# MODELING THE WATER QUALITY IN ULUABAT LAKE

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# ABSTRACT

MODELING THE WATER QUALITY IN ULUABAT LAKE

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Lakes can be used for recreational purposes, agricultural irrigation, domestic water supply or industrial use. However, these functions of the lakes can be impaired due to excess nutrient loadings from industrial facilities, agricultural activities, and discharge of wastewaters from sewage systems. Uluabat Lake is one of the important lakes in Turkey faced with water quality problems due to excess nutrient loading from point and non-point sources. In this study, Water Quality Analysis Simulation Program (WASP Version 7.2), supported by the Environmental Protection Agency (EPA), was used to simulate the water quality of the lake. The model was calibrated using a set of data belonging to the time period from January 1$^{st}$ to June 1$^{st}$, 2000, for dissolved oxygen (DO), chlorophyll-a (Chl-a), ortho-phosphate ($PO_4$-P) and nitrate ($NO_3$-N). An acceptable similarity was obtained between the predicted and observed water quality values in calibration. Then, the model was verified for another data set. Verification period was from 1$^{st}$ of June to 31$^{st}$ of December, 1999. Verification results were also coherent with the observed values.

Following the calibration and verification, the model was used as a management tool to predict the future quality conditions for the lake for different management

scenarios. According to the results, sediment dredging had a significant impact on the water quality. Additional methods on top of sediment dredging improved the quality of the lake. The best result was obtained when buffer zone was formed, phosphorus loads from the Mustafakemalpaşa District sewage system and Emet and Orhaneli Watersheds were decreased by 50% and 50% decrease was considered in the fertilizer usage throughout the agricultural lands, in addition to sediment dredging.

Keywords: Calibration, modeling, Lake Uluabat, WASP7, water quality simulation

# ÖZ


ULUABAT GÖLÜ SU KALİTESİNİN MODELLENMESİ

YENİLMEZ, Firdes
Yüksek Lisans, Çevre Mühendisliği Bölümü
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Göller, rekreasyonel, zirai alanların sulanması, evsel ve endüstriyel su temini gibi pekçok amaç için kullanılabilir. Fakat günümüzde göller, endüstriyel tesislerden, zirai faaliyetlerden ve kanalizasyon sistemlerinden atıksuların deşarj edilmesinden dolayı aşırı organik madde girişine bağlı olarak fonksiyonlarını yitirmektedir.

Uluabat Gölü, noktasal ve noktasal olmayan kaynaklardan aşırı besin maddesi alması nedeniyle su kalite problemleri yaşayan ülkemizdeki en önemli göllerden biridir. Bu çalışmada, Uluabat Gölü'ndeki su kalitesi A.B.D. Çevre Koruma Ajansı (EPA) tarafından desteklenen Su Kalite Analiz Simülasyon Modeli (WASP7.2) kullanılarak modellenmiştir. Model, 01.01.2000-01.06.2000 zaman periyoduna ait veriler kullanılarak çözünmüş oksijen (ÇO), klorofil-a (Chl-a), orto-fosfat ($PO_4$-P) ve nitrat ($NO_3$-N), parametreleri için kalibre edilmiştir. Kalibrasyon sonucunda, model tahminleri ve gözlenen değerler arasında kabul edilebilir bir benzerlik elde edilmiştir. Daha sonra, simülasyon sonuçları başka bir veri seti ile doğrulanmıştır. Verifikasyon periyodu olarak 01.06.1999-31.12.1999 aralığı seçilmiştir. Verifikasyon sonuçlarının da gözlenen değerlerle uyum içinde olduğu görülmüştür.

Kalibrasyonu ve verifikasyonu tamamlanan model, bir yönetim aracı olarak, Uluabat Gölü'nün gelecekteki su kalitesinin ve çeşitli yönetim senaryolarının göl su kalitesi üzerine etkilerinin belirlenmesi amacıyla kullanılmıştır.

Yönetim senaryoları sonuçlarına göre, sediment uzaklaştırılması su kalitesi üzerinde önemli bir etkiye sahiptir. Sediment taramasına ek olarak uygulanan yöntemler de göl su kalitesini geliştirmiştir. En etkili sonuç, sediment uzaklaştırılmasına ek olarak Uluabat Gölü çevresindeki tarımsal alanların kaldırılıp, Mustafakemalpasa İlçesi kanalizasyon sistemi ve Emet ve Orhaneli Havzalarından gelen fosfor yükü, ve Uluabat Gölü havzasında yer alan tarımsal alanlarda kullanılan gübre miktarlarının %50 oranında azaltılması ile elde edilmiştir.

Anahtar Kelimeler: Kalibrasyon, modelleme, Uluabat Gölü, WASP7, su kalite simülasyonu

To My Mother

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# CHAPTER 1

# INTRODUCTION

The amount of freshwater on earth is scarce in comparison to oceans and seas. However, freshwater sources are threatened due to enrichment with nutrients which give rise to aquatic plant growth. This enrichment and proceeding DO depletion is called as eutrophication. Particularly, the main cause of eutrophication is the loading of the water body with nitrogen and phosphorus that are mainly released in the watershed from agricultural fertilization, sewage and industrial wastes, and enrichment via atmospheric pollutants (especially nitrate) (Wetzel 1983, Hein 2006). Eutrophication often leads to a decline in the supply of ecosystem services such as recreation, fisheries, or nature conservation in and around the water body (Hein 2006).

The response of a shallow lake to eutrophication is generally different from the more often-studied deeper lakes. Some features of these lakes are a lack of stable long-term thermal stratification, frequent mixing of the entire water column, resuspension of unconsolidated sediments and substantial internal loading of nutrients from the sediments to water column. Water quality conditions in the mentioned lakes may have complex relationships with external nutrient loads and in-lake processing (Havens et. al, 2001).

Lake Uluabat is a large shallow freshwater lake in Bursa. The lake has great importance due to its rich biodiversity, its location on the migratory bird route, and its vast areas of suitable habitats for many bird species. Due to its ecological significance, it was included in the RAMSAR List in 1998. The lake was considered as a potential freshwater source for Bursa nearly 30 years ago. However, biodiversity has decreased in recent years because of the water quality

problems in the lake (DHKD, 2002). It has become eutrophic due to point and non-point pollution sources around the lake.

In this study, the water quality in Lake Uluabat is investigated. In the first stage of the study, the current status in the lake is analyzed using in-situ measurements, field sampling and laboratory analysis for parameters such as DO, $PO_4$-P, $NO_3$-N, etc. These values are compared to the ones obtained in previous studies in order to determine the progression of the water quality.

Water quality models are useful tools that are widely used in the world to understand and predict the water quality responses to natural phenomena, man made pollution, and a variety of pollution management decisions (Wool, T. *et al*, 2001). In the second stage of the study, water quality of the lake is modeled using the Water Quality Analysis Simulation Program (WASP Version 7.2), supported by US EPA. For this purpose, the model is calibrated using a set of data for DO, Chl-a, $PO_4$ and $NO_3$. Then, the model is verified for another data set. The calibrated and verified model is used as a management tool to predict the future quality conditions for different abatement scenarios.

# CHAPTER 2

# BACKGROUND

## 2.1. Eutrophication

Eutrophication is one of the most important water quality problems in lakes and reservoirs in the world (Kuo et al., 2006). Eutrophication is the process of nutrient enrichment of a water body due to an increase in nutrient loading. The most important nutrients that cause eutrophication are phosphates, nitrates and ammonia that are released mainly in the watershed from agricultural or urban sources. Increased nutrients raise excessive plant growth such as algae and periphyton attached algae (Horne and Goldman, 1994; Hein, 2006). This enhanced plant growth depletes dissolved oxygen in the water when dead plant material decomposes and can cause other organisms to die (USGS, 2006).

Nutrients come from both point sources and non-point sources. Phosphorus comes from wastewater treatment effluent, agricultural land and discharge of raw sewage including that from stock and fish-farm animals (Moss, 1998). Nitrogen originates from excretal sources, fertilizers applied to agricultural fields, deposition of nitrogen from atmosphere, oxidation of nitrogen oxides produced by vehicle engines, burning of vegetation to clear land for agriculture (Moss, 1998; USGS, 2006). It is more difficult to monitor and control non-point sources of nutrients than point sources due to scattering over wide areas of land and change with time because of weather and other conditions (Bulut, 2005; Smith, 1999).

Lakes are classified according to trophic states. The trophic state of lakes indicates their biological productivity which is the amount of living material within lakes (EPA, 2006). Oligotrophic lakes are characterized by relatively low

nutrient concentrations, clear blue water and high secchi disk transparencies. They are generally deep, with steep sides and have relatively small drainage areas (Horne and Goldman, 1994). Eutrophic lakes are often shallow, with gradually sloping edges and have large drainage areas. The most characteristic features of these lakes are high nutrient concentrations, abundance of planktonic or attached algae, presences of blue-green algae blooms on the water surface, variable oxygen concentrations, low water clarity and low water transparency. Mesotrophic lakes are intermediate between oligotrophic and eutrophic lakes (Horne and Goldman, 1994). General trophic classification of lakes and reservoirs in relation to concentrations of phosphorus, nitrogen, phytoplankton pigment, Chl-a, and water transparency are summarized in Table 2.1.

**Table 2. 1** General Trophic Classification of Lakes and Reservoirs in Relation to Phosphorus and Nitrogen[a] (Wetzel, 2001)

| Parameter (annual mean values) | | Oligotrophic | Mesotrophic | Eutrophic | Hyper-eutrophic |
|---|---|---|---|---|---|
| Total Phosphorus ($mg\ m^{-3}$) | Mean | 8.0 | 26.7 | 84.4 | - |
| | Range | 3.0-17.7 | 10.9-95.6 | 16-386 | 750-1200 |
| N | | 21 | 19 | 71 | 2 |
| Total Nitrogen ($mg\ m^{-3}$) | Mean | 661 | 753 | 1875 | - |
| | Range | 307-1630 | 361-1387 | 393-6100 | - |
| N | | 11 | 8 | 37 | - |
| Chlorophyll a ($mg\ m^{-3}$) of phytoplankton | Mean | 1.7 | 4.7 | 14.3 | - |
| | Range | 0.3-4.5 | 3-11 | 3-78 | 100-150 |
| N | | 22 | 16 | 70 | 2 |
| Chlorophyll a maxima ($mg\ m^{-3}$) (worst case) | Mean | 4.2 | 16.1 | 42.6 | - |
| | Range | 1.3-10.6 | 4.9-49.5 | 9.5-275 | - |
| N | | 16 | 12 | 46 | - |
| Secchi transparency depth (m) | Mean | 9.9 | 4.2 | 2.45 | - |
| | Range | 5.4-28.3 | 1.5-8.1 | 0.8-7.0 | 0.4-0.5 |
| N | | 13 | 20 | 70 | 2 |

[a]Based on data of an international eutrophication program. Trophic status based on the opinions of the experienced investigators. (Modified from Vollenweider, 1979)

N = number of cases considered

## 2.2. Lake Uluabat

Lake Uluabat is a large lake located at south of Marmara Sea. It is about 30 km away from Bursa. The geographic coordinates of the lake are 40°10'N, 28°35'E (Salihoğlu and Karaer, 2004). It is surrounded by Karaağaç-Kirmikir at the north, Mustafakemalpaşa at the west, Akçalar Plain at the east, Çamlıtepe and Hisartepe Mountains at the south (Celtemen et al., 2000). The Lake is fed mainly by the Mustafakemalpaşa Stream at southwest (Salihoğlu and Karaer, 2004). Orhaneli and Emet Streams combine and form the Mustafakemalpaşa Stream. After flowing in a channel of about 45 km long, this stream drains into the lake. Uluabat Stream forms the outlet of the lake in the northwest. This stream drains into Karadere, flows through Karacabey passage and finally discharges to the Marmara Sea (Bulut, 2005).

The average depth of the lake is 3 m. However, average depth decreases to 0.8–1 m in the summer months (Salihoğlu and Karaer, 2004). Bathymetry map of the lake is illustrated in Figure 2.1. The length of the lake from east to west is approximately 24 km and that from south to north is approximately 12 km (Karacaoğlu et al., 2004). Surface area of the lake changes seasonally. The surface area of the lake is between 135 and 155 $km^2$, depending on the water level. Previous studies showed that excluding the Mustafakemalpaşa Stream Delta, the lake itself covered an area of 133.1 $km^2$ in 1984, 120.5 $km^2$ in 1993, and 116.8 $km^2$ in 1998. Therefore, during this time frame, the coverage area of the lake has decreased at a rate of about 12% mainly due to sediments transported by the surface runoff from the surrounding irrigated agricultural lands, tributary streams and particularly the Mustafakemalpasa Stream. In addition, in a 14-year period, agricultural lands in the delta have been expanded at a rate of 26% from 2508 ha to 3393 ha in the delta of the Mustafakemalpaşa Stream (Salihoğlu and Karaer, 2004; Aksoy and Özsoy, 2002).

