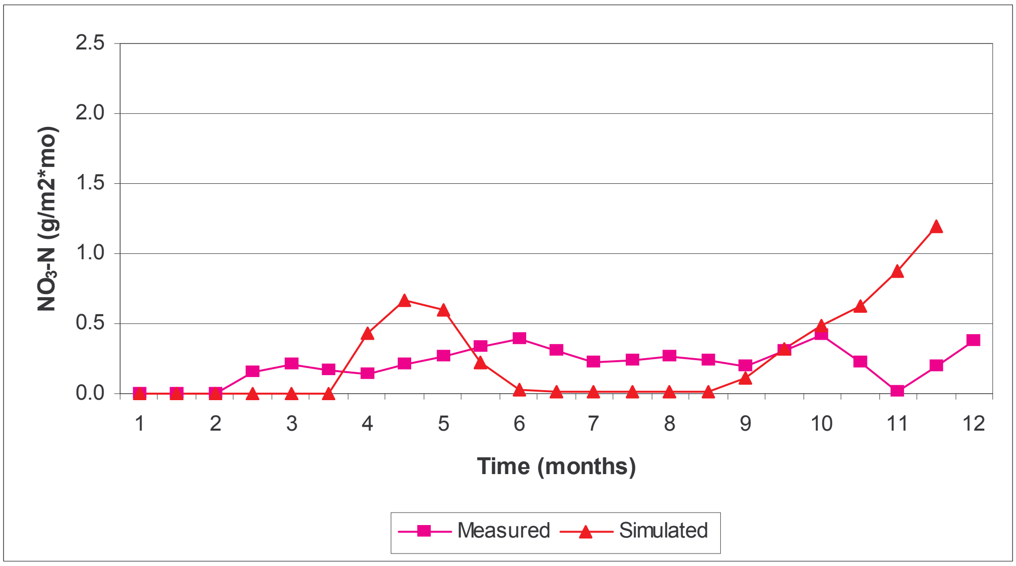


Figure 4.2.10 Model simulation for NO_3-N in 2001

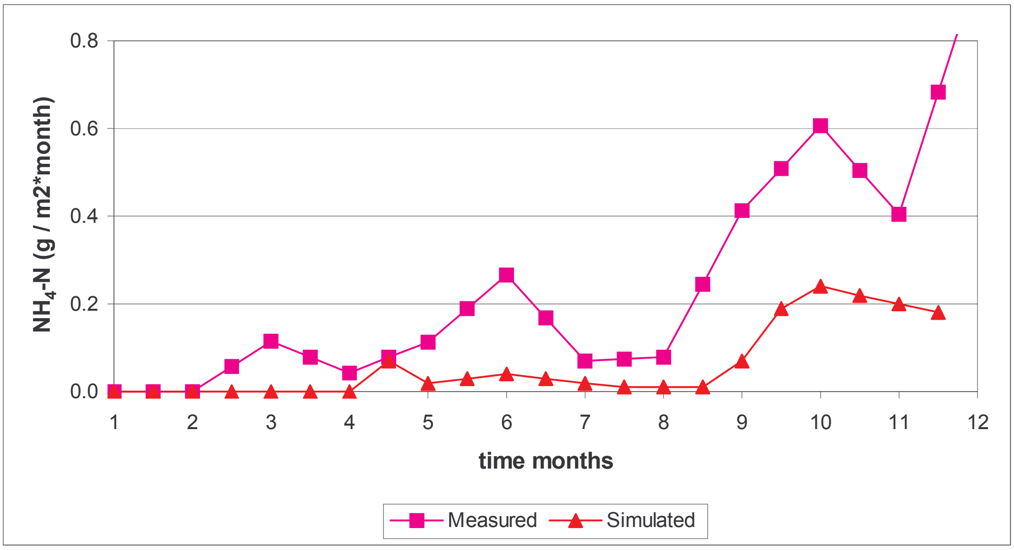


Figure 4.2.11 Model simulation for NH_4-N in 2001

In overall evaluation of model validation, high hydraulic conductivity was valid enough for both modeling the cumulative observed NO_3-N retention and simulating the seasonal dynamics of observed NO_3-N outputs from Lake Mogan reed beds in both of the study years. Whereas for NH_4-N output, the model results were not satisfactory for modeling NH_4-N output trends in none of the study years used for validation. As a result of model validation, the model was ready for making predictions but only for NO_3-N outputs from Lake Mogan reed beds.

4.2.3. Model predictions

After validation of high hydraulic conductivity model set for modeling NO_3-N retention dynamics, two model scenarios were created to predict the

NO_3-N output dynamics from Lake Mogan reed beds in wet and dry year conditions.

a) Wet year scenario

For creation of a wet-year scenario, surface NO_3-N load of year 1998, highest among the study years, was doubled (Table 4.2.6). In this wet year scenario, the model predicted a decrease in the overall retention rate of NO_3-N in Lake Mogan reed beds comparing to other high surface load years (1998, 1999, 2002) (Table 4.2.1, 4.2.3, 4.2.4 & 4.2.6).

Table 4.2.6. Summary of model predictions for wet year scenario

	High surface load
NO_3-N input (g/m^2*year)	45.61
NO_3-N output (g/m^2*year)	13.83
Rate of reduction (%)	69.7

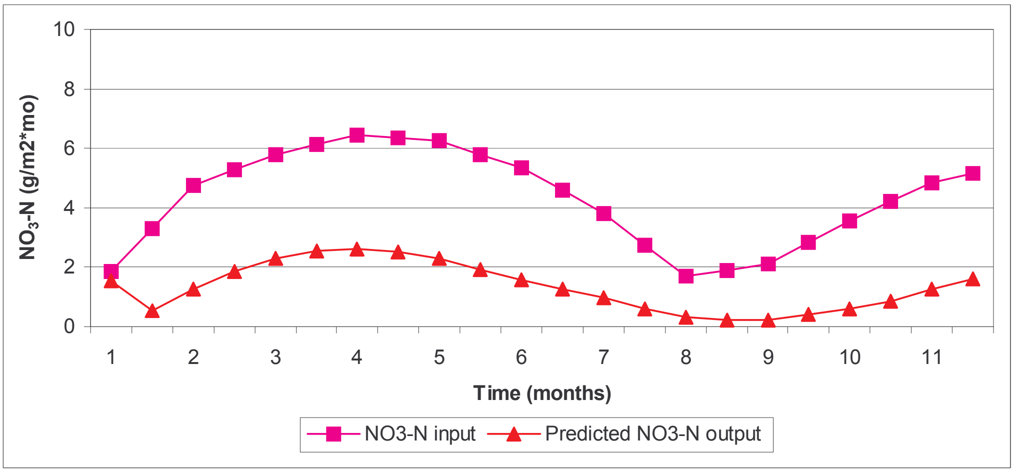


Figure 4.2.12 Model prediction of NO_3-N output for wet year scenario

b) Dry year scenario

As dry-year scenario, surface NO_3-N load of year 2001, lowest among the study years, was lowered to the half (Table 4.2.7). In this dry year scenario, the model predicted a similar retention rate of NO_3-N in Lake Mogan reed beds comparing to other low surface load year, 2001 (Table 4.2.5 & 4.2.7).

Table 4.2.7. Summary of model predictions for dry year scenario

	Low surface load
NO_3-N input (g/m^2*year)	1.75
NO_3-N output (g/m^2*year)	1.18
Rate of reduction (%)	32.6

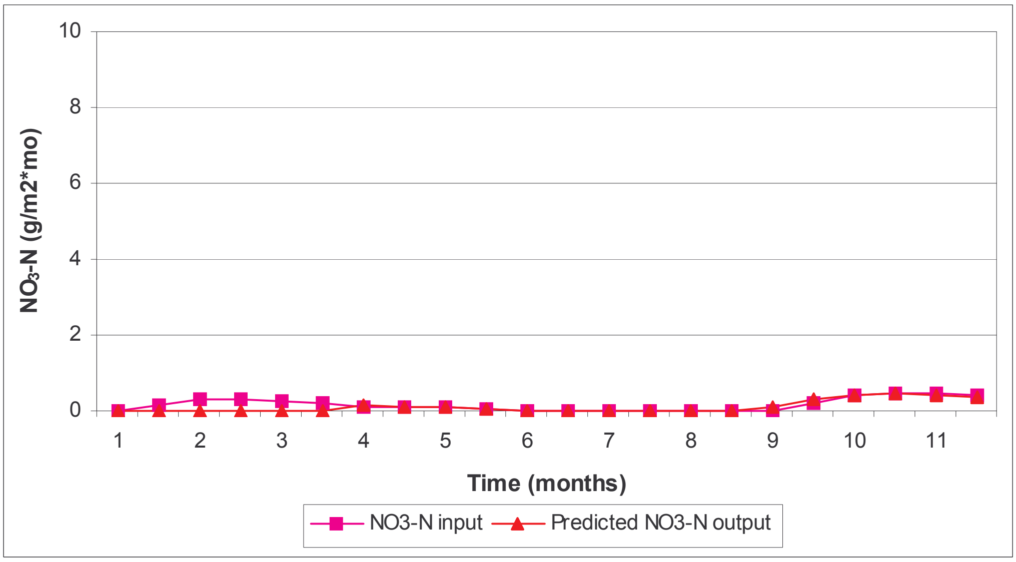


Figure 4.2.13 Model prediction of NO_3-N output for dry year scenario

4.3 GIS Analyses

4.3.1 Land-use changes in Lake Mogan Catchment

Population of Gölbaşı district increased more than three fold over a 20 years period of time (Table 4.3.1). Agricultural areas in the vicinity of Lake Mogan showed a slight decrease from 1978 to 1991 and stayed almost constant until the year 1999. The amount of wetland area surrounding the lake did not show a considerable increase from 1991 to 1999. Lake Mogan surface area and the surrounding wetland could not be measured for year 1978, because of unavailability of an aerial photo covering the whole lake in that year. Surface area of the lake increased from 1991 to 1999.

Then, for the given study year, nitrogen load to Lake Mogan from the human population was estimated based on the available census data and literature values of wastewater nitrogen content per capita (Table 4.3.2).