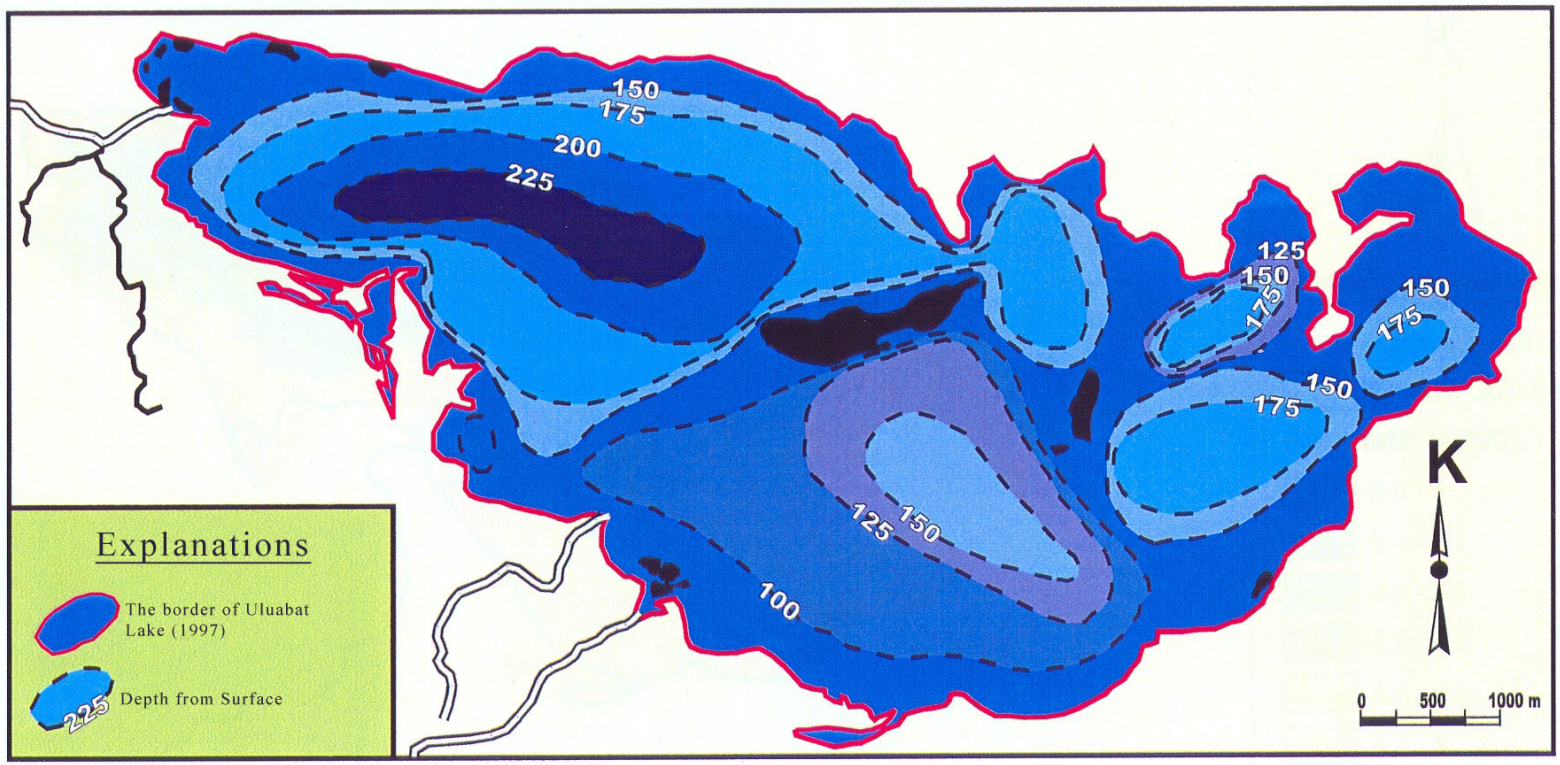


**Figure 2. 1** Bathymetry map of Uluabat Lake belonging to 1997 (Kurttaş et al., 2002)

Uluabat Lake has the largest water lily beds in Turkey. It is located on the route of migratory birds. Therefore, it is an important feeding and shelter area for them. These vast areas of suitable habitats for many bird species are particularly located at the north-eastern shores and in the Mustafakemalpaşa Stream Delta. The lake has great importance due to its rich biodiversity. Plants, plankton, and small aquatic fauna in the lake play a very important role for various species of wildlife and fisheries (Salihoğlu and Karaer, 2004).

The lake exhibits a typical eutrophic character. Sewage water, agricultural fertilizers and chemicals, animal waste, process wastes of food industries, tanneries, slaughterhouses, and mining wastes from the watersheds of Emet and Orhaneli Streams find way to the lake and contribute to the eutrophication problem. Majority of the nitrogen and phosphorus load to the lake is carried via Mustafakemalpasa Stream (Salihoğlu and Karaer, 2004; Bulut, 2005; Bulut and Aksoy, 2006). Figure 2.2 shows Lake Uluabat Watershed and Subbasins delineated by SWAT (Bulut, 2005).

Uluabat Lake and its surrounding area were included in the RAMSAR List in 1998 by the Turkish Ministry of Environment. Therefore, the lake gained a new status both at the national and the international level (Salihoğlu and Karaer, 2004). The protection of the Lake has been undertaken at the international level. Figure 2.3 shows the Lake Uluabat and the boundary set by the RAMSAR Agreement. The red dashed line indicates the RAMSAR boundary.

Global protection status of the lake required the development of an environmental management plan and received more attention from the Ministry of Environment and certain NGOs such as the like Natural Life Protection Association (Doğal Hayatı Koruma Derneği - DHKD). DHKD started a joint project with the Ministry of Environment and the State Hydraulic Works to prepare a management plan for Uluabat Lake in 1998. Background data was gathered by conducting several studies on biological diversity, water quality, and

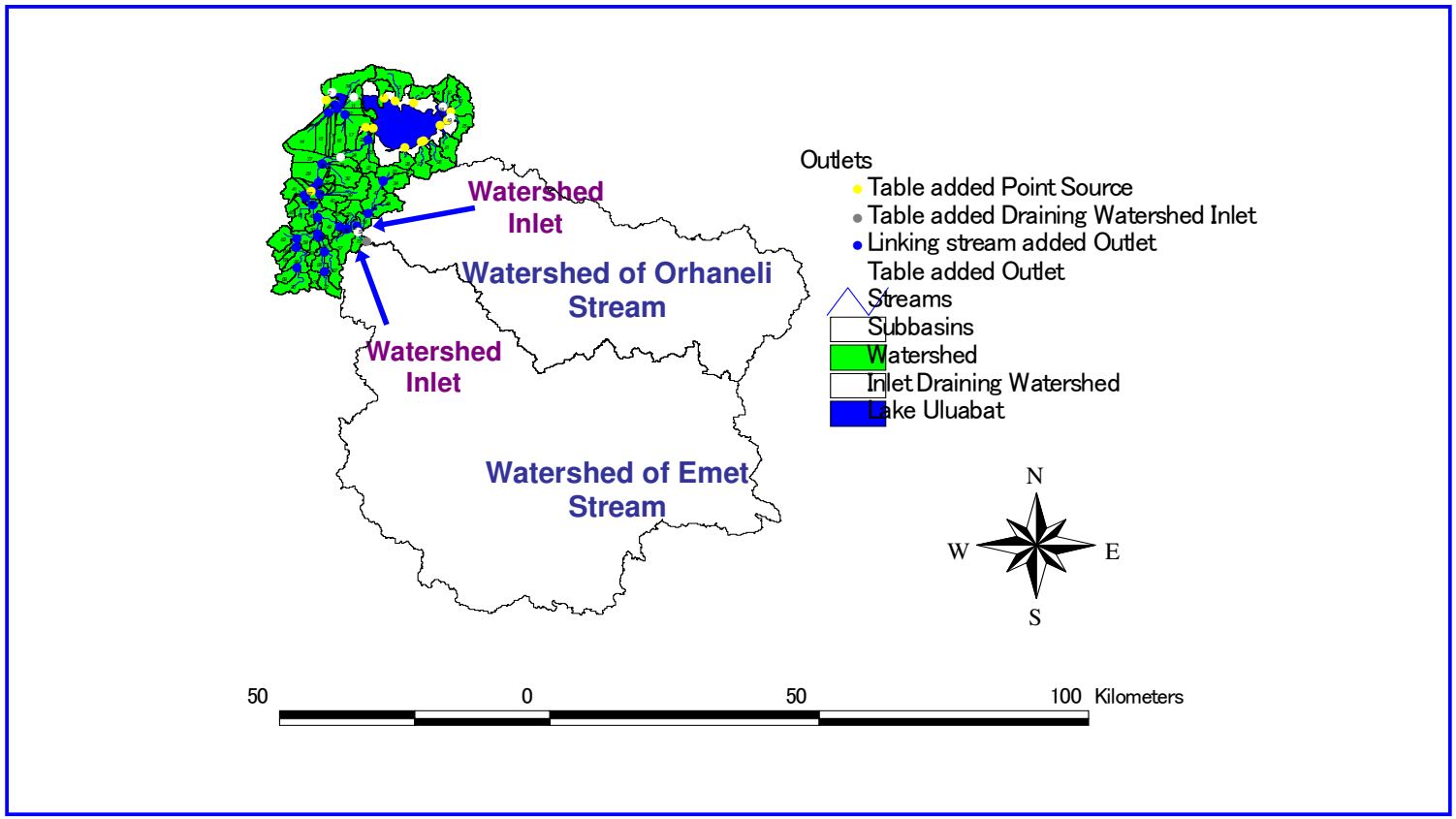


**Figure 2.2** Lake Uluabat Watershed and Subbasins Delineated by SWAT (Bulut and Aksoy, 2006)

socioeconomic conditions. Suggestions were made for the protection of the lake. However, they are not fully implemented.

**Figure 2. 3** Lake Uluabat and Boundary Set by the RAMSAR Agreement (depicted by red dashed line) (Ministry of Environment and Forestry, 2007)

The lake was considered as a potential freshwater resource for Bursa City nearly 30 years ago. The State Hydraulic Works Water Affairs executed water quality monitoring until the quality of the lake had deteriorated to a level prohibiting its use as a freshwater resource. The water quality of the lake was reported to be in the first or second class according to the classes given in the Turkish Water Pollution Control Regulations in 1970s. However, the quantities for 1999–2000 indicated that the quality downgraded to second, third, and even fourth (worst quality) class for several parameters. Therefore, there is need for further effort for the protection of the lake which would require the contributions of all stakeholders for a realistic and applicable management plan (Salihoğlu and Karaer, 2004).

Several studies conducted related to Uluabat Lake in the last decade. One of them is the "Uluabat Lake Mustafakemalpaşa Stream System Sediment Accumulation Research Sub-Project" (Kurttaş et al., 2002). In the study, the focus was on the sediment transport to the lake. It was concluded that Mustafakemalpaşa Stream was the main contributor of sediment load to the lake.

Salihoğlu and Karaer (2004) performed ecological risk assessment and problem formulation for Lake Uluabat. This assessment contributed to the basic elements of an environmental management plan such as the partnership-building process, prioritization of the problems and issues of the ecosystem and development of the action plan. Ecosystem risks were ranked and an action plan was formed. The results revealed that SS was the stressor that had the most negative impact on the integrity of the Uluabat Lake ecosystem. Therefore, it was suggested to focus on SS reduction. Other important parameters that should be controlled to improve the lake quality were the chemical substances, nutrients and stress due to fishing activities.

Akdeniz (2005) conducted a study to investigate the water quality status of the lake. For that purpose, parameters such as TN, TP, DO, Chl-a, $PO_4$-P, $NO_3$-N were measured. The relationships between these parameters and the seasonal changes in each of them were analyzed. This study concluded that nutrient concentrations in Uluabat Lake were considerably higher than the limit values. According to the results, trophic level of the lake changed between eutrophic and hypereutrophic levels for different parameters. It was suggested to minimize the external loads and apply holistic solutions to accelerate the recovery of the lake. Their recommendation was to employ biomanipulation.

Bulut (2005) assessed phosphorus loads and management practices to control phosphorus transport to Lake Uluabat using the watershed scale modeling approach. AVSWAT was utilized for this purpose. This study drew a conclusion that agricultural sites were among the major contributors of

phosphorus loads to Lake Uluabat. In addition, point sources in Emet and Orhaneli Watersheds significantly contributed to the total phosphorus load. It was suggested to control the agricultural activities in the Mustafakemalpaşa Stream Basin and the point sources in the Emet and Orhaneli Watersheds in order to reduce the phosphorus loads to lake. Furthermore, this study pointed out that a management strategy in a watershed scale is necessary for the protection of the lake.

Celtemen et al. (2000) examined the water quality of Uluabat Lake in June 1999-February 2000. They have measured $PO_4$, $NH_3$, $NO_3$, Chl-a, SS, Turbidity, COD, Boron and Fe at eleven stations. The average concentrations for $PO_4$ and $NO_3$ were compared with the quantities for the 1983-1986 and 1987-1996 periods. The results indicated increase in $PO_4$ with time. However, decline in $NO_3$ concentration was observed compared to the previous years.

### 2.3. General Description of the Model

The Water Quality Analysis Simulation Program Version 7.2, WASP7.2, is an improvement of the original WASP supported by US EPA Centre for Exposure Assessment Modeling (CEAM) in Athens, Georgia (Di Toro et al., 1983; Connolly and Winfield, 1984; Ambrose et al., 1988). Model supplies a tool for understanding and predicting the water quality responses to natural phenomena, man made pollution, and a variety of pollution management decisions. WASP7.2 is a dynamic model that can be used to analyze a variety of water quality problems in diverse water bodies such as ponds, streams, lakes, reservoirs, estuaries, and coastal waters, including both the water column and the underlying benthos (Wool, *et al*, 2001).

The WASP7 system consists of two separate computer programs, DYNHYD7 and WASP7.2 that can be run in conjunction or separately. The hydrodynamics program, DYNHYD7, simulates the movement of water and velocities in the

lake. The water quality program, WASP7.2, simulates the movement and interaction of pollutants within the water.

WASP7.2 has a unique flexibility. It allows the modeler to structure one, two, and three dimensional models. In addition, time variable exchange coefficients, advective flows, waste loads and water quality boundary conditions can be specified (Wool *et al*, 2001).

Water quality processes are represented in special kinetic subroutines that are either chosen from a library or written by the user. WASP permits easy placement or removal of kinetic subroutines into the overall package to form problem specific models. WASP7.2 consist of two sub models; TOXI7 for the simulation of toxicants involving organic chemicals, metals, and sediment interactions, and EUTRO7 for the simulation of conventional water quality problems involving DO, biochemical oxygen demand, nutrients and eutrophication (Wool, *et al*, 2001).

The fundamental equations solved by the hydrodynamics and water quality program are based on the conservation of mass. The water volume and water quality constituent being studied are tracked and accounted for over time and space using a series of mass balance equations. The hydrodynamics program also considers momentum, or energy conservation throughout time and space (Wool, *et al*, 2001).

A mass balance equation for dissolved constituents in a body of water must include all the material entering and leaving through direct and diffuse loading, advective and dispersive transport, and physical, chemical, and biological transformations. In Equation 1, the general mass balance equation around an infinitesimally small fluid volume for a three-dimensional system is given where x and y coordinates are in the horizontal plane, and the z coordinate is in the vertical plane:

$$\frac{\partial C}{\partial t} = -\frac{\partial}{\partial x}(U_x C) - \frac{\partial}{\partial y}(U_y C) - \frac{\partial}{\partial z}(U_z C) + \frac{\partial}{\partial x}(E_x \frac{\partial C}{\partial x}) + \frac{\partial}{\partial y}(E_y \frac{\partial C}{\partial y}) + \frac{\partial}{\partial z}(E_z \frac{\partial C}{\partial z}) + S_L + S_B + S_K \quad (2.1)$$

where;

$C$ = concentration of the water quality constituent (mg/L or g/m$^3$),

$t$ = time (days)

$U_x$, $U_y$, $U_z$ = longitudinal, lateral, and vertical advective velocities, respectively (m/day),

*Ex, Ey, Ez* = longitudinal, lateral, and vertical diffusion coefficients, respectively (m$^2$/day),

$S_L$ = direct and diffuse loading rate (g/m$^3$-day),

$S_B$ = boundary loading rate including upstream, downstream, benthic, and atmospheric (g/m$^3$-day),

$S_K$ = total kinetic transformation rate; (+ is for source, - is for sink) (g/m$^3$-day)

By increasing the infinitesimally small control volumes into larger adjoining segments, and by specifying appropriate transport, loading, and transformation parameters, WASP implements a finite-difference form of Equation 1.