Table 4.3.1. Land use and population trend of the catchment over time

Year	# of houses (*)	Population (**)	Agricultural area (*)	Lake Mogan Wetland –the reed beds (*)	Settement area (*)	Mogan Lake (*)
1978	400	10491	2432.5 ha	-	12.6 ha	-
1991	2196	25123	2195.2 ha	157.7 ha	66.3 ha	491.9 ha
1999	2775	35308	2172.1 ha	154.7 ha	95.8 ha	542.5 ha

(*) estimated from the aerial photographs
(**) Census data (SIS, State Institute of Statistics)

Table 4.3.2. Nitrogen load caused by the population

Year	Population	NO_3-N (g/cap*day) (1)	Wetland area (ha)	Total TN Load (kg/day)	Total NO_3-N load (kg/day)	NO_3-N aerial load (g/m^2*year)
1978						
	10491	4.5	-	94.42	**47.21**	-
1991						
	25123	4.5	157.7	226.1	**113.05**	(25123 * 4.5 * 365) / 1577000 = **26.17**
1999						
	35308	4.5	154.7	317.78	**158.89**	(35308 * 4.5 * 365) / 1547000 = **37.49**

(1) TN = 6 - 12 g/cap*day (9 g/ cap*day taken)
NO_3-N = 0 - 0.5 * TN (Wastewater treatment for the control of environmental pollution, Prof. Soli J Arceivala, 1986)

The highest total NO_3-N load resulting form the population was in 1999, in which the population was highest (Table 4.3.2). In the same manner, highest TP input to the wetland occurred in 1999, the highest population year (Table 4.3.3).

Table 4.3.3. TP load caused by the population

Year	Population	TP (g/cap*day) (2)	Wetland (ha)	Total TP Load (kg/day)	TP areal load (g/m2*year)
1978					
	10491	0.6	-	**6.3**	-
1991					
	25123	0.6	157.7	**15.1**	(25123 * 0.6 * 365) / 1577000 = **3.48**
1999					
	35308	0.6	154.7	**21.2**	(35308 * 0.6 * 365) / 1547000 = **5.00**

(2) TP = 0.6 - 4.5 g/cap*day (Wastewater treatment for the control of environmental pollution, Prof. Soli J Arceivala, 1986)

After population based calculations, land-use maps of Lake Mogan near catchment were created using the aerial photos belonging to the years of 1978, 1991 and 1999. Then, the changes occurred in the near catchment were determined by overlay of the layers between the mentioned years using ESRI's ArcGIS 8.1 software.

The following figures reflect the changes occurred in the agricultural and settlement land-use patterns between the years 1978, 1991 and 1999 in the close vicinity of Lake Mogan catchment (Figure 4.3.1 – Figure 4.3.4). As it was also given in the Table 4.3.1, the amount of agricultural use of the land didn't changed significantly between 1991 and 1999 (Figure 4.3.1). However, when the agricultural lands of 1978 and 1999 compared, it was observed the agricultural area decreased especially around Lake Eymir and northeast part of Lake Mogan (Figure 4.3.2).

The amount of settlements increased distinctively from 1978 to 1991 and 1999 (Table 4.3.1, Figure 4.3.3 & Figure 4.3.4). Also looking at the earlier photos of the region, it was observed that there were almost no settlement areas in Golbasi district and there were very little agricultural activities which located only in the south of Lake Mogan (Figure 4.3.5).