In order to model a system, the user must provide WASP7 with input data classified in ten important groups. These are:

A- Model identification and simulation control

B- Exchange coefficients

C- Volumes

D- Flows

E- Boundary concentrations

F- Waste loads

G- Environmental parameters

H- Chemical constants

I- Time functions

J- Initial conditions.

Data group A provides information for descriptive model identification and simulation control options. The user must specify the number of segments and number of systems in this data set. Data group B contains dispersive exchange information. Dispersion occurs between segments and along a characteristic length. Data group C supplies initial segment volume information, and information on the segment type and underlying segment numbers. Hydraulic geometry can be given to derive segment average depth and velocity as a function of flow. Data group D employs flow and sediment transport information between segments. Flows may be contained in the WASP input data set, or may be imported from an external hydrodynamic file. Data group E contains concentrations for each system at the boundaries. All system concentrations must be supplied for each boundary. Boundary concentrations vary with time in a piecewise linear time function. Data group F defines the point and diffuse waste loads and segments that receive them. Data group G contains environmental characteristics of the water body. These parameters are spatially variable and varying with each model segment. Data group H contains chemical characteristics or constants of the system. Constants used in the model remain constant in both time and space. Data group I contains relevant environmental or kinetic time functions. Data group J stores the initial concentrations for each segment and each system, along with dissolved fractions and the density of solids systems (Ambrose et al., 1993). Using these inputs, the mass balance equations for each constituent in the system is numerically integrated over time as simulation proceeds.

The results given by the model are saved for display. The output data is retrieved by the post processor programs W7DSPLY and W7PLOT at user

specified print intervals. These programs enable the user to interactively produce graphs and tables for all variables (Wool *et al*, 2001).

The model network, presented by a set of segments, symbolizes the physical configuration of the water body. The network can divide the water body laterally, vertically and longitudinally as depicted in Figure 2.4. Benthic segments can be integrated with water column segments as well. WASP segments are of four types. These are the epilimnion (surface water), hypolimnion (subsurface), upper benthic, and lower benthic layers. The segment type plays a significant role in bed sedimentation and in certain transformation processes. The arrangement of underlying segments is important when light has to pass from one segment to the next in the water column, or when material is buried or eroded in the bed.

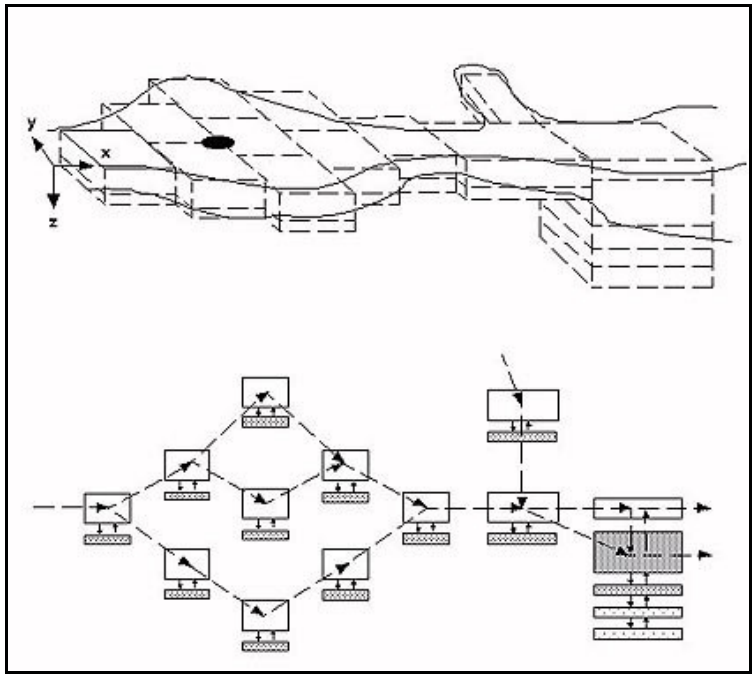


**Figure 2. 4** Segmentation in WASP

Segment sizes are determined more by the spatial and temporal scale of the problem being analyzed than by the characteristics of the water body or the pollutant. The important spatial characteristics within a segment must be homogeneous. This constraint sometimes can be relaxed by sensible averaging over width, depth, and length. The segment sizes are also affected by the usual spatial variability of the water quality concentrations (Wool *et al*, 2001).

When the network is set up, the model study will continue through four steps including hydrodynamics, mass transport, water quality transformations, and environmental toxicology. Hydrodynamics explain the question of where the water goes by using special studies and hydrodynamic modeling. Flow data can be interpolated or extrapolated using the principle of continuity. Flow data can be appropriately averaged over time and space by using very simple flow routing models and very complicated multi dimensional hydrodynamic models (ODU, 2006). In cases where adequate information is lacking for the hydrodynamic model, WASP7.2 can utilize the water balance for calculations.

Earlier versions of WASP have been widely used all around the world to examine water quality related problems. Tufford and McKellar (1999) developed a two-dimensional water quality model of Lake Marion, which had a surface area of 330.7 $km^2$ and a mean depth of 4 m, using the WASP5 modeling system. The lake was divided into 31 segments. The model was parameterized using the field data gathered in 1985 to 1990. The water quality model was calibrated for ammonia ($NH_3$), $NO_3$, $PO_4$, DO, Chl-a, biochemical oxygen demand (BOD), organic nitrogen, and organic phosphorus. Following the calibration, the model was validated by re-parameterizing the nutrient loading functions for an independent set of field data. Sensitivity analysis for the phytoplankton kinetic rates suggested that site-specific estimates were important for obtaining fit to field data. Sediment sources of $NH_3$ (10-60 mg $m^{-2}$ $day^{-1}$) and $PO_4$ (1-6 mg $m^{-2}$ $day^{-1}$) were important to achieve model calibration, especially during periods of high temperatures and low DO. The sediment flux accounted

for 78% and 50% of the annual nitrogen and phosphorus loads. According to the conclusion of the study, the implemented WASP5 model did not fully accommodate the ecological variability in the lake due to constraints on the specification of rate constants. However, it was successful in pointing out several issues about the ecology of Lake Marion. In overall, the model was both versatile and useful as a research tool at least by showing where knowledge was lacking or the specific locations that are of interest.

Fitzpatrick and Di Toro (1999) conducted a modeling study on the eutrophication in Lake Erie using WASP. This study concluded that observations were consistent with WASP predictions for a variety of parameters and model could accurately predict the water quality parameter concentrations of interest.

Lung and Larson (1995) modeled the water quality of Upper Mississippi River and Lake Pepin using WASP5 to address the cause and effect relationship between the Metropolitan Wastewater Treatment Plant phosphorus load and eutrophication in Lake Pepin. The lake had a surface area of 101.6 $km^2$ and had a mean depth of 5.4 m. The model was calibrated and verified for the summer average conditions of 1988, 1990 and 1991. Projections of water quality impact in Lake Pepin were conducted under various phosphorus load reduction scenarios for the Metropolitan Wastewater Treatment Plant. The model results in general matched with the field data closely and demonstrated that the Upper Mississippi River, including Lake Pepin, contained abundance of nutrients. It was concluded that basin-wide reductions were needed before any long-term improvements could be achieved.

Himesh et al. (2000) modeled the water quality of Tapi estuary, located in a highly industrialized zone, for different state variables (BOD, TKN, DO) using WASP3. The model was calibrated and validated against measured field data collected at different time intervals. Results of the study showed that the performance of the model was excellent for the hydrodynamic compartment of

the model and fair for the process compartment of the model. WASP3 was effective for short-term predictions of water quality in the estuary influenced by tidal effects and salinity. It was also suggested to use the calibrated model for different sites with similar environmental conditions (boundary conditions, flow regime etc.) without going for expensive and time consuming exercise of calibration for that site.

Chao *et al*. (2005) developed a three-dimensional numerical model to simulate the concentrations of water quality constituents in shallow Oxbow lakes where sediment-related processes are important. The model scheme generally followed WASP6 (Wool et al., 2001). In this model, the effect of sediment on the growth of phytoplankton was analyzed. The processes of adsorption-desorption and bed releases of nutrients from the sediment layer were simulated. The model was verified using the analytical solutions of pollutant transport in an open channel flow. The agreement between the computed and theoretical solutions was excellent. The model was successfully applied to simulate the concentration of water quality constituents in Deep Hollow Lake which had a water depth ranging from 0.5 m to 2.6 m. Trends and magnitudes of nutrient and phytoplankton concentrations obtained from the numerical model generally agreed with the field observations.

Tetra Tech prepared the Jordan Lake Nutrient Response Model in 2002. The model was a linked system that relied on Environmental Fluid Dynamics Code (EFDC) and WASP (Ambrose et al., 1993) simulations. Hydrodynamic output and temperature simulation generated by EFDC were input to WASP to account for the dynamic processes of lake mixing, seasonal changes, and nutrient cycling. The model was developed and calibrated based primarily on the data collected in 1997 to 2000 and accurately predicted the flow of water, lake temperature profiles, pollutant concentrations, and algal response. Then, 2001 observations were used for validation. A second round of recalibration was undertaken for 2003 with the intention of obtaining the best fit to both the 1997-

1999 and 2001 data. Results showed that, despite some inaccuracies in the simulation, the model continued to provide a good representation of nutrient response in the Jordan Lake, and was an appropriate tool to analyze the nutrient reduction scenarios. It was concluded that the inorganic nitrogen was the limiting nutrient in the upper portion of the lake while both nitrogen and phosphorus might limit the algal growth in most of the remainder of the lake (Tetra Tech, 2003).

James et al. (1997) studied Lake Okeechobee, a large and shallow lake with a surface area of 1732 $km^2$ and a mean depth of 2.7 m, by modifying the WASP model to include the processes that are important for the system. In this lake, the wind driven re-suspension of the sediment increased the phosphorus and nitrogen concentrations in the water column as well as increasing the light attenuation. The model calibration and validation were successful with the exception for the dissolved inorganic nitrogen species. This was attributed to the inaccurate algal nitrogen preference and the absence of nitrogen fixation in the model. The model correctly predicted that the light was limited due to resuspended solids, and algal growth was primarily nitrogen limited. The addition of sediment resuspension to the WASP model enhanced the accuracy of nutrient and Chl-a predictions for Lake Okeechobee.

Nikolaidis *et al*. (2006) conducted a nutrient modeling study on the Thermaikos Gulf. Thermaikos Gulf was considered to be one of the most polluted coastal zones in Greece. It acted as the final receptor of both municipal and industrial wastewaters of Thessaloniki and two heavily polluted rivers, Axios and Aliakmon. Nutrient enrichment and consequent eutrophication was the primary water quality issue of Thermaikos. WASP6 was used to simulate the hydrodynamics, nutrient dynamics and phytoplakton evolution in the gulf. The gulf was segmented into 12 compartments consisting of two layers each consisting of six horizontal segments. Monthly hydrologic and water quality data were used to calibrate the dynamics of nitrogen and phosphorus species and Chl-

a. Model calibration was conducted by minimizing the root mean square error between the field observations and model simulations. The model was able to capture the variation in salinity and nutrient concentrations. The results showed the importance of Thessaloniki effluents and the loads from the Axios and Aliakmon Rivers in the eutrophication of the entire gulf. Load sensitivity analysis showed that a 10% reduction in the nutrient concentrations in Thessaloniki Bay and Gulf could be achieved by decreasing the nutrient loads from the City of Thessaloniki by 30%. Moreover, a reduction of 40% in the nutrient loads from the Axios River would result in a significant reduction in the nutrient concentrations (up to 38% in $NO_3$ and 20% in $PO_4$) in the Thermaikos Gulf. The model was successful in assessing the management scenarios to improve the water quality of the gulf.

Warwick *et al*. (1999) modeled the impacts of regional groundwater quality and local agricultural activities on the water quality of the Lower Truckee River in Nevada. For this purpose, a detailed program of monitoring and computer simulation was performed. Agricultural diversion and return-flow were monitored in great detail to determine the mass loading rates of nutrients from agricultural areas. The cumulative impacts of agricultural diversions and return-flows were evaluated using WASP through simulating nitrogen and phosphorus species, periphyton, and DO. It was concluded that a significant portion of the water diverted for agricultural use returned to the river as a surface point return-flow (13.9±0.1%), and as a groundwater diffuse return-flow (27±6 %). Modeling results demonstrated the significant effect of regional groundwater quality ($NO_3$) on the predicted periphyton growth and associated diel fluctuations in DO.

Pickett conducted a Total Maximum Daily Load (TMDL) study for the Black River in 1997. This river suffered from low DO, anoxia in some locations, and fish kills. Environmental data was collected during two summer dry seasons and simulations were conducted using WASP5 to model the effect of BOD, $NH_3$ and

nutrient loads on DO. After calibration and verification of the model, anthropogenic pollutant sources were removed. The results indicated that even in the absence of anthropogenic pollutant loading, DO fell below the 8.0 mg/L regulatory criterion. A TMDL was proposed for BOD and $NH_3$ to prevent the significant degradation of DO in the Black River. In addition, for the phosphorus limited middle river section, a TMDL for total phosphorus was proposed that established a protective criterion of 0.05 mg/L. To meet this level, it was suggested to limit the TP discharge of the aquaculture facility and eliminate the dairy facility discharges.

As given in the above study, several TMDL studies were conducted using WASP. Examples for these are the amendment of the 2001 Murderkill River TMDLs (2004), proposed TMDL for nutrients, BOD and DO for the Caloosahatchee River Basin (2005), TMDLs for DO, nutrients, turbidity, and TSS for the Peace River Basin, Florida (Giattine, 2006).