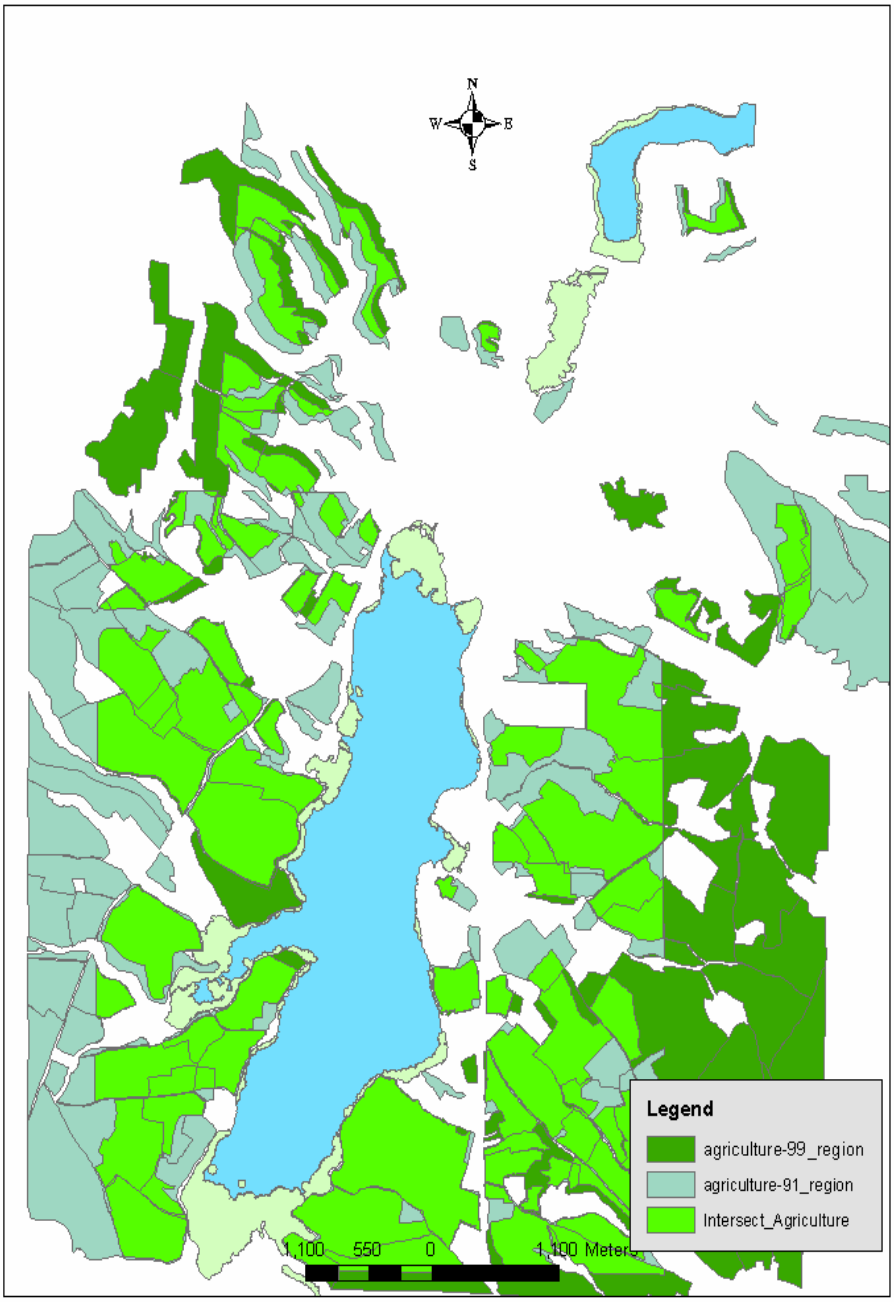


Figure 4.3.1. Changes in the agricultural land-use from 1991 to 1999

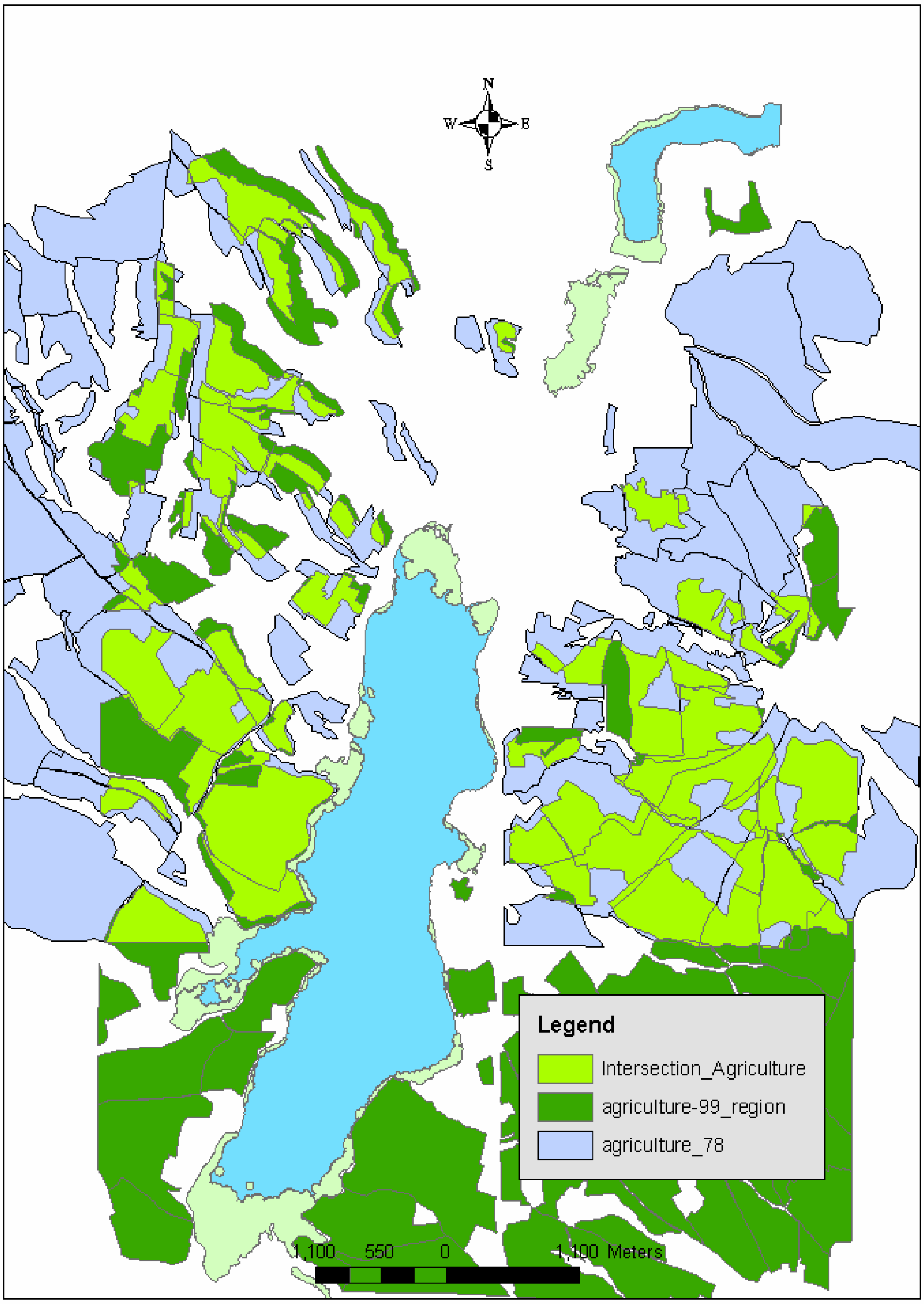


Figure 4.3.2. Changes in the agricultural land-use from 1978 to 1999

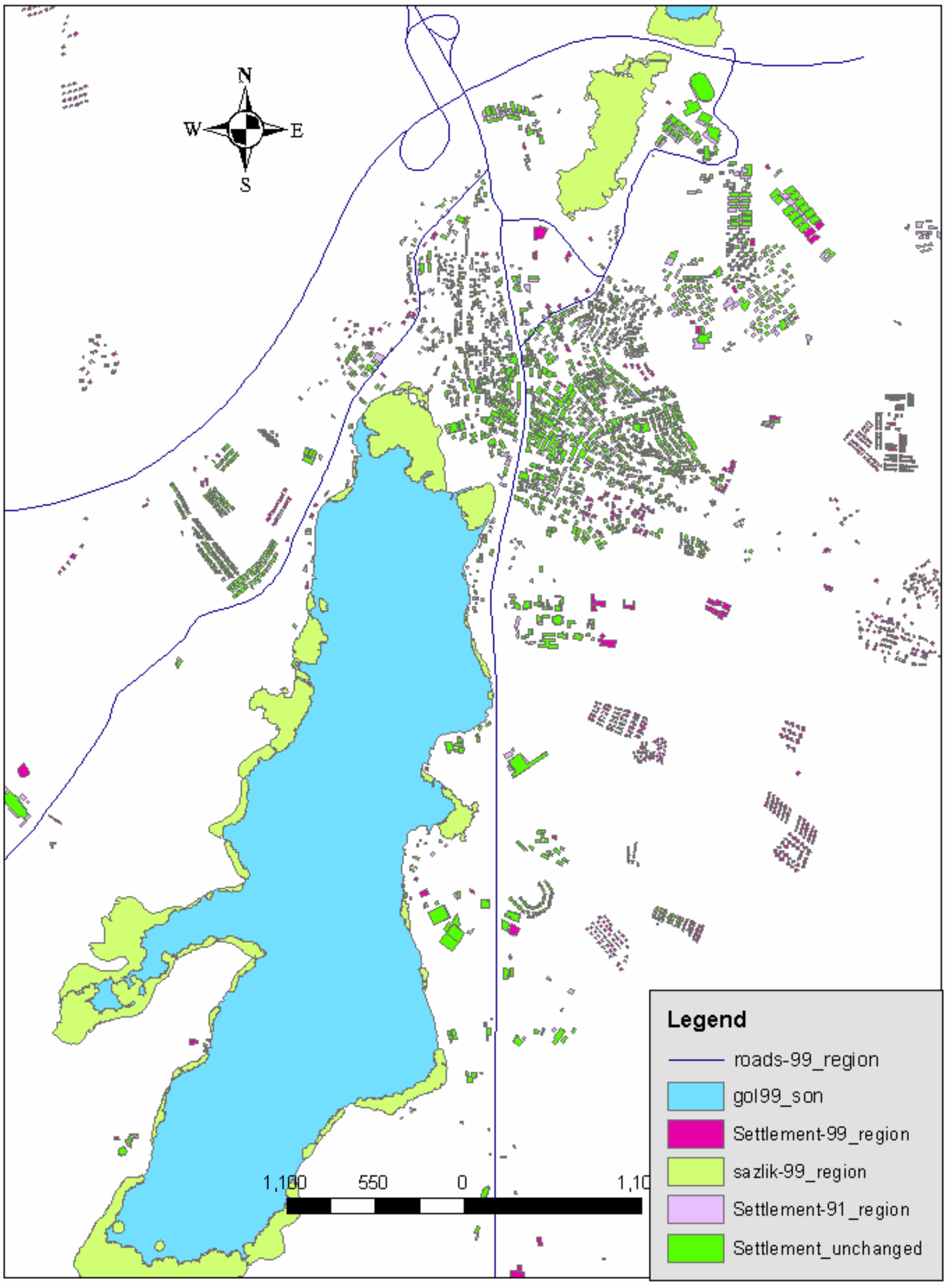


Figure 4.3.3. Changes in the settlement areas from 1991 to 1999

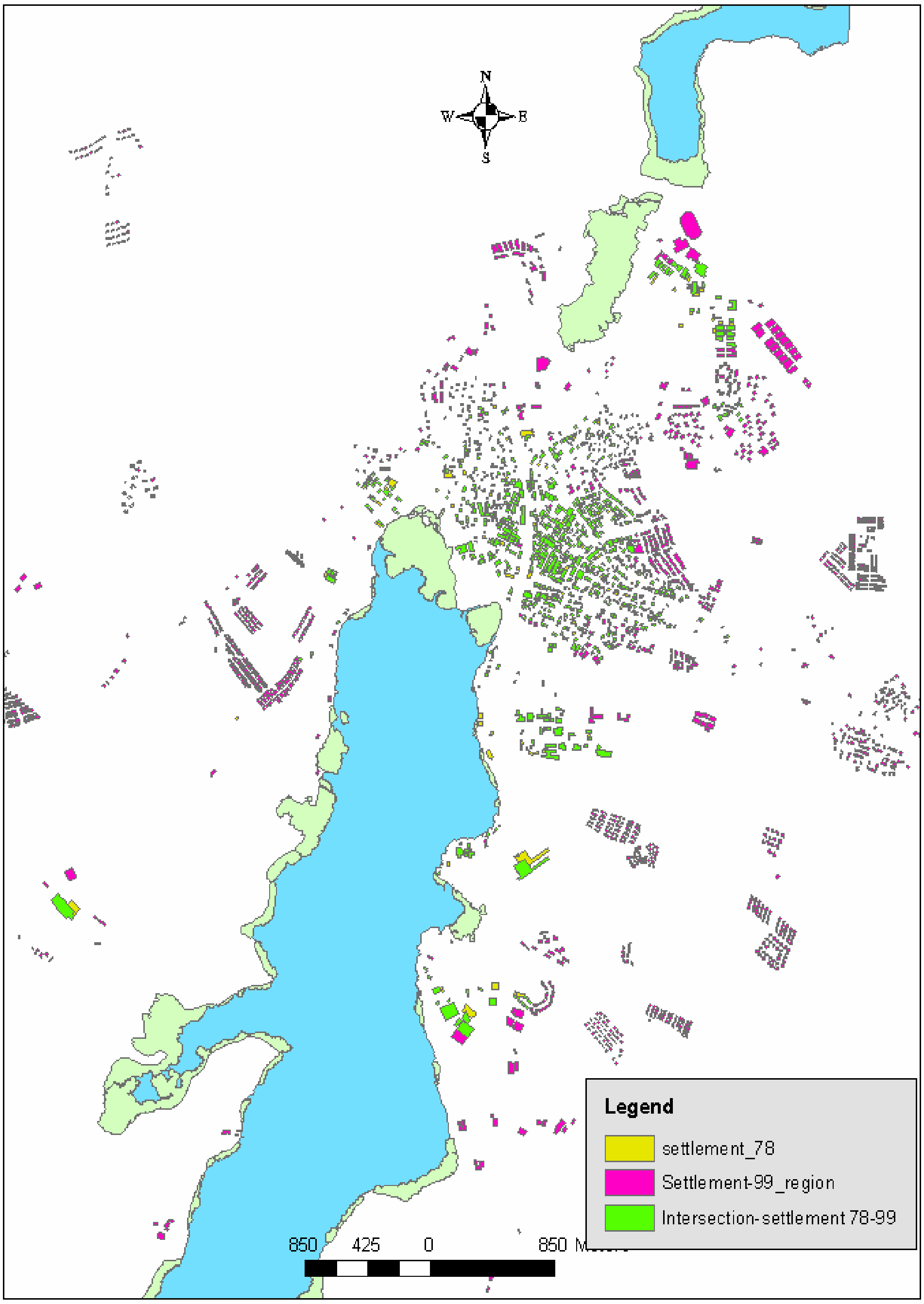


Figure 4.3.4. Changes in the settlement areas from 1978 to 1999

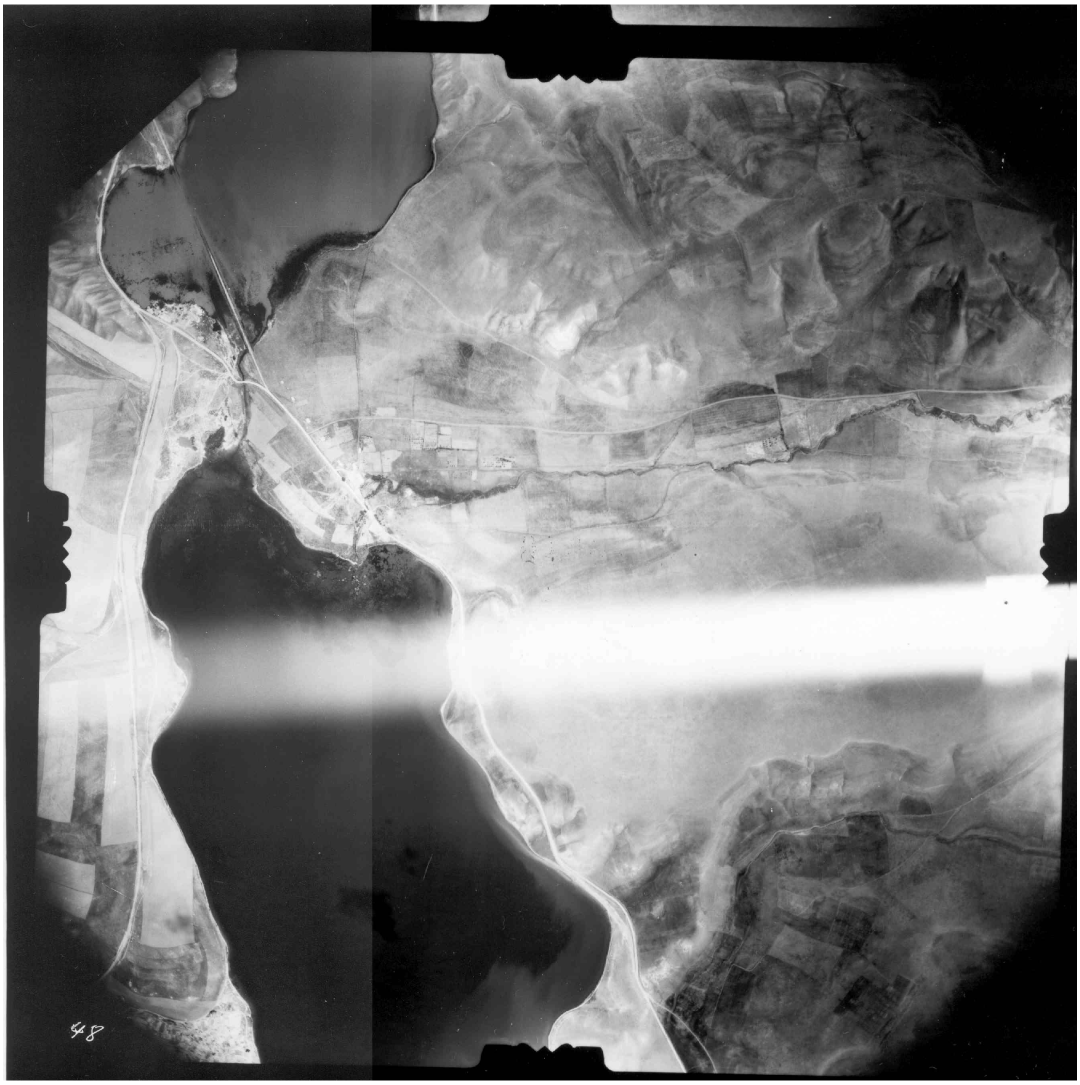


Figure 4.3.5. 1940's view of the region showing Lake Mogan and Eymir

4.3.2 Determining the potential risk areas for non-point N input to Lake Mogan

Spatial Analyst Tool of ArcGIS 8.1 was used to determine the potential non-point nitrogen source sites around Lake Mogan and the below procedure followed for the evaluation;

As intermediate steps, firstly slope classification map was drawn giving higher values (indicating higher risk) to steeper slopes. After the slope classification, distances to rivers and lake layers were classified in to 1 to 10 values with equal interval method, giving a 10 value to areas closest to the layers. Finally, a potential risk areas map was drawn by the combination of above three maps (Figure 4.3.6). In the risk map, lighter colors or higher risk values indicated higher risk areas for potential nonpoint nutrients inputs from the extensive agricultural areas to Lake Mogan. Then, it was observed that highest potential risk area for non-point nitrogen input around the lake was north-east of the lake. Also some high risk areas was observed in the west side of the lake. In contrary, least potential risk area for non-point N input was observed as the north end of the lake, where had the lowest slope and almost a flat landscape character (Figure 4.3.6).

The parameters affecting the risk were chosen according to their regulating capacity and simplicity. In this respect, slope is the strongest controlling mechanism in the transport of N to the lake and it had the highest weight in the calculations. The other parameter, distance to creek was decided because of occurrence of main N transport to the lake through the creeks flowing into. The final parameter, distance to the lake was decided because of the addition effect of N amount while the surface flow of it, which indicates N content of the surface flow increases getting closer to the lake.

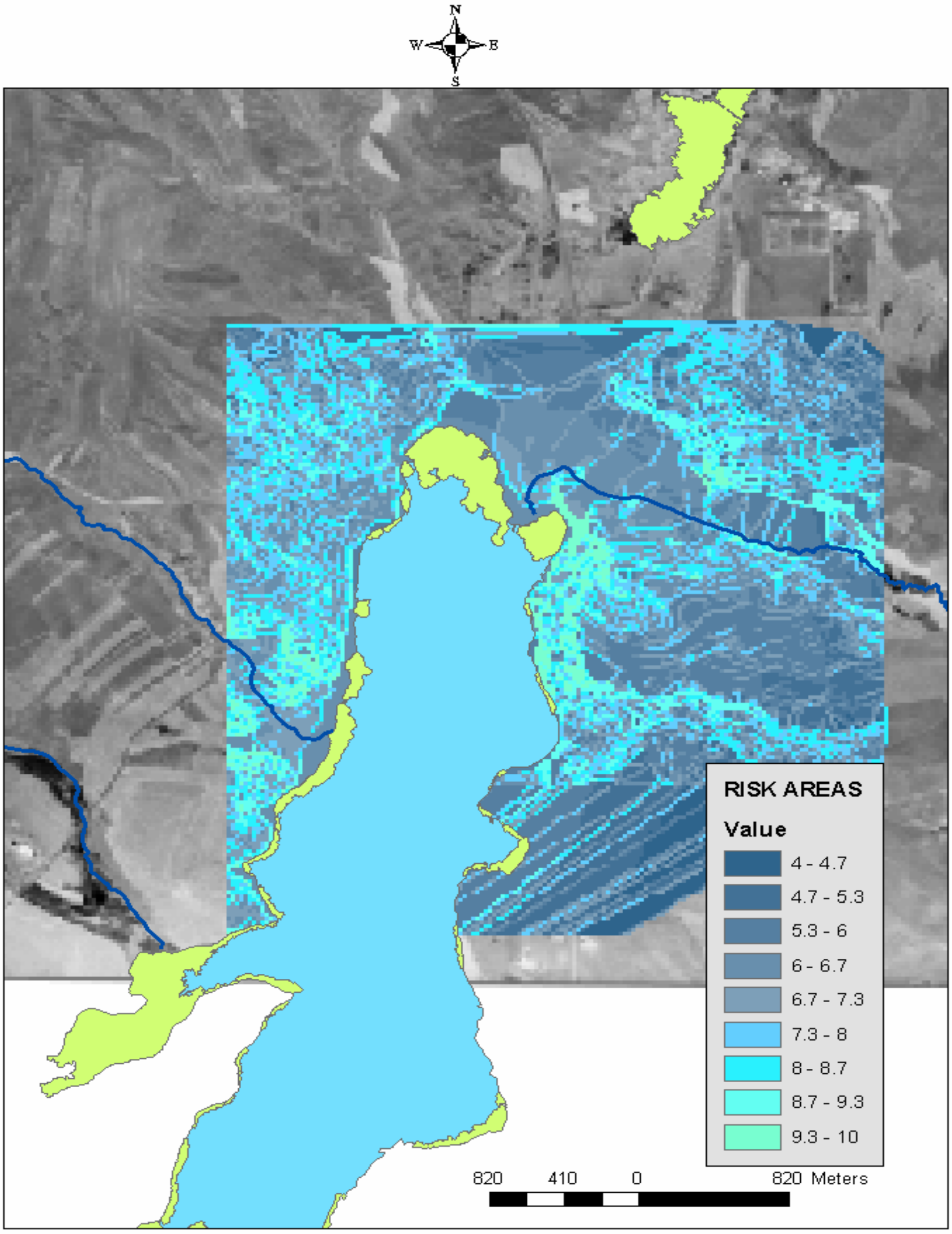


Figure 4.3.6. Map of potential risk areas for non-point N input to Lake Mogan

CHAPTER 5

DISCUSSION

5.1. Analysis of DIN dynamics in Lake Mogan reed beds

Seasonal and yearly dynamics of NO_3-N and NH_4-N inputs and outputs for related years were analyzed for each year separately and. Then, yearly DIN retentions estimated to compare the changes between the years and to find out main controlling mechanisms of DIN retention in Lake Mogan reed beds for long term.

Most of DIN inflow to Lake Mogan reed beds occurred through the surface run-off in the study years. DIN surface load to Lake Mogan reed beds was considerably higher in spring and autumn months when the surface run-off water to the system was high. In the inflows, NO_3-N + NO_2-N (total oxidized nitrogen) and NH_4-N surface loads were considered as the main nitrogen inflows to Lake Mogan reed beds. For the study years, N-input to Lake Mogan reed beds system occurred almost 90% in the form of NO_3-N (Table 4.1.1), especially in the wet years. This indicated that non-point nitrogen in the form of NO_3-N came from extensive agricultural lands which mainly contributed for DIN-input to Lake Mogan. Agriculture is one of the important NO_3-N sources since inorganic fertilizers frequently applied in agricultural activities contain large amount of NO_3-N.

Hammer and Knight (1994) reported that 43 constructed and natural treatment reed beds had average loadings of 0.81 to 1.kg kg N ha^{-1} day^{-1}. In present study, average DIN loads to Lake Mogan reed beds changed between 0.16 (2001) to 0.74 (1998) kg DIN ha^{-1} day^{-1}. These range of DIN loading to Lake Mogan reed beds can actually be considered as average loading since Johnston *et al.*, (1991) stated a high load to be 2 kg N ha^{-1} day^{-1} and low to be 0.005 kg N ha^{-1} day^{-1}.

Still, 1998, 1999 and 2002 were relatively high-load years in terms of N-input to Lake Mogan reed beds, compared with the DIN-loadings of 1997, 2000 and 2001. Specifically, NO_3-N input of 1998 was significantly different from that of years 1997, 2000 and 2001 (Table 4.1.1). In the same manner, a significant difference was observed between NO_3-N input and output for the relatively high-load years indicating significantly high NO_3-N retention rates for that periods (Table 4.1.2), while no significant difference was observed in the relatively low-load years. Higher retention of NO_3-N in relatively high-load years can be attributed to enhanced denitrification in the reed beds under waterlogged conditions (Beklioglu & Tan, submitted; Coops, Beklioglu & Crismen, 2003). In contrast, in the low-load periods the reed beds might have had limited denitrification capacity at receding water because of drought and, this probably in turn leading to higher N-output from the wetland (Beklioglu & Tan, submitted; Coops et al., 2003). However, this increase in the N-output was still not high enough to be reflected as the increase in the lake N-concentrations (Beklioglu & Tan, submitted). NH_4-N input and output were not significantly different in none of the study years (Table 4.1.3).

In terms of seasonality, main nitrogen input occurred in spring and autumn months when most of surface water inflow also occurred. Main retention of nitrogen also happened especially in spring that can be explained by both high DIN-input and plant uptake as it is the growing season of emergant plants (Davidsson & Leonardson, 1996). Relatively lower retentions or N-release despite high N input in autumn months can be attributed to dying of wetland plants and less denitrification capacity caused by colder weather conditions. Generally, temperature is considered to be an important factor regulating denitrification activity in soil and sediment, and positive relationships are usually shown (Hill, 1988; van Kessel, 1976). However, as it is stated before, denitrification process is mainly regulated by the amount of nitrate input to a wetland (Davidsson & Leonardson, 1996; Davidsson *et al.*, 1998). This is why, there was very little or no N-retention in summer months in the reed beds since the load was absent at the same time period. Also, the effect of temperature on NO_3-N showed an asymptotic pattern, which indicated that after certain value of temperature it had negative effect on retention DIN in Lake Mogan reed beds (Figure 4.1.13).