Mahajan *et al.* (1999) investigated the application of WASP3 to determine the optimum treatment level required for the municipal wastewater so that the discharge into the Mumbai coastal waters through Thana and Malad Creeks would not violate the coastal water quality criteria. Analysis was performed with respect to BOD and DO values. Results indicated that the improvement in the Thana Creek water quality due to the treatment of sewage was insignificant. This was largely due to the dilution available in the Creek and the small amount of discharge. The improvement in absolute DO values for the Malad Creek was more pronounced. This was attributed to the large volumes of sewage that was discharged into the Malad Creek. The study concluded that, unless timely preventive and remedial measures were taken, the proposed treatment of sewage alone was unlikely to produce the desired results in terms of improvement in the water quality of the creek.

### 2.3.1. EUTRO

Eutrophication, nutrient enrichment and DO depletion processes are simulated using the EUTRO program which is a sub-model of WASP. The transport and interactions among nutrients, phytoplankton, carbonaceous material, and DO in the aquatic environment can be affected by several physical-chemical processes. Figure 2.5 illustrates the principal kinetic interactions for the nutrient cycles and DO.

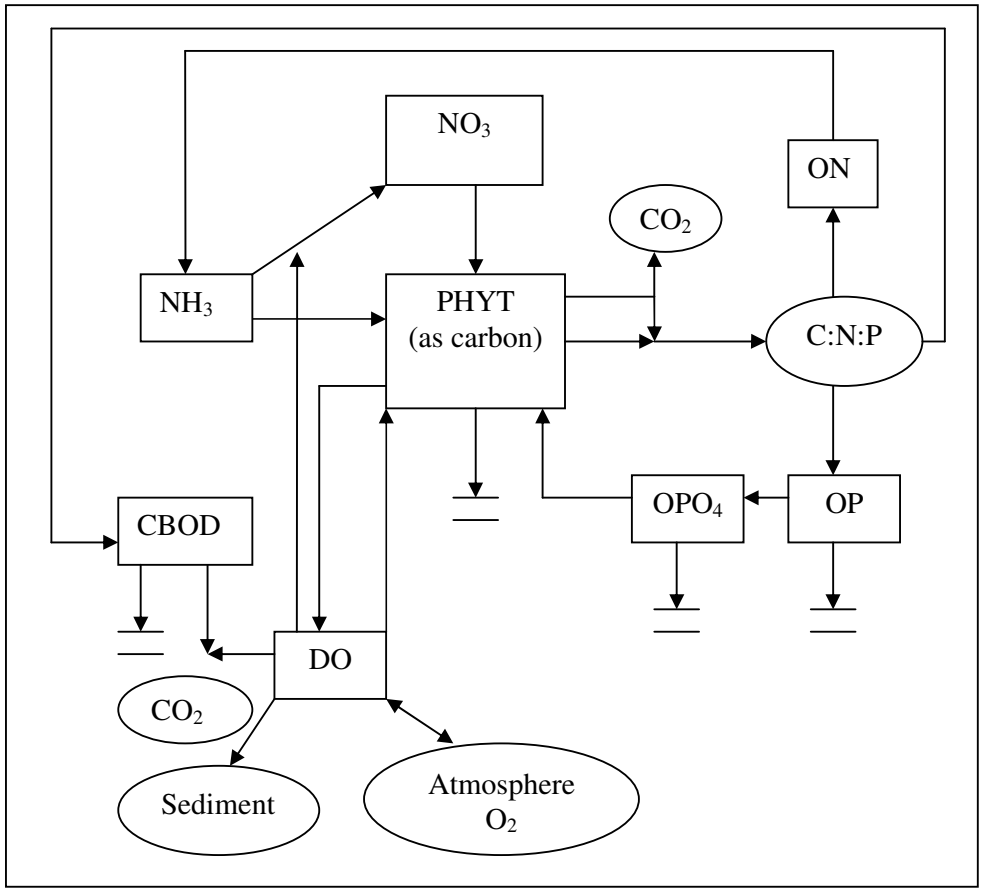


**Figure 2.5** EUTRO State Variable Interactions

EUTRO simulates the transport and transformation reactions of up to eight state variables ($NH_3$, $NO_3$, $PO_4$, Chl-a, CBOD, DO, organic nitrogen, organic phosphorus), as illustrated in Figure 2.3. These variables are considered in four interacting systems; the phytoplankton kinetics, the phosphorus cycle, the nitrogen cycle, and the DO balance. Short summary of the material cycles are given below (Wool *et al*, 2001).

#### 2.3.1.1. Phosphorus Cycle

Dissolved or available inorganic phosphorus (DIP) interacts with particulate inorganic phosphorus by means of the sorption-desorption mechanism. DIP is taken up by the phytoplankton for growth, and incorporated into the phytoplankton biomass. Phosphorus is returned from the phytoplankton biomass pool to dissolved and particulate organic phosphorus and to DIP through endogenous respiration and nonpredatory mortality. Organic phosphorus is converted to DIP at a temperature-dependent mineralization rate. Mass balance equations for phytoplankton, organic and inorganic phosphorus are given in Equations 2.2, 2.3, and 2.4 in one dimension, respectively (Wool *et al*, 2001).

$$\frac{d(C_4 a_{pc})}{dt} = G_{P1} a_{pc} C_4 - D_{P1} a_{pc} C_4 - \frac{v_{s4}}{D} a_{pc} C_4 \tag{2.2}$$

$$\frac{dC_8}{dt} = D_{P1} a_{pc} f_{op} C_4 - k_{83} \Theta_{83}^{(T-20)} \left( \frac{C_4}{K_{mPc} + C_4} \right) C_8 - \frac{v_{s3}(1 - f_{D8})}{D} C_8 \tag{2.3}$$

$$\frac{dC_3}{dt} = D_{P1} a_{pc} (1 - f_{op}) C_4 + k_{83} \Theta_{83}^{(T-20)} \left( \frac{C_4}{K_{mPc} + C_4} \right) C_8 - G_{P1} a_{pc} C_4 \tag{2.4}$$

where;

$C_3$ = Inorganic phosphorus ($PO_4$) (mgP/L)

$C_4$ = Phytoplankton carbon (PHYT) (mgC/L)

$C_8$ = Organic phosphorus (OP) (mgP/L)

$G_{P1}$ = Specific phytoplankton growth rate ($day^{-1}$)

$D_{P1}$ = Phytoplankton loss rate ($day^{-1}$)

$a_{PC}$ = Phosphorus to carbon ratio (mgP/mgC)

$k_{83}$ = Dissolved organic phosphorus mineralization rate at 20 $^oC$ ($day^{-1}$)

$\Theta_{83}$ = Temperature correction coefficient for $k_{83}$ (-)

$K_{mPc}$ = Half saturation constant for phytoplankton limitation of phosphorus recycle (mgC/L)

$f_{op}$ = Fraction of dead and respired phytoplankton recycled to the organic phosphorus pool (-)

$1- f_{op}$ = Fraction of dead and respired phytoplankton recycled to the phosphate phosphorus pool (-)

$f_{D8}$ = Fraction dissolved organic phosphorus in the water column (-)

$f_{D3}$ = Fraction dissolved inorganic phosphorus in the water column (-)

$V_{s3}$ = Organic matter settling velocity (m/day)

$V_{s4}$ = Phytoplankton settling velocity (m/day)

D = Depth of segment, equal to volume/surface area (m)

#### 2.3.1.2. Nitrogen Cycle

The kinetics of the nitrogen species is fundamentally the same as the phosphorus system. $NH_3$ and $NO_3$ are taken up by phytoplankton for growth and incorporated into phytoplankton biomass. The rate at which each is taken up is a function of the relative concentration to the total inorganic nitrogen ($NH_3$+$NO_3$) available. Nitrogen is returned to the system from the phytoplankton biomass to dissolved and particulate organic nitrogen and to ammonia through endogenous respiration and nonpredatory mortality. Organic nitrogen is converted to $NH_3$ at a temperature-dependent mineralization rate, and $NH_3$ is then converted to $NO_3$ at a temperature- and oxygen-dependent nitrification rate. $NO_3$ may be converted to nitrogen gas ($N_2$) in the absence of oxygen at a temperature- and oxygen-

dependent denitrification rate. Mass balance equations for phytoplankton, organic, $NH_3$ and $NO_3$ nitrogen species are given in Equations 2.5, 2.6, 2.7, and 2.8, respectively (Wool *et al*, 2001).

$$\frac{d(C_4 a_{nc})}{dt} = G_{P1} a_{nc} C_4 - D_{P1} a_{nc} C_4 - \frac{v_{s4}}{D} a_{nc} C_4 \tag{2.5}$$

$$\frac{dC_7}{dt} = D_{P1} a_{nc} f_{on} C_4 - k_{71} \Theta_{71}^{(T-20)} \left( \frac{C_4}{K_{mNc} + C_4} \right) C_7 - \frac{v_{s3}(1 - f_{D7})}{D} C_7 \tag{2.6}$$

$$\begin{aligned} \frac{dC_1}{dt} = D_{P1} a_{nc} (1 - f_{on}) C_4 + k_{71} \Theta_{71}^{(T-20)} \left( \frac{C_4}{K_{mNc} + C_4} \right) C_7 - G_{P1} a_{nc} P_{NH_3} C_4 \\ - k_{12} \Theta_{12}^{(T-20)} \left( \frac{C_6}{K_{NIT} + C_6} \right) C_1 \end{aligned} \tag{2.7}$$

$$\begin{aligned} \frac{dC_2}{dt} = k_{12} \Theta_{12}^{(T-20)} \left( \frac{C_6}{K_{NIT} + C_6} \right) C_1 - G_{P1} a_{nc} (1 - P_{NH_3}) C_4 \\ - k_{2D} \Theta_{2D}^{(T-20)} \left( \frac{K_{NO_3}}{K_{NO_3} + C_6} \right) C_2 \end{aligned} \tag{2.8}$$

$$P_{NH_3} = C_1 \left( \frac{C_2}{(K_{MN} + C_1)(K_{MN} + C_2)} \right) + C_1 \left( \frac{K_{MN}}{(C_1 + C_2)(K_{MN} + C_2)} \right) \tag{2.9}$$

where;

$C_1$ = Ammonia nitrogen ($NH_3$) (mgN/L)

$C_2$ = Nitrate nitrogen ($NO_3$) (mgN/L)

$C_6$ = DO ($mgO_2$/L)

$C_7$ = Organic Nitrogen (ON) (mgN/L)

$a_{NC}$ = Nitrogen to carbon ratio (mgN/mgC)

$k_{71}$ = Organic nitrogen mineralization rate at 20 $^{o}$C ($day^{-1}$ )

$\Theta_{71}$ = Temperature correction coefficient for $k_{71}$ (-)

$k_{12}$ = Nitrification rate ($day^{-1}$)

$\Theta_{12}$ = Temperature correction coefficient for $k_{12}$ (-)

$K_{mNc}$ = Half saturation constant for phytoplankton limitation of nitrogen recycle (mgC/L)

$K_{NIT}$ = Half saturation constant for oxygen limitation for nitrification ($mgO_2$/L)

$k_{2D}$ = Denitrification rate at 20 $^{o}$C ($day^{-1}$)

$\Theta_{2D}$ = Temperature correction coefficient for $k_{2D}$ (-)

$K_{NO3}$ = Michaelis constant for denitrification ($mgO_2$/L)

$f_{ON}$ = Fraction of dead and respired phytoplankton recycled to the organic nitrogen pool (-)

1- $f_{ON}$ = Fraction of dead and respired phytoplankton recycled to the ammonia nitrogen pool (-)

$P_{NH3}$ = Preference for ammonia uptake term (-)

$f_{D7}$ = Fraction dissolved organic nitrogen in the water column (-)

### 2.3.1.3. DO

DO concentrations are dependent to the other state variables. The sources of oxygen are reaeration and evolution by phytoplankton during growth. The sinks of oxygen are algal respiration, oxidation of detrital carbon and carbonaceous material from waste effluents, non-point discharges, and nitrification. The general mass balance equation is given below (Wool *et al*, 2001).

$$\frac{dC_6}{dt} = k_2(C_S - C_6) - k_d\Theta_d^{(T-20)}\left(\frac{C_6}{K_{BOD}+C_6}\right)C_5 - \frac{64}{14}k_{12}\Theta_{12}^{(T-20)}\left(\frac{C_6}{K_{NIT}+C_6}\right)C_1 - \frac{SOD}{D}\Theta_S^{(T-20)} + G_{P1}\left(\frac{32}{12}+\frac{48}{14}\frac{14}{12}(1-P_{NH_3})\right)C_4 - \frac{32}{12}k_{1R}\Theta_{1R}^{(T-20)}C_4 \quad (2.10)$$

where;

$k_d$ = Deoxygenation rate at 20 $^oC$ ($day^{-1}$)

$k_2$ = Reaeration rate at 20 $^oC$ ($day^{-1}$)

$C_S$ = DO Saturation ($mgO_2/L$)

$C_5$ = Carbonaceous BOD (CBOD) ($mgO_2/L$)

$\Theta_d$ = Temperature correction coefficient for $k_d$ (-)

$K_{BOD}$ = Half saturation constant for oxygen limitation ($mgO_2/L$)

SOD = Sediment Oxygen Demand ($g/m^2$-day)

$\Theta_S$ = Temperature correction coefficient for SOD (-)

$k_{1R}$ = Phytoplankton respiration rate at 20 $^oC$ ($day^{-1}$)

$\Theta_{1R}$ = Temperature correction coefficient for $k_{1R}$ (-)

### 2.3.1.4. Phytoplankton Kinetics

Phytoplankton kinetics affects all other systems and has a central role in eutrophication. An overview of this system is given in Figure 2.6. The general equation describing the change in concentration with respect to time is given in Equation 2.11 (Wool *et al*, 2001).