When NO_3-N and NH_4-N retention dynamics are compared for the given years, main DIN-retention in Lake Mogan reed beds took place for NO_3-N part whereas, whether NH_4-N was retained in or released from the reed beds showed a great variability. Main reason of this difference can be explained by the different denitrification rates of two nitrogen forms. Denitrification forms the primary mechanism of N removal from wetland systems and the pathway for denitrification of NO_3-N is much more shorter than the one for NH_4-N.

Lower or no retention of NH_4-N in Lake Mogan reed beds can be attributed to NH_4-N production in the wetland caused by the mineralization of organic nitrogen to NH_4-N (Cabrera, 1993; Leonardson *et al.*, 1994) and also relatively lower NH_4-N load to the wetland. Main source of NH_4-N is the domestic wastewater load to the freshwater systems, which can be assumed lower compared to agricultural nitrogen pollution in the case of Mogan reed beds. However this situation might change in the future due to increasing settlement areas around Lake Mogan.

It has been well established that nitrogen retention increases with nitrogen loading in aquatic systems (Jensen *et al.*, 1990; Gale et al., 1993; Windolf *et al.*, 1996). A study by Fleischer & Stibe (1991) found that nitrogen loading was an excellent predictor (R^2 = 0.94, $p<0.05$, $n=50$) of nitrogen retention in lakes, rivers and wetlands in Europe. In this study, it was observed that there was a clear linear relationship (R^2 = 0.975) between amount of NO_3-N retention and amount of NO_3-N input to the system (Figure 4.1.9). The same was also true for the relation of DIN input and retention (r^2 = 0.91) (Figure 4.1.10). Consequently, amount of NO_3-N and indeed DIN load can be used as an effective predictor of NO_3-N and DIN retention for Lake

Mogan reed beds. The importance of input quantity also can be used to explain low or no NH_4-N retention in the reed beds because amount of NH_4-N input constituted only around 10% of DIN input.

The water level & the retention of nitrogen in Lake Mogan reed beds for the given years well correlated. The higher water level was, the higher the N-retention was in the reed beds. This can be attributed to formation of suitable oxic-anoxic conditions for denitrification. This result was in line with the efficiency of denitrification, then the rate of NO_3-N retention, is enhanced in wetlands by occurrence of anoxic and oxic conditions with inundation at high lake levels (Saunders & Kalff, 2001; Sanchez-Carrillo & Alvarez-Cobeleas, 2001). Although, there was not a linear relationship between depth and retention, a threshold water level existed, below which an abrupt decrease in retention rate occurred.

5.2. Modeling

Environmental management is concerned with the question: how much nitrogen can be removed if a wetland is not drained or re-established (Jorgensen, 1994). In this study, a relatively simple model with the nine state variables was developed to attempt to answer this question. The model is general in the sense that it can be applied to all wetlands provided that a five site specific parameters are known, that the local climatic conditions are used as forcing functions in the model and that some basic information about the wetland is applied in the calibration phase (Jorgensen, 1994). This property of the model that developed by Jorgensen (1994), made it possible to apply it in Lake Mogan reed beds with the availability of necessary site specific information.

Diffusive transport of N species from the aerobic water and substrate to the anaerobic substrate layer is an important regulator of denitrification (Martin and Reddy, 1997). In this respect, three different sets of hydraulic conductivity (infiltration rate of water) were used in the model calibration because it determines the efficiency of denitrification in the wetland environment by regulating the horizontal and vertical movements of water or simply the retention of water.

As it is stated before, study years can be divided into two sub-groups as high load years and low load years in terms hydraulic and nutrient load; 1998, 1999 and 2002 were relatively high-load years; 1997, 2000 and 2001 were relatively low-load years.

Data sets of year 1999, 2000 and 2002 were used to calibrate the model parameters. Firstly, uptake rates for nitrate-N and ammonium-N and the mineralization rate were calibrated to give the observed trends in maximum value of plant nitrogen content and also seasonal dynamics of the mentioned parameters (Figure 4.2.1). After calibration of uptake rates, the model was simulated with different values of hydraulic conductivity to find out best-suited one.

In model calibration, high hydraulic conductivity model set best simulated the observed NO_3-N output from Lake Mogan reed beds. Whereas low hydraulic conductivity model set generally better predicted cumulative NH_4-N output, high hydraulic conductivity still satisfactorily simulated seasonal output dynamics of NH_4-N, too. As a result, high hydraulic conductivity model set was chosen for further validation of Lake Mogan Wetland Model.

High hydraulic conductivity indicates higher rate of infiltrating water in the reed beds soil. Davidsson & Leonardson (1997) stated that the infiltrating water may enhance leaking of dissolved forms of both organic and inorganic nitrogen from the soil. In this respect, Davidsson & Leonardson *et. al.*, (1997) proposed several reasons why high hydraulic conductivity and in turn high infiltration rate in wetland soil should promote efficient nitrogen removal. Since nitrate is the main fraction of nitrogen being transported in steams, an efficient removal relies on high denitrification activity. The denitrification potential can be high for at least two reasons: 1) The deposited organic matter may serve as an energy source for micro-organisms including denitrifiers, 2) The downward movement of the water and nutrients promotes a large contact area between denitrifiers, nitrate and soil organic carbon (Davidsson & Leonardson, 1997).

After calibrating the model, data sets of year 1998 and 2001, the most wet and dry study years, were used to validate the model to be able to validate it in different and more extreme conditions. High hydraulic conductivity was valid enough for both modeling the cumulative observed NO_3-N retention and simulating the seasonal dynamics of observed NO_3-N outputs from Lake Mogan reed beds in both of the study years used for validation. However, the model results were not satisfactory for modeling NH_4-N output trends in none of the study years used for validation. As a result of model validation, the model was ready for making predictions but only for NO_3-N outputs from Lake Mogan reed beds.

The model predictions revealed that, there is an upper limit for the retention capacity of NO_3-N in Lake Mogan reed beds after which the rate of retention starts to decrease. Still, the model predictions also showed that NO_3-N retention efficiency is distinctively higher in wet rather than the dry year conditions since the reed beds might have limited denitrification capacity due to unavailability of enough NO_3-N load (Beklioglu & Tan, submitted; Coops et al., 2003).

Lake Mogan reed bed model is comparable to several other models of nitrate removal in wetlands. Dorge (1994) presented a model of nitrogen retention in freshwater wetlands that includes hydrology, nitrogen cycling dynamics and plant uptake in the prediction of N removal. Similar to our model, it included a biological submodel in addition to the hydrology submodel and applied to the prediction and understanding of the specific nitrogen cycling of a given wetland.

Yamaguchi *et al.*, (1992) presented a model of spatial variation of nitrification and denitrification based on soil column leaching experiments. In his model, soil composition, temperature, and column depth had a large influence on model performance. Nitrate availability has been shown to be as important as these physical parameters in wetland denitrification (Boustany *et al.*, 1997). However, in our model, nitrate availability was not a limiting factor to removal efficiency as Lake Mogan reed beds were reasonably nitrate enriched.

Breen (1990) suggested a conceptual model of nutrient mass balance in wetlands and identified plant uptake, mineralization, and denitrification as important processes in the total nutrient flux among compartments. Martin and Reddy (1997) introduced a relatively complex model of nitrogen transformations in the water column and wetland soil profile which draws on extensive literature data on the wetland nitrogen cycle. The model used in this study also followed a similar approach with Martin and Reddy (1997) and included all the important nitrogen processes in the model.

Spieles and Mitsch (2000) developed a more general and simple Vollenweider-type model of nitrate retention based on seasonal temperature, hydraulic loading, and nitrate loading. Their study demonstrated that, in simulations of wetland ecosystem-level nitrate removal, the biological submodel may be approximated with a minimal amount of detail. Their model aggregated many pathways of the nitrogen cycle and still predicted nitrate retention with reasonable accuracy, which made it a useful tool for management purposes (Spieles & Mitsch, 2000). For management purposes, such a model would be more easy to apply in the case of Lake Mogan reed beds than the model applied in this study.

Oğuz & Dinç (2003) applied a 1^{st} layer PAMOLARE model Developed by S.E. Jorgensen, H. Tsuno and T. Hidaka (2000) which consists of a combination of causal dynamic model, and a set of associated empirical models and used for the prediction of water quality changes in Sapanca Lake under six different scenarios. Oğuz & Oğuz (1995) also applied a Vollenweider-type model to Sapanca Lake for prediction of lake response to phosphorus loading.

5.3 Land-use changes in the Lake Mogan Basin

Change detection is the use of remotely sensed data (based on aerial photos or satellite images) to map the spatial extent and magnitude of changes in the land surface through time, which may cover vegetation, urban growth, industrial development, surface water changes, etc.

Land use and land cover inventory methods are generally accomplished in one of three ways (Swanson, 2003). First method is based upon human interpretation of aerial photography. Using this method an air photo interpreter would identify different classes of land use on the photography and draw boundaries around them. This method results in a map that depicts land use and land cover as a series of polygons. Another method is based upon the automated classification of satellite imagery. Using this method, the image interpreter uses complex software and knowledge of the specific location of different types of land cover in the image, to instruct the computer to classify the entire image into land cover classes. The third method involves the identification of sample points across the landscape and determining land use and cover at these points. Statistical methods are then used to make inferences about land use and cover across the entire landscape (Swanson, 2003).

In this study, the first methodology was applied since the aerial photo interpretation approach is generally best suited for inventories where land use is more important than the land cover and its simplicity compared to statistical method. Although the time period used in this study was not long enough to detect distinct changes in Lake Mogan Basin, some clear changes still could be observed. Firstly, there was a high rate of formation of new settlement areas in the basin especially in the southeastern part of the study area, which can be attributed to the new trend of building summer resorts around Lake Mogan.

Secondly, there was a decrease in the agricultural landuse. This can simply be explained by the increase in the settlement areas because of the suitability and preference of agricultural lands for construction.

When taking into account the early photos of the region (Figure 4.3.5), the region has undergone a distinctive change from a wetland area with almost no settlements and very few agricultural lands to a dense and developing settlement and extensive agricultural lands. As it can be expected, these changes had important impacts on the ecological situation of the lake and will have also in the future.

5.4 Determining the risk areas for non-point N sources to Lake Mogan

Risk assessment analysis revealed that highest potential risk area for non-point N input around the lake was north-east of the lake. Also some high risk areas was observed in the west side of the lake. In contrary, least potential risk area for non-point N input was observed as the north end of the lake, where had the lowest slope and almost a flat landscape character.