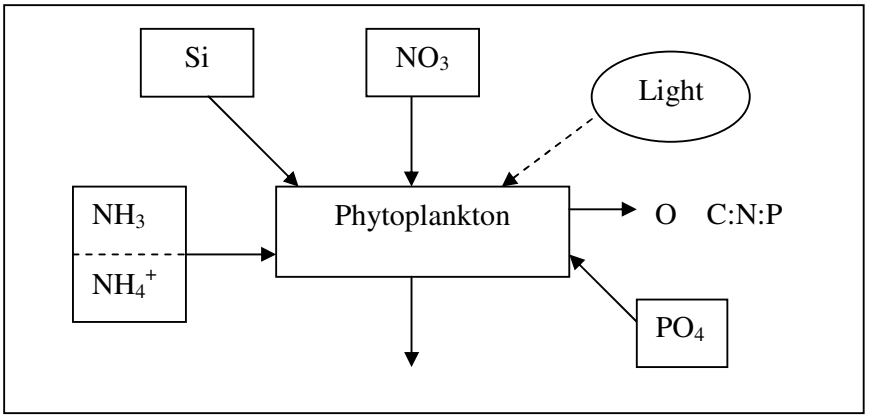


**Figure 2.6** Phytoplankton Kinetics

$$\frac{dC_p}{dt} = (R_g - R_d - R_s)C_p \qquad (2.11)$$

where;

$C_p$ = phytoplankton concentration ($mL^{-3}$)

$R_g$ = growth rate ($T^{-1}$)

$R_d$ = death rate ($T^{-1}$)

$R_s$ = settling rate ($T^{-1}$)

# CHAPTER 3

# METHODOLOGY

## 3.1. Field Study

In order to determine the current state of the water quality in Lake Uluabat, field studies were conducted on 01.05.2005, 30.06.2005 and 16.08.2005 with the help of a team including Muhittin Aslan, Elif Bulut, and Gamze Güngör Demirci. Water samples were taken from different locations. These were selected based on the previous studies as much as possible, in order to make a comparison between the current and previous conditions. Wheather conditions on the sampling dates were effective in selection of the sampling locations as well, since stormy conditions impacted the safety of the operation on two of the sampling dates. The locations of the sampling points for the field work for May 2005 are depicted in Figure 3.1.

During the sampling study, the location coordinates were recorded using the Magellan Sportrak GPS receiver. This is employed since practically it was not possible to get samples from the same exact coordinates at different sampling dates. At each sampling location, in-situ measurements were taken for temperature, DO, turbidity, oxidation-reduction potential and pH at several points along the water depth. For this purpose, a YSI 6600 EDS multiparameter water quality sonde (Figure 3.2) was utilized. The sonde works under -5 and +45 $^{0}$C. The data is automatically recorded in user specified time intervals with respect to depth. The time interval was set to two seconds, and the probe was suspended into water very slowly. Data was recorded until the sonde hit to the bottom of the lake. Then, the data was transferred to a regular computer for processing in the office.

**Figure 3. 1** Approximate locations of sampling points (U1-U4)

**Figure 3. 2** YSI 6600 EDS sonde (Elahdab, 2006)

In addition to in-situ measurements, water samples were taken using Wildco 1120 D40 model vertical sample collecter. Samples were taken from the surface, mid-depth and bottom of the lake for the months of May and July in 2005 . The apparatus was setup and lowered to the specified depth. Then, its messenger was dropped to close the caps of the cylindrical container, resulting in the entrapment of about 3 lt of lake water. Then, the container was pulled out of water and the sample was emptied into PVC containers. These containers were then immediately put in a cooler and kept away from direct sun. In the laboratory, sample containers were transferred into a large capacity cooler. In June 2005, samples were taken only from the surface since the messenger on the sampler did not operate. The locations of sampling points and definitions about the samples are given in Table 3.1.

Samples were analyzed for $NO_3$, TKN, total phosphorus (TP), and Chl-a in the laboratory. The methods used for this purpose are as follows:

- Ascorbic Acid Method (Method 4500-P E) (APHA 1997) for TP,
- Macro-Kjeldahl Method (Method 4500-$N_{org}$ B) (APHA 1997) for TKN,
- Colorimetric Brusin Method (Method 352.1) (U.S. EPA, 1971) for $NO_3$,
- Ethanol extraction method (ISO: 10260, 1992 standard) by Papista et al. (2002) for Chlorophyll-a were used .

In addition to the listed water quality parameters above, secchi disc (SD) depths were measured to check the transparency of the water. For this purpose a standard 20 cm weighted disk of alternating black-and-white SD was used. The disk was lowered in the water until the black-and-white quadrants were no longer distinguishable. At that point, the depth was recorded using the graduated line connected to the SD.

**Table 3. 1** The locations of sampling points and sample definitions

| Sampling station | Sample No. | Depth | Coordinates |
|---|---|---|---|
| **Date: 01.05.2005** | | | |
| U1 | U11 | Surface | 40° 09.990 N, 28° 39.609 E |
| | U12 | Mid | |
| | U13 | Bottom | |
| U2 | U21 | Surface | 40° 11.222 N, 28° 34.416 E |
| | U22 | Mid | |
| | U23 | Bottom | |
| U3 | U31 | Surface | 40° 11.854 N, 28° 35.236 E |
| | U32 | Mid | |
| | U33 | Bottom | |
| U4 | U4 | Only sonde measurement | 40° 10.620 N, 28° 38.124 E |
| **Date: 30.06.2005** | | | |
| H1 | H1 | Surface | 40° 10.163 N, 28° 38.508 E |
| H2 | H2 | Surface | 40° 09.073 N, 28° 37.807 E |
| H3 | H3 | Surface | 40° 08.684 N, 28° 37.553 E |
| H4 | H4 | Surface | 40° 08.117 N, 28° 37. 090 E |
| H5 | H5 | Surface | 40° 09.391 N, 28° 34.768 E |
| H6 | H6 | Surface | 40° 10.673 N, 28° 33.263 E |
| H7 | H7 | Surface | 40° 11.148 N, 28° 32.701 E |
| H8 | H8 | Surface | 40° 11.577 N, 28° 32.443 E |
| H9 | H9 | Surface | 40° 11.276 N, 28° 35.884 E |
| **Date: 16.08.2005** | | | |
| A1 | A11 | Surface | 40° 08.896 N, 28° 38.780 E |
| | A12 | Mid | |
| | A13 | Bottom | |
| A2 | A21 | Surface | 40° 08.489 N, 28° 36.260 E |
| | A22 | Mid | |
| | A23 | Bottom | |
| A3 | A31 | Surface | 40° 09.260 N, 28° 34.798 E |
| | A32 | Mid | |
| | A33 | Bottom | |
| A4 | A41 | Surface | 40° 11.045 N, 28° 33.164 E |
| | A42 | Mid | |
| | A43 | Bottom | |
| A5 | A5 | Only sonde measurement | 40° 09.168 N, 28° 37.614 E |
| A6 | A6 | Only sonde measurement | 40° 11.596 N, 28° 34.273 E |

All the laboratory analysis has been performed based on the standard methods. However, for Chl-a concentration determination, the improved ethanol extraction method (ISO: 10260, 1992 standard) by Papista et al. (2002) was used. Ethanol extraction was preferred over the standard acetone extraction (Method 10200-H, APHA 1997) since it was shown to be superior (Papista et al., 2002).

The procedure applied is summarized below:

1- 250 ml of sample was filtered using a glass wool filter.
2- The filter paper was put into a covered glass container.
3- 20 ml of 90% pure ethanol was poured over the filter paper into the container.
4- The container was closed and put into a warm water bath at 75 $^{o}$C for 15 minutes.
5- After removing the container from water bath, it was left to cool down to room temperature (30 min to 1 hr). The filter paper was shaken off and removed from the container. Then the container was put into a cooler and kept in there for about a day.
6- Absorbed light by the processed sample was measured using the HACH 2400 Spectrophotometer at 665 nm wavelenght against an ethanol blank.
7- Step 6 was repeated at 750 nm.
8- Then about 0.01-0.02 ml hydrochloric acid (HCl) was added into the cuvette containing the sample and mixed occasionally in an interval of about 30 minutes (Initial testing indicated that absorption readings were stabilized after mixing for 30 minutes).
9- Steps 6 and 7 were repeated.

The Chl-a concentrations were determined using the below equation;

$$\text{Chl-a } (\mu g/lt) = (A-A_a)*29.6*V_e/(V_n*L) \quad (3.1)$$

where;

$V_e$ = extract volume, ml

$V_n$ = sample volume, l

L = light path length, mm

A = $A_{665}$-$A_{750}$ = absorbance difference before acid addition

$A_a$ = $A_{665acid}$-$A_{750acid}$ = absorbance difference after acid addition

### 3.2 Model Application

In order to model the water quality in Uluabat Lake, the surface area was divided into segments. In doing that, the number and location of few observation points were taken into consideration. Based on the historical data, the lake was divided into a total of 10 segments. Out of those, 5 of them were the benthic ones. No epilimnion or hypolimnion division was made since the lake was shallow and exhibited very weak temperature stratification. On the horizontal plane, the lake was divided into 5 segments. This was achieved by using the satellite image of the Uluabat Lake obtained from the Statistical Agency of Turkey in 2004 and the bathmetry map of the lake (Kurttaş *et al.*, 2002).

MapInfo Professional, version 7.5, was utilized to define the coordinates of the lake area and divide the lake into segments by taking the sampling station locations used in the present and previous studies into account. In addition to this, the number and location of available data for model calibration was also considered during segmentation. Surface area of each segment and total area of the lake was calculated using the same program. Exchange areas between adjacent segments were calculated by combining the bathmetry and the coordinate information in the maps of the lake. Figure 3.3 shows the water quality model segments and the horizontal exchanges between them.

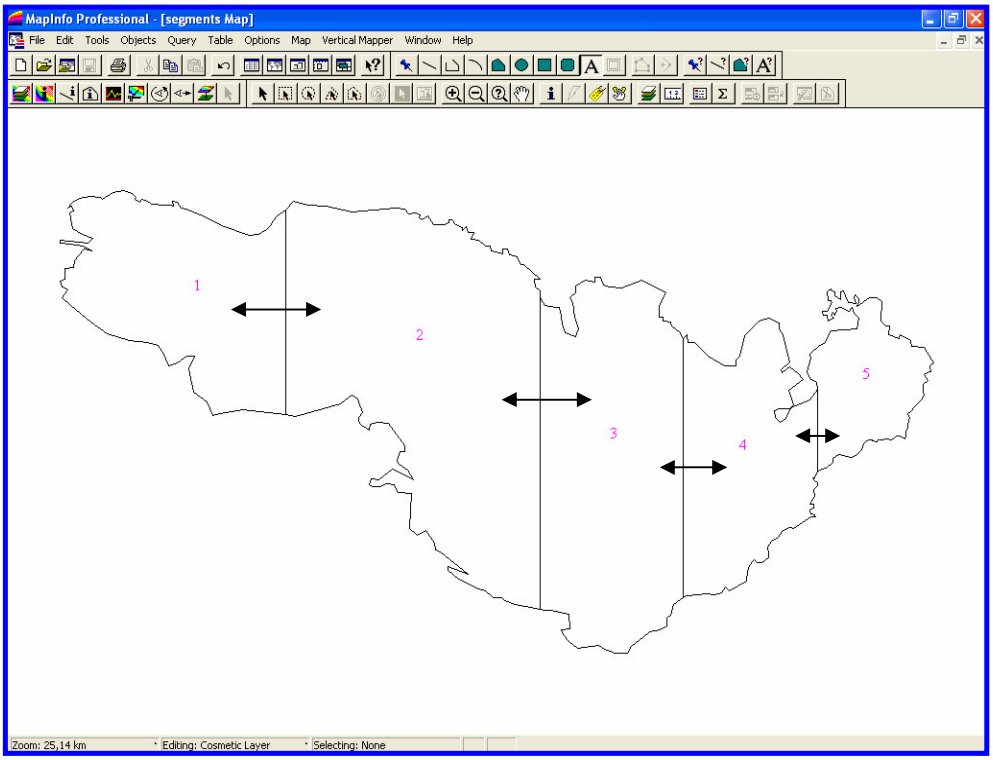


**Figure. 3. 3** The Water Quality Model Segments for Uluabat Lake and Exchange Directions Between Them

In order to use a model to examine the water quality in a water body, first it should calibrated and verified for that system. In this iterative process, first the model is run for a specified time period for the purpose of adjustment of rate constants. When an agreement is attained between the simulation and observation values, the rate constants are verified against another time period representing a different condition. If similarity still holds for the observed and calculated water quality parameter quantities, then the calibration and verification of the model is completed. Only after successful calibration and verification, a model can be used to make predictions for different management scenarios.

Calibration of the model was performed for the time period January 1st to June 1st of 2000. The data belonging to this period was obtained from the "water quality assessment report for Uluabat and vicinity" (Yersiz *et al.*, 2001) published by the Turkish Ministry of Environment and Forestry, Environmental Reference Laboratory. The water quality data at the beginning of January was used as the initial conditions for the water quality parameters considered in this study.

Since there was no data available showing the water level changes with respect to time, the "net flow option" was used to calculate the hydrodynamic input. According to this option, WASP7.2 sums all the inflows and outflows at the interface of the segments. Then, the mass of concern is moved in the direction of net flow (Wool et al., 2001). The input flow to the Lake is governed by the Mustafakemalpaşa Stream. The water flow rates during the time period used for calibration were obtained from the General Directorate of Electrical Power Resources Survey and Development Administration, Station number AGI-302. The only station that existed at the outlet of the lake, on the Uluabat Stream, belonged to the State Hydraulic Works. However, insufficient data prohibited the use of flow data belonging to that monitoring station. The water budget study for the lake, as presented in the "water quality assessment report for Uluabat and vicinity" (Yersiz *et al.*, 2001), showed that the total amount of water entering and leaving the lake was close to each other. Therefore, except February, March, April, September, October and November, it was assumed that the inflow to the lake was equal to the outflow. For the months of February, March, April, September, October and November, the flow direction was reversed and Uluabat Stream supplied inflow to the lake (Yersiz *et al.*, 2001; Akdeniz, 2005). The rates of inflow for these months were calculated using below water budget (Yersiz *et al.*, 2001, Akdeniz, 2005). Segment volumes calculated within WASP7.2 are illustrated in Figure 3.4.

Input:

- Precipitation falling on the surface area of the lake : $92.72*10^6$ $m^3$
- Return flow from the outlet of Uluabat Stream : $97.58*10^6$ $m^3$
- Inflow from Mustafakemalpaşa Stream : $1550.68*10^6$ $m^3$

TOTAL : $1740.98*10^6$ $m^3$

Output:

- Outlet flow to Uluabat Stream : $1553.20*10^6$ $m^3$
- Evaporation loss : $176.20*10^6$ $m^3$
- Water withdrawal for irrigation : $11.53*10^6$ $m^3$

TOTAL : $1740.93*10^6$ $m^3$

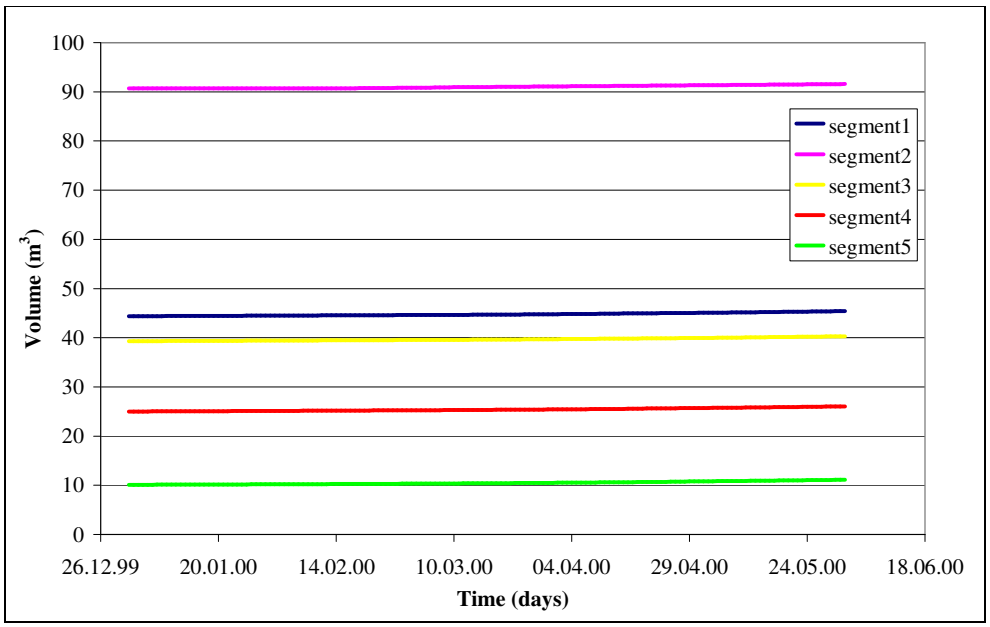


**Figure 3. 4** Segment volumes through calibration period

The daily meteorological data including the total daily solar radiation, wind velocity, air temperature, and precipitation were obtained from the Turkish State Meteorological Service. Data belonged to the nearest Mustafakemalpaşa Monitoring Station. At this station, data on wind speed and solar radiation was missing. For those parameters, data belonging to the Bursa Monitoring Station was employed in the simulations. Minimum, average and maximum values for wind speed and total solar radiation through calibration period are given in Table 3.2.

**Table 3. 2** Environmental Functions for calibration

| Environmental Function | | January | February | March | April | May |
|---|---|---|---|---|---|---|
| Wind Speed (m/s) | Minimum | 0.80 | 0.50 | 0.30 | 0.60 | 0.60 |
| | Average | 2.17 | 1.91 | 1.85 | 1.98 | 2.02 |
| | Maximum | 4.60 | 4.20 | 4.20 | 5.10 | 4.10 |
| Total Solar Radiation ($cal/cm^2/day$) | Minimum | 3.60 | 52.20 | 105.60 | 72.60 | 139.20 |
| | Average | 139.18 | 213.43 | 325.39 | 327.22 | 485.63 |
| | Maximum | 279.60 | 420.60 | 502.20 | 523.20 | 624 |

Evaporation data was obtained from the water budget (Yersiz *et al.*, 2001, Akdeniz, 2005) and distributed proportionally according to the temperature quantities. Figure 3.5 depicts inflow, outflow, precipitation and evaporation data used in calibration. The water quality data of Mustafakemalpaşa and Uluabat Streams were set as the boundary concentrations into the model. The data belonged to the State Hydraulic Works Monitoring Stations on these Streams. The water quality model time step was selected as 1 hour throughout the simulation period.

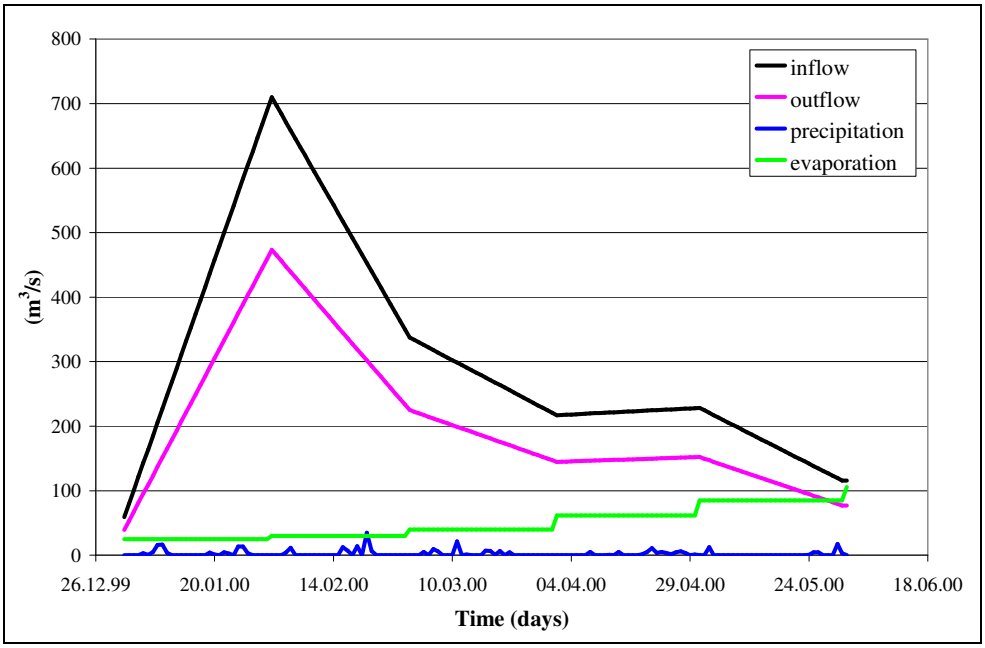


**Figure 3. 5** Inflow, outflow, precipitation and evaporation through calibration period in Uluabat Lake

In order to account for the sediment phosphorus load, a single benthic segment underlying all water column segments were employed. The benthic segment merely acted as a convenient sink for settling of organic matter. Model calculations within this segment were ignored by using the intermediate eutrophication kinetics in WASP (Wool, 2001) due to the insufficient data to simulate the benthic processes. Instead of internal model calculations, sediment loads were input as a constant $PO_4$ flux. The value, 13.62 mg/m$^2$/day, for this flux for all water columns was determined based on the study of Akdeniz (2005).

Following the calibration, verification was employed using another set of data representing different conditions. The time period for this process was June 1$^{st}$ to December 31$^{st}$, 1999. The measurement values for June 1$^{st}$ were treated as the initial conditions. Keeping the rate constants same as for the calibration period,

boundary conditions, flows and environmental time functions were changed for the verification period.

### 3.3 Water Quality Improvement Scenarios

Following the calibration and verification of the model, management options were considered for the improvement of the water quality in the lake. It is a common procedure to control the phosphorus loads for the minimization of eutrophication. As a result, the management scenarios were mainly concentrated on phosphorus load reduction. The details about these cases will be presented in the relevant section of "Results and Discussion".

# CHAPTER 4

# RESULTS AND DISCUSSION

## 4.1 Water Quality Status of the Lake

As discussed in the methodology section, monitoring was applied through May-August period, to observe the water quality in the lake. The results of this monitoring study are given in Figures 4.1, 4.2, 4.3, 4.4, 4.5, 4.6, and 4.7 for temperature, DO, turbidity, pH, Chl-a, total phosphorus (TP), and Kjehdahl nitrogen, respectively. In these figures, the minimum, average, and maximum values for each parameter at each sampling time are shown.