These findings actually caused more important and dramatic results in terms of management when they were combined with the map of wetland areas surrounding the lake. It was observed that higher risk areas around the lake were least or less reed bed covered areas, especially in the north-east part, indicating those parts of the lake did not benefit from the buffer and preventive mechanisms of the wetland and more prone to direct non-point N inflow (Figure 4.3.6). More dramatically, most protected part of the lake by the presence of dense and large reed bed area was the north end of the lake, the least risky area for the non-point N inflow. This result can be explained by the suitability of flat areas for the growth of wetland plants.

On the other hand, the results of DIN input vs. output estimation section also clearly revealed that, as a buffer zone, Lake Mogan reed beds had an important role in the retention of non-point source nitrogen from the extensive agricultural areas in the catchment. Then, combining these results with the results of risk assessment analysis, it can be concluded that it would be required to re-establish the reed beds especially in the areas prone to non-point nitrogen input risk.

CHAPTER 6

CONCLUSION

Higher retention of NO_3-N in relatively high-load years can be attributed to enhanced denitrification in the reed beds under waterlogged conditions (Beklioglu & Tan, submitted; Coops, Beklioglu & Crismen, 2003).

Main retention of nitrogen also happened especially in spring that can be explained by both high DIN-input and plant uptake as it is the growing season of emergent plants (Davidsson & Leonardson, 1996). NH_4-N input and output were not significantly different in none of the study years. Lower or no retention of NH_4-N in Lake Mogan reed beds can be attributed to NH_4-N production in the wetland caused by the mineralization of organic nitrogen to NH_4-N (Cabrera, 1999; Leonardson *et al.*, 1994) and also relatively lower NH_4-N load to the wetland.

There was a clear linear relationship (R^2 = 0.975) between amount of NO_3-N retention and amount of NO_3-N input to the system. High hydraulic conductivity model set was valid enough for both modeling the cumulative observed NO_3-N retention and simulating the seasonal dynamics of observed NO_3-N outputs from Lake Mogan reed beds in both of the study years used for validation. However, the model results were not satisfactory for modeling NH_4-N output trends in none of the study years used for validation. As a result, the model was applied for making predictions under wet and dry year scenarios but only for NO_3-N outputs from Lake Mogan reed beds.

The model predictions revealed that, there is an upper limit for the retention capacity of NO_3-N in Lake Mogan reed beds after which the rate of retention starts to decrease. Still, the model predictions also showed that NO_3-N retention efficiency is distinctively higher in wet rather than the dry year conditions since the reed beds might have limited denitrification capacity due to unavailability of enough NO_3-N load (Beklioglu & Tan, submitted; Coops et al., 2003).

Risk assessment analysis revealed that highest potential risk area for non-point N input around the lake was north-east of the lake. Also some high risk areas was observed in the west side of the lake. In contrary, least potential risk area for non-point N input was observed as the north end of the lake, where had the lowest slope and almost a flat landscape character.

It was observed that higher risk areas around the lake were least or less reed bed covered areas, especially in the north-east part, indicating those parts of the lake did not benefit from the buffer and preventive mechanisms of the wetland and more prone to direct non-point N inflow (Figure 3.6). More dramatically, most protected part of the lake by the presence of dense and large reed bed area was the north end of the lake, the least risky area for the non-point N inflow.

On the other hand, the results of DIN input-output estimations and modeling also clearly revealed that, as a buffer zone, Lake Mogan reed beds had an important role in the retention of non-point source nitrogen from the extensive agricultural areas in the catchment. Then, combining these results with the results of risk assessment analysis, it can be concluded that it would be required to re-establish the reed beds especially in the areas prone to non-point nitrogen input risk.

As further studies it is suggested to build a GIS-based decision support system for Lake Mogan catchment. A GIS-based decision support system would provide an invaluable tool for all aspects of the land use planning process: conducting a land suitability analysis, conducting future land use demand, allocating this demand to suitable locations, and evaluating the likely impacts alternative policy choices and assumptions.

REFERENCES

Ambus, P., Lowrance, R., 1991. Comparison of denitrification in two riparian soils. Soil Sci. Soc. Am. J. 55: 994-997

Bartlett, M.S., Brown, I.C., Hanes, N.B., Nickerson, N.H., 1979. Denitrification in freshwater wetland soil. J. Environ. Qual. 8: 460-464

Basnyat, P., L.D. Teeter, B.G. Lockaby & K.M. Flynn, 1999. The use of remote sensing and GIS in watershed level analysis of non-point source pollution problems. Forest Ecology and Management 128: 65-73.

Beasley, D.B., Huggins, L.F., 1982. ANSWERS. Department of Agricultural Engineering, Pudue University, West Lafayette, USA.

Beklioğlu, M., G. Altınayar & C.O. Tan, 2000. Role of water level fluctuations, nutrients and fish in determining the macrophyte-dominated clear water states in five Turkish shallow lakes. Shallow Lake Wetlands: Ecology, Eutrophication and Restoration. International Workshop. 28-30 October 2001, Ankara, Turkey.

Beklioglu, M & C.O. Tan, submitted (a). The role of water level fluctuations for the submerged plant development and the subsequent ecological processes in Lake Mogan, Turkey. Hydrobiologia.

Boustancy, R.G., Crozier, C.R., Rybezyk, J.M., Twilley, R.R., 1997. Denitrification in a South Louisiana wetland forest receiving treated sewage effluent. Wet. Ecol. Manag. 4, 273-283.

Breen, P.F., 1990. A mass balance method for assessing the potential of artificial wetlands for wastewater treatment. Water Res. 24, 689-697.

Brix, H., Schierup, H., 1989. The use of aquatic macrophytes in water pollution control. Ambio 18: 100-107.

Brodrick, S.J., Cullen, P., Maher, W., 1988. Denitrification in a natural wetland receiving secondary treated effluent. Water Res. 22: 431-439.

Burnak, S.L., & M. Beklioglu, 2000. Macrophyte-dominated clearwater state of Lake Mogan. Turk. J. Zool. 24: 305-313.

Burrough, P.A., McDonnell R., Principles of Geographic Information Systems. Oxford: Clarendon Press, 1998.

Cabrera, M.L., 1993. Modeling the flush of nitrogen mineralization caused by drying and rewetting soils. Soil. Sci. Soc. Am. J. 57: 63-66.

Chaney, A., & E.P. Morbach, 1962. Modified reagents for the determination of urea and ammonia. Clin. Chem. 8: 130-132.

Chapin, F.S. III, Zavaleta, E.S., Eviner, V.T., Naylor, R.L., Vitousek, P.M., Reynolds, H.L., Hooper, D.U., Lavorel, S., Sala, O.E., Hobbie, S.E., Mack, M.C., & Diaz, S., 2000. Consequences of changing biodiversity. Nature, 405, 234-242.

Commission of the European Communities, 2002. Report on the implementation of Council Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources. COM (2002) 407 final.

Cooke, J.G., 1994. Nutrient transformations in a natural wetland receiving sewage effluent and the implications for waste treatment. Water Sci. Technol. 29: 209-217.

Coops, H., Beklioglu, M., Chrisman, T.L., 2003. The role of water level fluctuations in shallow lakes ecosystems – Workshop conclusions. Hydrobiologia.

Davidsson, T.E., L. Leonardson, 1996. Seasonal dynamics of denitrification activity in two water meadows. Hydrobiologia 364: 189-198.

Davidsson, T.E., L. Leonardson & P. Balkhag, 1998. Small-scale variation in denitrification, nitrate, dissolved organic carbon and nitrous oxide in a flooded wetland soil. Accepted for publication in Verh. Int. Ver. Limnol. Vol 26: 1328-1333.

De Roo, A.P.J., 1993. Modeling Surface Runoff and Soil Erosion in Catchments Using Geographical Information Systems. Faculteit Ruirntelijke Wetenschappen, Universiteit Utretch, Netherlands.

Dorge, J., 1994. Modeling nitrogen transformations in freshwater wetlands – estimating nitrogen retention and removal in natural wetlands in relation to their hydrology and nutrient loadings. Ecol. Model. 75, 409-420.

Douglas, I., 1999. Hydrological investigations of forest disturbance and land cover impacts in South-East Asia: a review. Philosophical Transactions of the Royal Society of London, Series B, 354, 1725-1738.

DSI. 1993. Mogan Gölü Limnolojik Etüd Raporuç Technıcal Reportç T.C. Enerji ve Tabi Kaynaklar Bakanlığı Devlet Su İşleri Genel Müdürlüğü İşletme ve Bakım Dairesi Başkanlığı, Ankara: 212 ppç (in Turkish).

EIE, 2001. Mogan ve Eymir Gölleri havzasının hidrometeorolojik özellikleri. Technical Report. Turkish General Directorate of Electrıcal Power, Resource Survey and Development Administration, Ankara: 159 pp. (in Turkish).

Fleischer, S. & L. Stibe, 1991. Drainage basin management – reducing river transported nitrogen. Verh. Int. Ver. Limnol. 24: 1753-1755.

Gale, P.M., K.R. Reddy & D.A. Graetz, 1993. Nitrogen removal from reclaimed water applied to constructed and natural wetland microcosms. Wat. Envir. Res. 65: 162-168

GIS by ESRI, 1994. Cell-based modeling with grid. Environmental Systems Research Institute Inc. (ESRI), USA.

Hammer, D.A., Knight, R.L., 1994. Designing constructed wetlands for nitrogen removal. Water Sci. Techol. 29: 15-27.

Hensel, H., Bork, H.R., 1988. Computer aided construction of soil erosion and deposition maps. Geol. Jahrb. A 104, 357-371.

Hill, A.R., 1988. Factors influencing nitrate depletion in a rural stream. Hydrobiologia 160: 11-122.

Hosomi, M., Murakami, A., Sudo, R., 1994. A four-year mass balance for a natural wetland system receiving domestic wastewater. Water Sci. Technol. 30: 235-244.

Houghton, R.A., 1994. The world-wide extent of land-use change. Bioscience. 44: 305-313.

Jansson, M., R. Andersson, H. Berggren & L. Leonardson, 1994. Wetlands and lakes as nitrogen traps. Ambio 23: 320-325

Jensen, J.P., E. Jeppesen, P. Kristensen, P.B. Christensen & M. Sondergaard, 1992. Nitrogen loss and denitrification as studied in relation to reductions in nitrogen loading in a shallow hypertrophic lake (Lake Sobygard, Denmark). Int. Rev. ges. Hydrobiol. 77: 29-42.