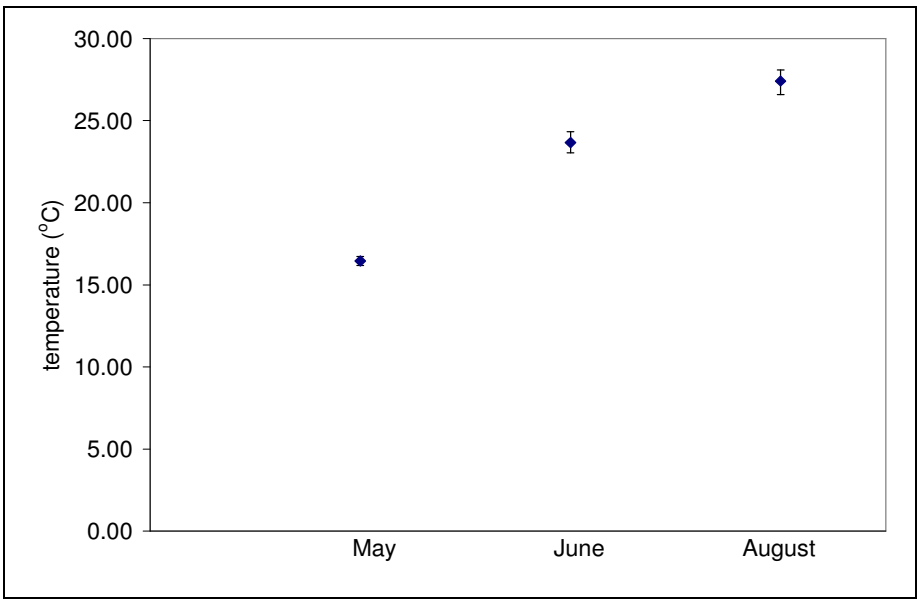


**Figure 4. 1** Monitoring data obtained from field work – temperature

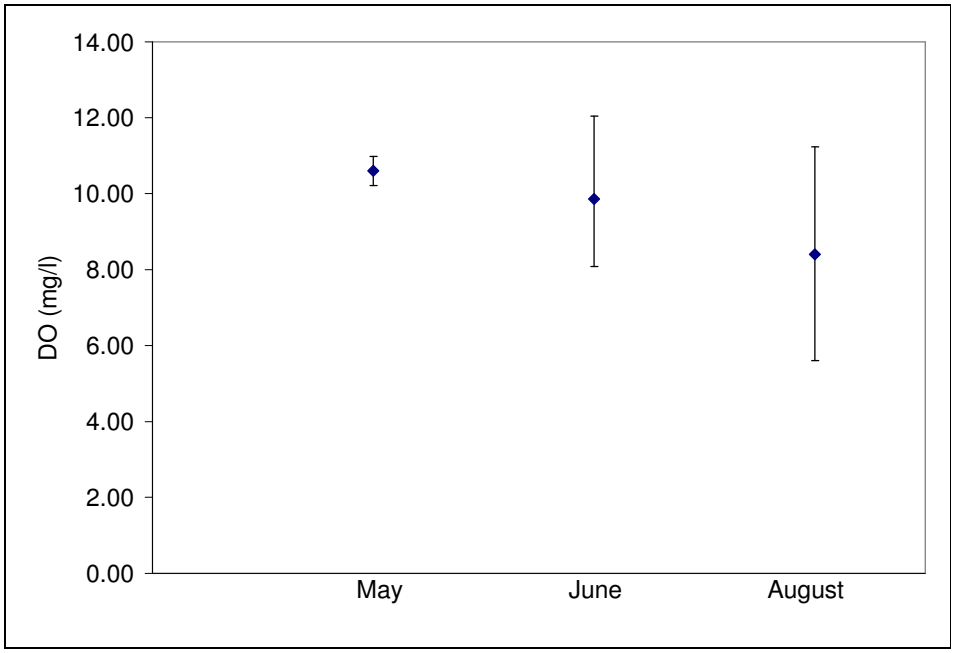


**Figure 4. 2** Monitoring data obtained from field work – DO

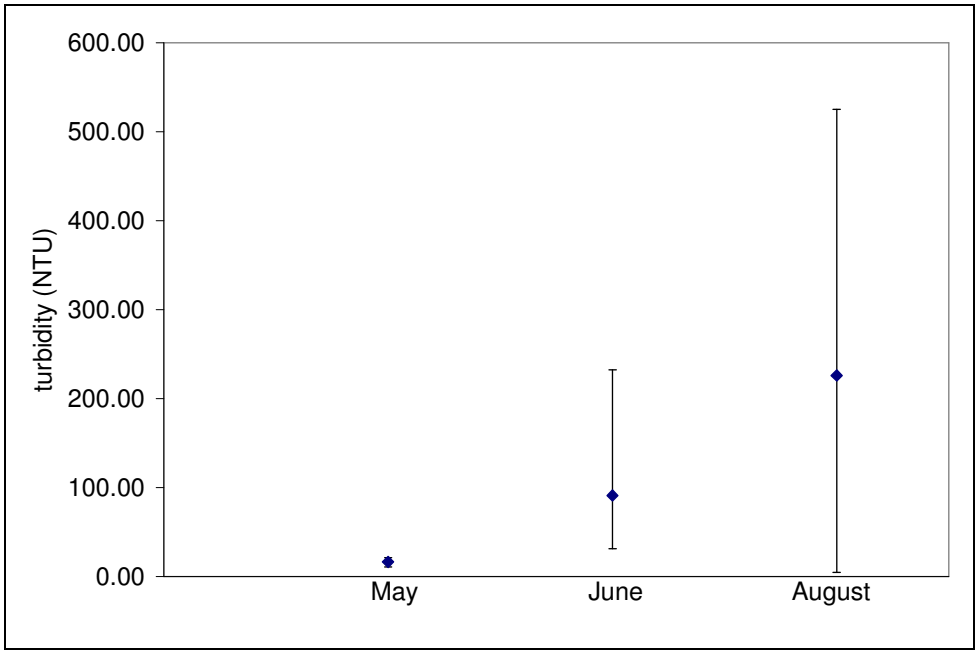


**Figure 4. 3** Monitoring data obtained from field work – turbidity

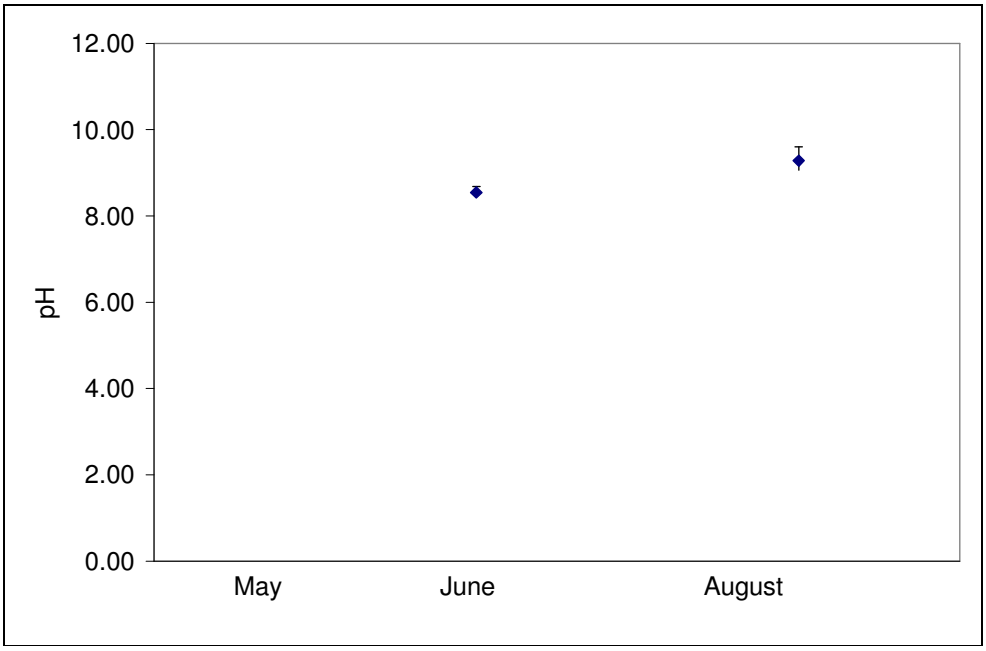


**Figure 4. 4** Monitoring data obtained from field work – pH

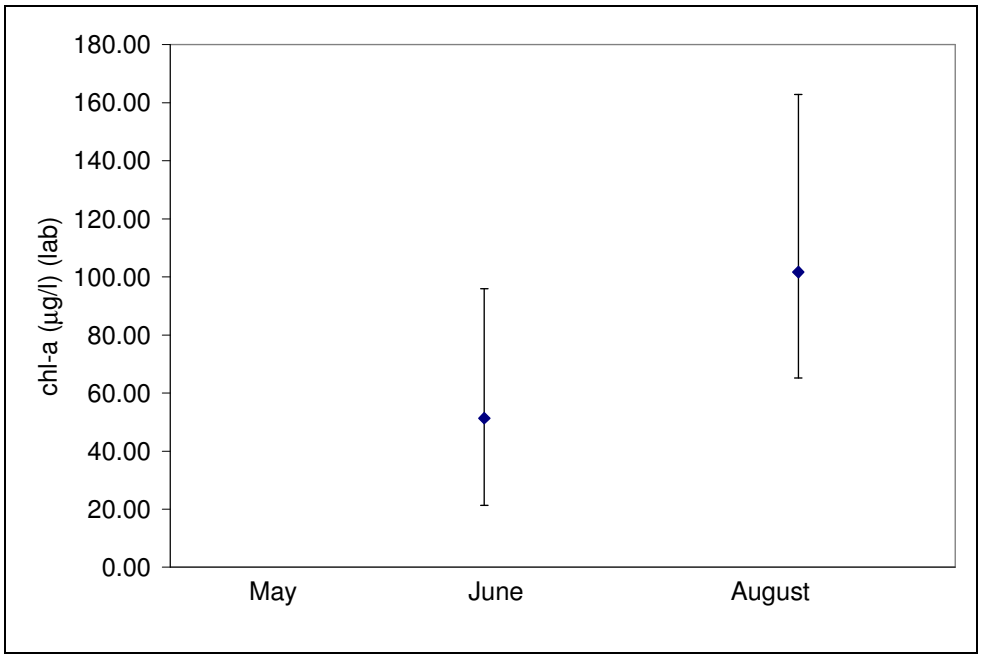


**Figure 4. 5** Monitoring data obtained from field work – Chl-a

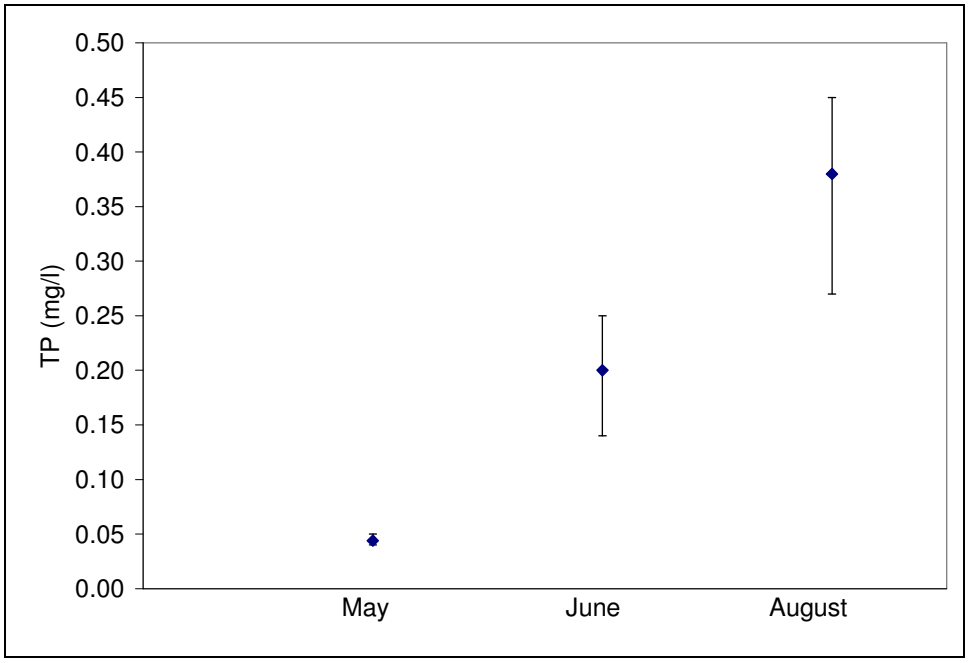


**Figure 4. 6** Monitoring data obtained from field work – TP

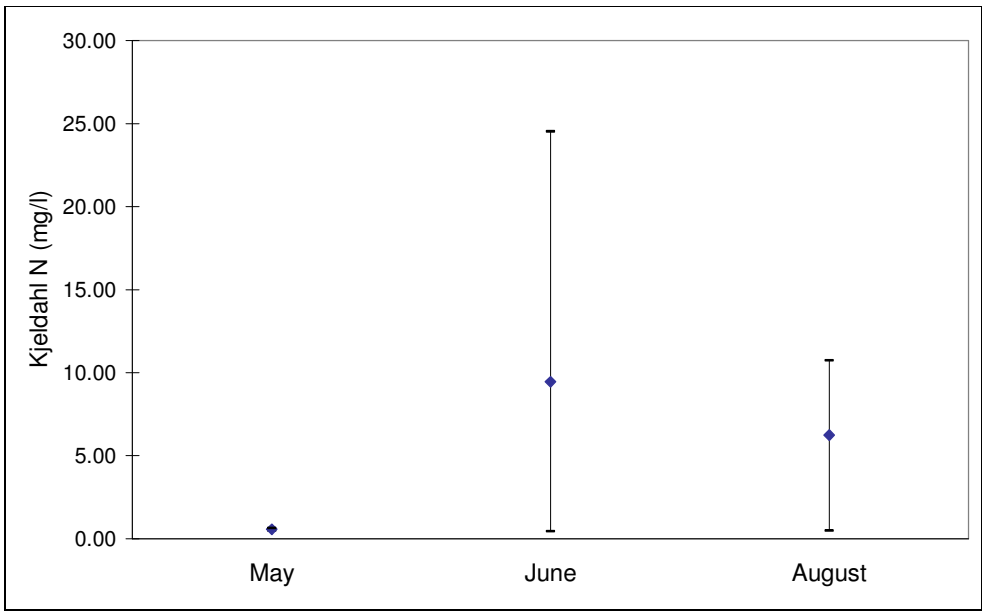


**Figure 4. 7** Monitoring data obtained from field work – Kjehdahl N

As can be derived from Figure 4.1, from May to June, there was an increase in the water temperature. In August, the average water temperature in the lake reached to about $27^{o}$ C. This is due to the shallow depth of the lake. As the temperature increased, the DO content of the lake decreased (Figure 4.2).

The algal production in the lake increased as time progressed from June to August as depicted in Figure 4.5. In August the average Chl-a concentration was above 100 μg/L. Increase in algal production brought about the increase in pH (Figure 4.4) and turbidity (Figure 4.3). However, turbidity may also be related with the re-suspension of solids from the sediment by the turbulence mainly created by the wind action. This may especially be more pronounced in summer months as the depth of water declines.

There existed an increasing trend for TP from May to August 2005 as shown in Figure 4.6. This may be due to decline in the water depth which may increase the concentration even if there exists a comsumption by algal growth. In addition, as pH increases, the P-binding capacity of the soil, or sediment, decreases. The hydroxide ions replace $PO_{4,}$ which is soluble in water. This case may be significant for pH values greater than 8 (Brönmark, 1999). As seen in Figure 4.4, the pH values were above 8 in June and August. Therefore, P release from the sediment might have enhanced. Kjehdahl nitrogen concentrations were in general increased in June and August compared to May.

Additionally, DO and temperature profiles along the water depth for the months of May, June and August in 2005 are depicted in Figure 4.8, 4.9 and 4.10, respectively. According to these profiles, the lake is considered as a completely mixed system for the modeling study.

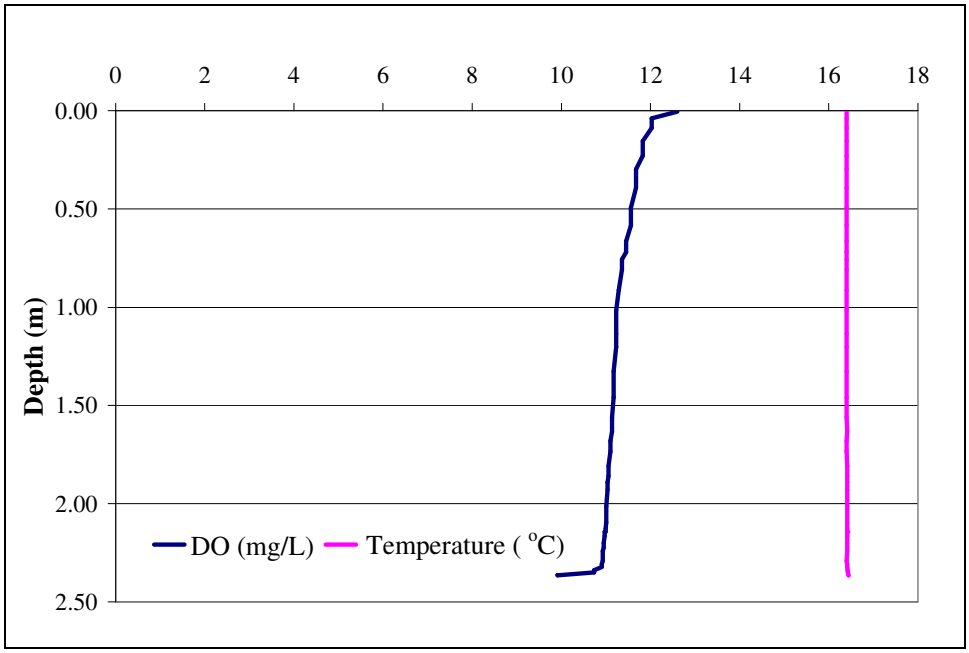


**Figure 4. 8** DO and Temperature Profiles in May 2005

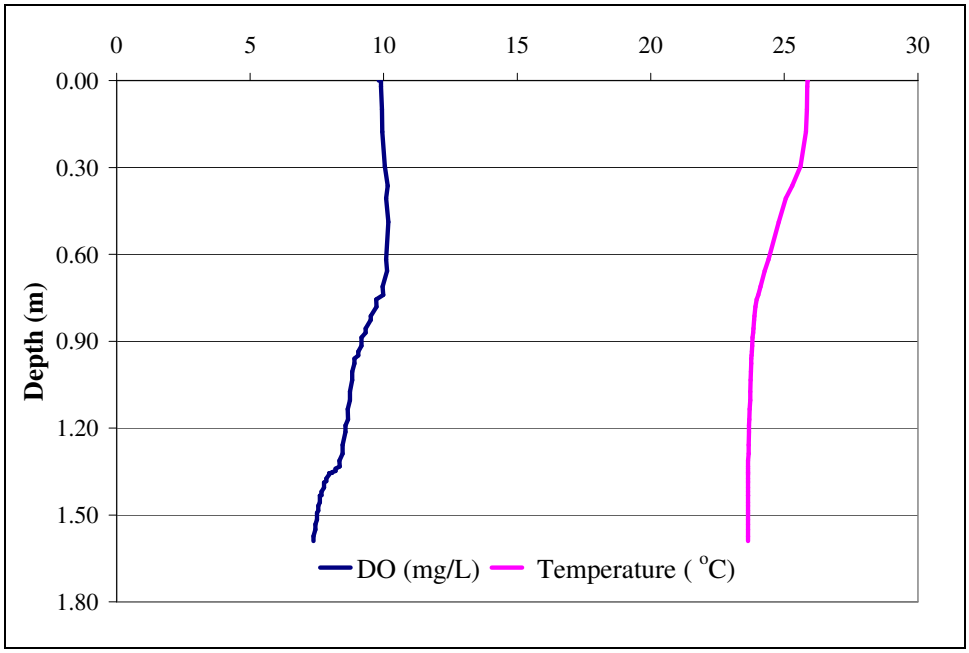


**Figure 4. 9** DO and Temperature Profiles in June 2005

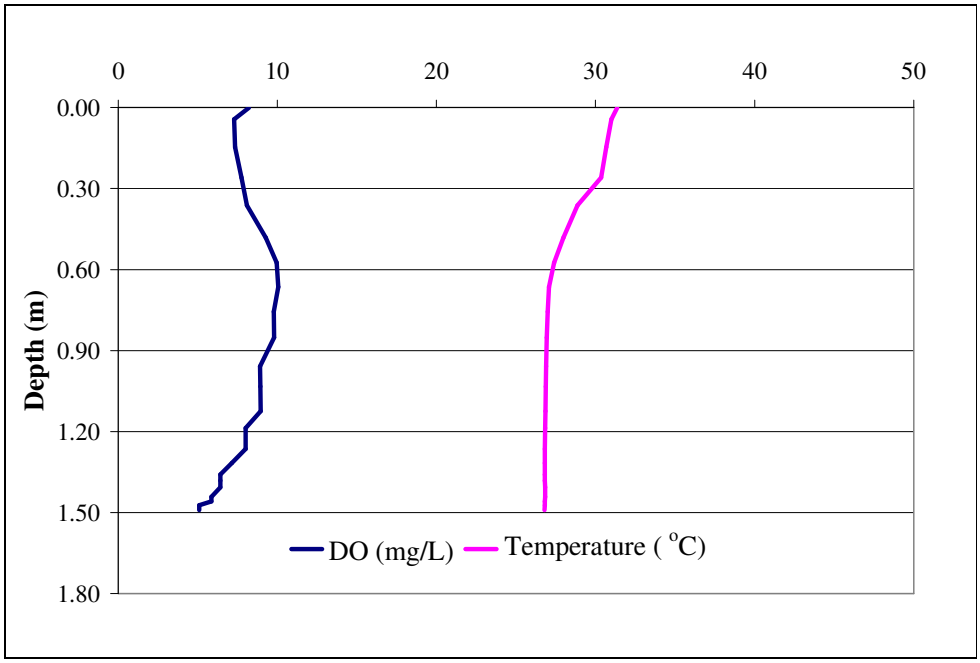


**Figure 4. 10** DO and Temperature Profiles in August 2005

An evaluation was employed to compare the progression of the general trophic status of the lake within years. For this purpose, the results obtained from past studies (Yersiz *et al.*, 2001; Akdeniz, 2005) and in-situ and laboratory analysis performed in this study were compared. Since the monitoring period in this study spanned a limited number of months, the water quality variable values belonging to similar months in the previous studies were considered. Comparison was based on the average values for the lake.