Johnston, C.A., 1991. Sediment and nutrient retention by freshwater wetlands: effects on surface water quality, Crit. Rev. Environ. Control 21: 491-565.

Jorgensen, S. E., 1989, "Principles of Ecological Modeling", in William J. Mitch and S. E. Jorgensen, Ecological Engineering: An Introduction to Ecotechnology, New York: John Wiley, 39-55.

Jorgensen, S. E., 1994. A general model of nitrogen removal by wetlands. Global Wetlands: Old World and New: 575-583

Kadlec, R.H., Knight, R.L., 1996. Treatment Wetlands. CRC Press. Boca Raton, FL.

Kirac, C.O., S. Karauz, A. Aykurt, M. G]rsan & O Arihan, 1995. Birds of Lake Mogan. Bird Count Report (in Turkish).

Knowles, R., 1982. Denitrification. Microbiol. Rev. 46: 43-70.

Leonardson, L., L. Bengtsson, T. Davidsson, T. Perrson & U. Emanuelsson, 1994. Nitrogen retention in artificially flooded meadows. Ambio 23: 332-341.

Mackereth, F.J., H.J. Heron & J.F. Talling, 1978. Water analyses: some methods for limnologists. Freshwater Biological Assoc. Scientific Publication No:36.

Martin, J.F., Reddy, K.R., 1997. Interaction and spatial distribution of wetland nitrogen processes. Ecol. Model. 105: 1-21.

Moffat, A. S., 1998. Global nitrogen overload problems grows critical. Science 279: 988-989

Moss, B., 1998. Ecology of Freshwaters: Man and Medium and Past to Future. 3rd edition. Blackwell Science, Oxford: 557 pp.

Narumalani, S., Y. Zhou & J. R. Jensen, 1996. Application of Remote Sensing and Geographical Information Systems to the delineation and analysis of riparian buffer zones. Aquatic Botany 58: 393-409.

Neely, R.K., Baker, J.L., 1989. Nitrogen and phosphorus dynamics and the fate of agricultural runoff. In: van der Valk, A.G. (Ed.), Northern Prairie Wetlands. Iowa State University Press, Ames, IA, pp. 92-131.

Oguz, M., Dinc, H., 2003. Pamolare Model for Management of Sapanca Lake. Proceedings of the Sixth International Conference on the Mediterranean Coastal Environment, MEDCOAST 03, E. Ozhan (Ed.), 7-11 October 2003, Ravenna, Italy.

Oguz, M., Oguz., M., 1995. Development of a Phosphorus Loading Model for Sapanca Lake. Intern. J. Environmental Studies, 1995, Vol. 47, pp. 87-103.

Ozesmi, U., 1999. Ecology and politics of rehabilitation: Mogan Lake wetland system, Ankara, Turkey. In Streever, W. (ed.), An International Perspective on Wetland Rehabilitation. Kluwer Academic Publishers, the Netherlands: 181-187.

Penner, J.E., 1994. Atmospheric chemistry and air quality. In: W.B. Meyer, B.L. Turner II (Eds.), Changes in land use and land cover: a global perspective (pp. 175-209). Cambridge: Cambridge University Press.

Phipps, R.G., Crumpton, W.G., 1994. Factors affecting nitrogen loss in experimental wetlands with different hydrologic loads. Ecol. Eng. 3: 399-408.

Reddy, K. R., Patrick, W.H., 1984. Nitrogen transformations and loss in flooded soils and sediments. CRC Crit. Rev. Environ. Control 13, 273-309.

Reddy, K. R., D'Angelo, E.M., 1994. Soil processes regulating water quality in wetlands. In: Mitsch, W.J. (Ed.), Global Wetlands: Old World and New. Elsevier, Amsterdam, pp. 309-324.

Renard, K.G., Foster, G.R., Weesies, G.A., Porter, J.P., 1991. Revised universal soil loss equation (RUSLE). J. Soil Water Conserv. 46 (1), 30-33.

Sanchez-Carrillo, S & M. Alvarez-Cobeleas, 2001. Nutrient dynamics and eutrophication patterns in a semi-arid wetland: the effects of fluctuating hydrology. Wat. Air Soil Pollut. 131: 97-118.

Saunders D.L., Kalf, J., 2001. Nitrogen retention in wetlands, lakes and rivers. Hydrobiologia 443: 205-212.

Schauble, 1999. Erosionsprognosen mit GIS and EDV - Ein Vergleich verschiedener Bewertungskonzepte arn Beispiel einer Gaulandschaft. Geographisches Institut, Universitat Tubingen, Germany.

Sivertun, A. & Prange, L., 2003. Non-point source critical area analysis in the Gisselo watershed using GIS. Environmental Modeling & Software 18 (2003) 887-898.

Sivertun, A., 1988. A GIS method to aid in non-point source critical area analysis. Int. J. Geogr. Inf. Syst. 2 (4), 365-378.

Skole, D.L., 1994. Data on global land-cover change: acquisition, assessment and analysis. In: W.B. Meyer, B.L. Turner II (Eds.), Changes in land use and land cover: a global perspective (pp. 437-471). Cambridge: Cambridge University Press.

Sommer, U., 1989. Nutrient status and nutrient competition of phytoplankton in a shallow lake. Limnol. Oceanogr. 34: 1162-1173.

Spieles, D.J., Mitsch, W.J., 2000. The effects of season and hydrologic and chemical loading on nitrate retention in constructed wetlands: a comparison of low- and high-nutrient riverine systems. Ecological Engineering 14: 77-91.

Stengel, E., Carduck, W., Jebsen, C., 1987. Evidence for denitrification in artificial wetlands. In: Reddy, K.R., Smith, W.H. (Eds.), Aquatic Plants for Water Treatment and Resource Recovery. Magnolia Publishing, Orlando, FL, pp. 543-550.

Sutton, P.M., Murphy, K.L., Dawson, R.N., 1975. Low temperature biological denitrification of wastewater. J. Water Pollut. Control Fed. 47: 122-134.

Tumas, R., 2000. Evaluation and prediction of nonpoint pollution in Lithuania. Ecological Engineering 14: 443-451.

Van Kessel, J.E., 1976. Factors affecting the denitrification rate in two water-sediment system. Wat. Res. 11: 259-267.

Viessman, W., Jr., Hammer, M.J., 1993. Water Supply and Pollution Control. Harper Collins College Publications, New York, 876 pp.

Vitousek, P. M., J. D. Aber, R. W. Howarth, G. E. Likens, P. A. Matson, D. W. Schindler, W. H. Schlesinger & D. G. Tilman, 1997. Human alterations of the global nitrogen cycle: sources and consequences. Ecol. Appl. 7: 737-750

Vollenweider, R.A., 1975. Input-output models with special reference to the phosphorus loading concept in limnology.

Wetzel, R. G., 1993. Future directions in research on controls of eutrophication. Mem. Ist. Ital. Idrobiol. 52: 71-78.

Windolf, J., E. Jeppesen, J.P. Jensen & P. Kristensen, 1996. Modeling of seasonal variation ib nitrogen retention and in-lake concentration: a four-year mass balance study in 16 shallow Danish lakes. Biogeochemistry 33: 25-44.

Wischmeier, W.H., Smith, D.D., 1978. Predicting Rainfall Erosion Losses. Agricultural Handbook 537. US Department of Agriculture, Washington, DC, USA.

Yamaguchi, T., Moldrup, P., Rolston, D.E., Hansen, J.A., 1992. A simple, inverse model for estimating nitrogen reaction rates from soil column leaching experiments at steady state water flow. Soil Sci. 154, 490-496.

Young, et al., 1987. AGNPS, Agricultural Non-point Source Pollution Model, a watershed analysis tool. Conservation Research Report 35, US Department of Agriculture, Washington, DC, USA.

APPENDIX A

Model equations of Lake Mogan Wetland Model

(Taken from Jorgensen et al., 1994)

absorbedN(t) = absorbedN(t - dt) + (exchNH4) * dt

INIT absorbedN = 200/9

INFLOWS:

exchNH4 = IF(absorbedN<200*NH4/(8+NH4)) THEN(NH4/(8+NH4)) ELSE(0)

detrN(t) = detrN(t - dt) + (decay - mineralization) * dt

INIT detrN = 1200

INFLOWS:

decay = (1.04^(temperature-20))*mort*0.9

OUTFLOWS:

mineralization = 0.0001*detrN*1.07^(temperature-20)

NH4(t) = NH4(t - dt) + (mineralization + inNH4 - uptake2 - exchNH4 - outNH4) * dt

INIT NH4 = 1.0

INFLOWS:

mineralization = 0.0001*detrN*1.07^(temperature-20)

inNH4 = (exch*surfNH4+0.01*(surfNH4-NH4))/soilw

```
OUTFLOWS:
uptake2 = IF(NH4>0) THEN (light*3.5*(1.05^(temperature-20))*NH4/(NO3+NH4)) ELSE (0)
exchNH4 = IF(absorbedN<200*NH4/(8+NH4)) THEN(NH4/(8+NH4)) ELSE(0)
outNH4 = outs*NH4/soilw
NO3(t) = NO3(t - dt) + (inNO3 - outNO3 - denit - uptake1) * dt
INIT NO3 = 15

INFLOWS:
inNO3 = (exch*NO3_surf_+0.01*(NO3_surf_ - NO3)) / soilw
OUTFLOWS:
outNO3 = outs*NO3/soilw
denit = 1.12 ^ (temperature - 20) * 16 * NO3 / (12+NO3)
uptake1 = if NO3 > 0 THEN light * 5.5* (1.05^(temperature-20)) * NO3 / (NO3 + NH4)  ELSE 0
NO3_surf_(t) = NO3_surf_(t - dt) + (insurfNO3_ + nitsurf - surfoutNO3 - downflowNO3 - denitsurf) * dt
INIT NO3_surf_ = 5

INFLOWS:
insurfNO3_ = GRAPH(t_)
(1.00, 0.00), (2.00, 0.00), (3.00, 4.13), (4.00, 4.48), (5.00, 1.77), (6.00, 0.829), (7.00, 0.153), (8.00, 0.01), (9.00, 0.106), (10.0, 0.397), (11.0, 0.89), (12.0, 0.927)
nitsurf = 8*(1.12^(temperature-20))*surfNH4/(8+surfNH4)
```

OUTFLOWS:

surfoutNO3 = ((outflow)*(NO3_surf_)+0.01*(NO3_surf_- NO3))/(sw)

downflowNO3 = exch*NO3_surf_/sw

denitsurf = (1.12^(temperature-20))*16*NO3_surf_/(12+NO3_surf_)

plantN(t) = plantN(t - dt) + (uptake1 + uptake2 - decay) * dt

INIT plantN = 10

INFLOWS:

uptake1 = if NO3 > 0 THEN light * 5.5* (1.05^(temperature-20)) * NO3 / (NO3 + NH4) ELSE 0

uptake2 = IF(NH4>0) THEN (light*3.5*(1.05^(temperature-20))*NH4/(NO3+NH4)) ELSE (0)

OUTFLOWS:

decay = (1.04^(temperature-20))*mort*0.9

soilw(t) = soilw(t - dt) + (exch - outs) * dt

INIT soilw = 2.0

INFLOWS:

exch = IF(sw>0) THEN(hydra_cond) ELSE(0)

OUTFLOWS:

outs = 0.1

surfNH4(t) = surfNH4(t - dt) + (insurfNH4 - nitsurf - surfoutNH4 - downflowNH4) * dt

INIT surfNH4 = 0.1

INFLOWS:

insurfNH4 = GRAPH(t_)

(1.00, 0.00), (2.00, 0.00), (3.00, 0.045), (4.00, 0.523), (5.00, 0.306), (6.00, 0.206),

(7.00, 0.044), (8.00, 0.009), (9.00, 0.002), (10.0, 0.034), (11.0, 0.198), (12.0, 0.202)

OUTFLOWS:

nitsurf = 8*(1.12^(temperature-20))*surfNH4/(8+surfNH4)

surfoutNH4 = ((surfNH4*outflow)+0.01* (surfNH4 - NH4))/(sw)

downflowNH4 = exch*surfNH4/sw

sw(t) = sw(t - dt) + (inflow + prec - evap - outflow - exch) * dt

INIT sw = 0.01

INFLOWS:

inflow = GRAPH(t_)

(1.00, 0.579), (2.00, 0.967), (3.00, 1.47), (4.00, 1.78), (5.00, 0.805), (6.00, 0.441),

(7.00, 0.08), (8.00, 0.16), (9.00, 0.074), (10.0, 0.393), (11.0, 0.527), (12.0, 0.697)

prec = GRAPH(t_)

(1.00, 0.037), (2.00, 0.078), (3.00, 0.05), (4.00, 0.028), (5.00, 0.02), (6.00, 0.049),

(7.00, 0.035), (8.00, 0.038), (9.00, 0.05), (10.0, 0.019), (11.0, 0.041), (12.0, 0.049)

OUTFLOWS:

evap = GRAPH(t_)

(1.00, 0.00), (2.00, 0.00), (3.00, 0.059), (4.00, 0.087), (5.00, 0.129), (6.00, 0.141),

(7.00, 0.246), (8.00, 0.235), (9.00, 0.121), (10.0, 0.072), (11.0, 0.023), (12.0, 0.00)

outflow = IF(sw>swmax) THEN (1.0*(sw-swmax)) ELSE(0)

exch = IF(sw>0) THEN(hydra_cond) ELSE(0)

hydra_cond = 0.3-1.5

light = 1.91-1.68*COS(6.1*((t_*30.42)-355)/365)

swmax = 0.05

total_wat = soilw+sw

t_ = TIME

mort = GRAPH(t_)

(1.00, 0.00), (2.00, 0.05), (3.00, 0.1), (4.00, 0.15), (5.00, 0.2), (6.00, 0.3), (7.00, 0.5), (8.00, 1.50), (9.00, 2.75), (10.0, 4.20), (11.0, 3.30), (12.0, 2.45)

temperature = GRAPH(t_)

(1.00, 3.30), (2.00, 3.30), (3.00, 6.60), (4.00, 12.1), (5.00, 16.9), (6.00, 20.0), (7.00, 24.4), (8.00, 23.8), (9.00, 18.8), (10.0, 13.9), (11.0, 6.70), (12.0, 5.00)

APPENDIX B

LINEAGE REPORT

Source Map 1: Lake Mogan Topology Base Map (3 Pafta)

Pafta No's: ANKARA-i29-b-22-a, ANKARA-i29-b-22-b, ANKARA-i29-b-22-c

Production Date: 1970

Provider: Tapu ve Kadastro Genel Müdürlüğü

Production Method: Aerial Photogometry

Scale: 1:5000

Modifications:

- Scanned to 72 pixels/inch resolution jpeg file
- Georeferenced using 9 Control Points given on the maps.
- Rectified to *Rectified.Mogan_a.tiff*, *Rectified.Mogan_b.tiff* and *Rectified.Mogan_c.tiff* files using first order affine.
- *UTM WGS_European_1984* (Zone 36) was assigned as the projection system

Source Map 2: Aerial Photo of Lake Mogan and Surroundings

Production Date: 11.09.1999

Provider: Harita Genel Komutanlığı

Producer: M.N.G. - ASKİ

Scale: 1:25 000

Modifications: - Scanned to 72 pixels/inch resolution jpeg file

- Georeferenced using Mogan Topology Base Map

- Rectified to *Mogan_99.tiff* file using first order affine function

- *UTM WGS_European_1984* was assigned as the projection system

Source Map 3: Aerial Photo of Lake Mogan and Eymir

Production Date: 11.09.1999

Provider: Harita Genel Komutanlığı

Producer: M.N.G. - ASKİ

Scale: 1:25 000

Modifications: - Scanned to 72 pixels/inc resolution jpeg file

- Georeferenced using Mogan Topology Base Map

- Rectified to *1999_2.tiff* file using second order affine function (to be able to combine with *Mogan_99.tiff*)

- *UTM WGS_European_1984* was assigned as the projection system

Source Map 4: Aerial Photo of Lake Mogan and Surroundings

Production Date: 1991

Provider: Harita Genel Komutanlığı

Scale: 1:25 000

Modifications: - Scanned to 72 pixels/inch resolution jpeg file

- Georeferenced using Mogan Topology Base Map

- Rectified to Mogan_91.tiff file using first order affine function

- *UTM WGS_European_1984* was assigned as the projection system

Source Map 5: Aerial Photo of Lake Mogan and Eymir

Production Date: 1991

Provider: Harita Genel Komutanlığı

Scale: 1:25 000

Modifications: - Scanned to 72 pixels/inch resolution jpeg file

- Georeferenced using Mogan Topology Base Map

- Rectified to *1991_2.tiff* file using second order affine function (to be able to combine with *Mogan_91.tiff*)

- *UTM WGS_European_1984* was assigned as the projection system

Data Name: Agriculture_99.shp

Data Type: Shape feature class

Data Geometry: Polygon

Datum: D_WGS_1984

Projection: UTM

Geog. Coor. System:	WGS_European_1984
Last Modification Date:	26.01.2003
Modifications:	- Digitized using *Mogan_99.tiff* and *1999_2.tiff*

Data Name:	Gol_99.shp
Data Type:	Shape feature class
Data Geometry:	Polygon
Datum:	D_WGS_1984
Projection:	UTM
Geog. Coor. System:	WGS_European_1984
Last Modification Date:	26.01.2003
Modifications:	- Digitized using *Mogan_99.tiff* and *1999_2.tiff*

Data Name:	Sazlik_99.shp
Data Type:	Shape feature class
Data Geometry:	Polygon
Datum:	D_WGS_1984
Projection:	UTM
Geog. Coor. System:	WGS_European_1984
Last Modification Date:	26.01.2003
Modifications:	- Digitized using *Mogan_99.tiff* and *1999_2.tiff*

Data Name:	Settlement_99.shp
Data Type:	Shape feature class
Data Geometry:	Polygon
Datum:	D_WGS_1984
Projection:	UTM
Geog. Coor. System:	WGS_European_1984
Last Modification Date:	26.01.2003
Modifications:	- Digitized using *Mogan_99.tiff* and *1999_2.tiff*

Data Name:	Meadow_99.shp
Data Type:	Shape feature class
Data Geometry:	Polygon
Datum:	D_WGS_1984
Projection:	UTM
Geog. Coor. System:	WGS_European_1984
Last Modification Date:	26.01.2003
Modifications:	- Digitized using *Mogan_99.tiff* and *1999_2.tiff*

Data Name:	Road_99.shp
Data Type:	Shape feature class
Data Geometry:	Line
Datum:	D_WGS_1984
Projection:	UTM

Geog. Coor. System: WGS_European_1984

Last Modification Date: 26.01.2003

Modifications: - Digitized using *Mogan_99.tiff* and *1999_2.tiff*

Data Name: Agriculture_91.shp

Data Type: Shape feature class

Data Geometry: Polygon

Datum: D_WGS_1984

Projection: UTM

Geog. Coor. System: WGS_European_1984

Last Modification Date: 26.01.2003

Modifications: - Digitized using *Mogan_91.tiff* and *1991_2.tiff*

Data Name: Gol_91.shp

Data Type: Shape feature class

Data Geometry: Polygon

Datum: D_WGS_1984

Projection: UTM

Geog. Coor. System: WGS_European_1984

Last Modification Date: 26.01.2003

Modifications: - Digitized using *Mogan_91.tiff* and *1991_2.tiff*

Data Name:	Sazlik_99.shp
Data Type:	Shape feature class
Data Geometry:	Polygon
Datum:	D_WGS_1984
Projection:	UTM
Geog. Coor. System:	WGS_European_1984
Last Modification Date:	26.01.2003
Modifications:	- Digitized using *Mogan_91.tiff* and *1991_2.tiff*

Data Name:	Settlement_91.shp
Data Type:	Shape feature class
Data Geometry:	Polygon
Datum:	D_WGS_1984
Projection:	UTM
Geog. Coor. System:	WGS_European_1984
Last Modification Date:	26.01.2003
Modifications:	- Digitized using *Mogan_91.tiff* and *1999_2.tiff*

Data Name:	Meadow_99.shp
Data Type:	Shape feature class
Data Geometry:	Polygon
Datum:	D_WGS_1984
Projection:	UTM

Geog. Coor. System: WGS_European_1984

Last Modification Date: 26.01.2003

Modifications: - Digitized using *Mogan_91.tiff* and *1991_2.tiff*

Data Name: Road_99.shp

Data Type: Shape feature class

Data Geometry: Line

Datum: D_WGS_1984

Projection: UTM

Geog. Coor. System: WGS_European_1984

Last Modification Date: 26.01.2003

Modifications: - Digitized using *Mogan_91.tiff* and *1991_2.tiff*