The change of phosphorus with time is shown in Figure 4.11. As can be observed, there is a decrease in TP concentration in 2002 and in 2005 compared to 1999 $PO_4$ concentration alone. This decrease can be due to implementation of the Lake Uluabat Management Plan conducted between 1998 and 2002 by the Turkish Ministry of Environment and Forestry and Association for the Protection of Nature (DHKD).

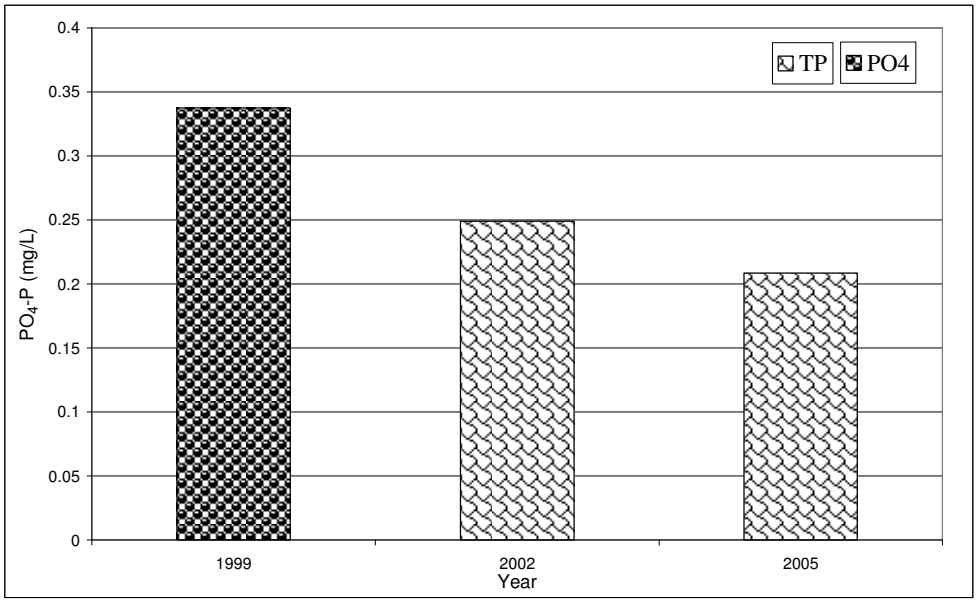


**Figure 4. 11** Change in average $PO_4$-P concentration with time for 1999-2005

General Trophic Classification of Lakes and Reservoirs in Relation to Phosphorus and Nitrogen (Wetzel, 2001) was taken into consideration in order to evaluate the present water quality conditions of the lake. In this classification, TP concentrations between 16 and 386 mg/$m^3$ are classified as eutrophic (Table 2.1). In this sense Lake Uluabat is eutrophic, at least in summer months, with respect to TP in all years.

Figure 4.12 depicts the change in the average $NO_3$ concentrations for years 1999, 2002 and 2005. As can be seen, in 1999, $NO_3$ was limited. However, 2002 and 2005 results exhibited higher values. In 2002, $NO_3$ concentration had the highest value. This may be due lower Chl-a production in this year as shown in Figure 4.13.

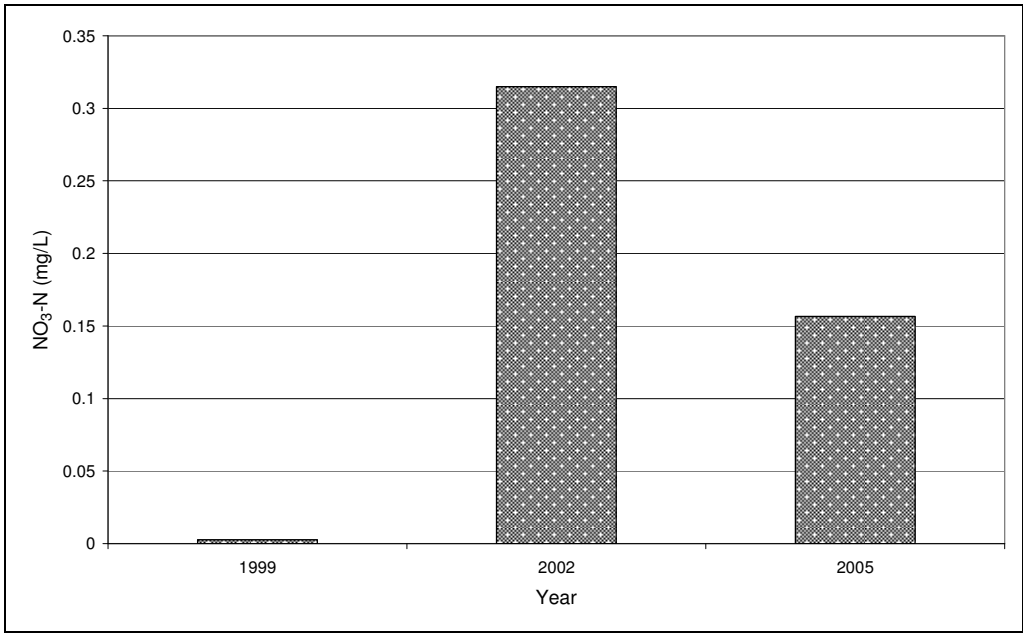


**Figure 4. 12** Change in the average $NO_3$ concentrations with time for 1999-2005

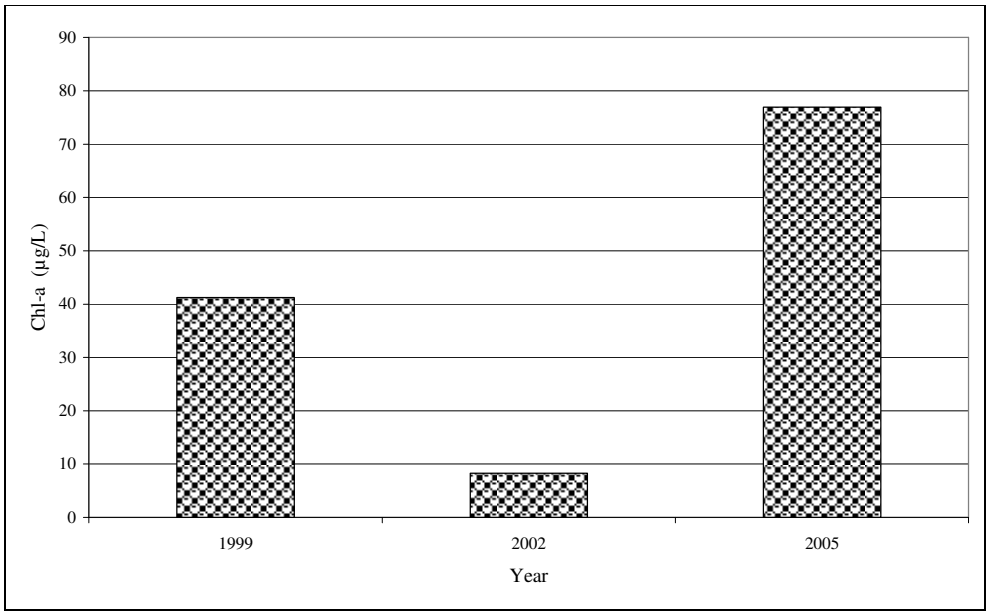


**Figure 4. 13** Change in the average Chl-a concentration with time for 1999-2005

In 2005, the average Chl-a concentration had the highest value among the other years. This may be due to declined water depth which may give rise to algal production. However, it should be noted that grazing is an important factor that may impact the phytoplankton concentrations, which was not considered in this study. Chl-a concentration is one of the main indicators for the trophic state of the lake. Regarding the Chl-a concentration, the lake is in hyper-eutrophic range according to the classiffication given by Wetzel (2001) for the summer months.

When SD values were examined (Figure 4.14), very low values were observed. These values were in the hyper-eutrophic range (Table 2.1) for the summer months. No data was available for 1999. Lower SD in 2005 compared to 2002 can be attributed to higher production of algae. However, it should be noted the lake is characterized with the presence of high concentrations of colloidal material (Celtemen et al., 2000). Therefore, low SD is related with these materials as well.

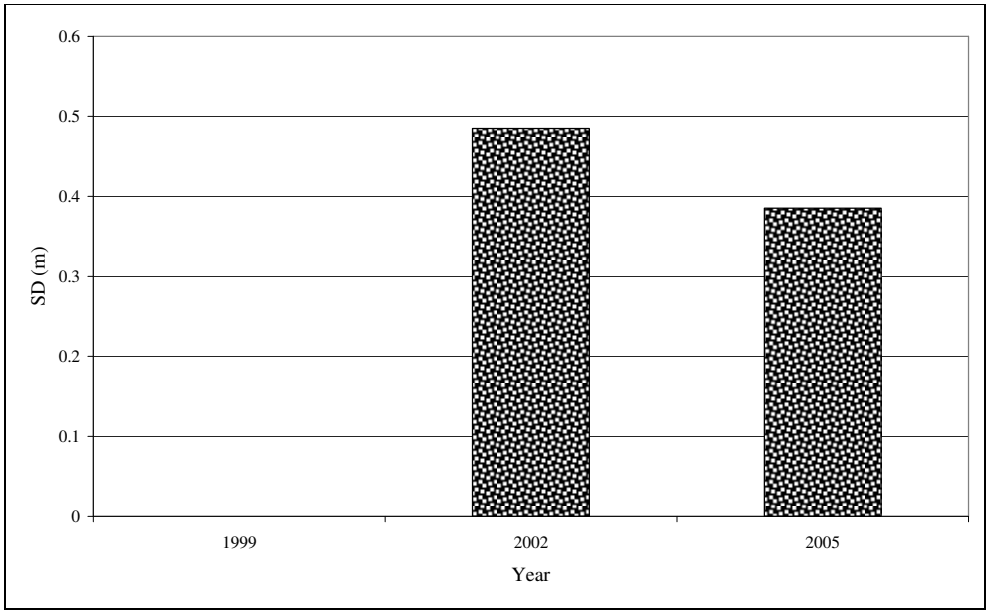


**Figure 4. 14** Change in SD with time for 1999-2005

The limiting nutrient for the system should be known to evaluate the nutrient and phytoplankton dynamics of a water body. The OECD (1982) proposed criteria to determine the limiting nutrient in the lakes as given in Table 4.1 and Table 4.2, for total and dissolved nutrient concentrations, respectively. Based on the availability of relevant data, nutrient limitations at different years were analyzed using both Table 4.1 and Table 4.2.

**Table 4. 1** Approximate Levels of Nitrogen/Phosphorus Limitation As Indicated by Total N:P Ratios in Lakes (OECD, 1982)

| Total Nitrogen/Total Phosphorus | | Limiting Nutrient | Year | |
|---|---|---|---|---|
| | | | 2002 | 2005 |
| By weight | In moles | | By weight | By weight |
| <10 | 22 | N | | |
| 10-17 | | N and/or P | | |
| >17 | 37.6 | P | 36.05 | 27.01 |

**Table 4. 2** Approximate Levels of Nitrogen/Phosphorus Limitation As Indicated by Dissolved N:P Ratios in Lakes (OECD,1982)

| Dissolved Nitrogen/Phosphate Phosphorus | | Limiting Nutrient | Year | |
|---|---|---|---|---|
| | | | 2000* | 2003** |
| By weight | In moles | | By weight | By weight |
| <5 | 10 | N | 0.686 | |
| 5-12 | | N and/or P | | |
| >12 | 26.5 | P | | 24.18 |

*: Average from January 2000 to May 2000
**: Average from January 2003 and March 2003

The cause of high productivity in lakes is most frequently because of P availability (Wetzel,2001). In Uluabat Lake, different nutrients became limiting as given in Table 4.1. In 2000, the lake was nitrogen limited. Beginning by 2002,

the limiting nutrient shifted to phosphorus. That may be due to the implementation of the management plan mentioned earlier.

### 4.2 Calibration and Verification of the Model

Calibration is a tedious process due to the number of kinetic parameters in the water quality model. Following hundreds of trials, the best combination of model parameters (Table 4.3) were obtained that provided a good match between the predictions and measurements. These kinetic parameter and constant values were reached by modifying them in the ranges given in WASP and in the literature (Lung and Larson, 1995; Celtemen, 1998; Tufford ve McKellar, 1999).

Based on the calibrated kinetic parameter values, simulation was performed to obtain the water quality parameter profiles for the lake. These results were compared to observation values and the profiles obtained for the kinetic parameter quantities of the model. These profiles will be referred to as the default profiles from herein. It must be noted that the exact dates of the observations were missing. Therefore, it was assumed that the measurements were taken at the beginning of the relevant month. The comparisons of simulated and default profiles and observed values for DO, $NO_3$, $PO_4$, and Chl-a for Segment 2 are depicted in Figures 4.15, 4.16, 4.17 and 4.18, respectively. Similar results were obtained for other segments. From herein the results will be presented for Segment 2 since this segment is associated with both the input and output to the lake. According to the figures, all simulated water quality profiles followed the trend given by the observation data. The confirmation of the fit between the observed and predicted values based on statistical tests was invalid for this study since the exact dates of the observations were not known.

**Table 4. 3** Calibrated Kinetic Parameter Values used in Modeling of the water quality in Lake Uluabat

| **Amonnia** | **Literature Value** | **Calibrated Value** |
|---|---|---|
| Nitrification Rate Constant @20 °C ($day^{-1}$) | 0-10 | 0.001 |
| Nitrification Temperature Coefficient | 0-1.08 | 1.08 |
| Half Saturation Constant for Nitrification Oxygen Limit (mg O/L) | 0-2 | 2.0 |
| **Nitrate** | | |
| Denitrification Rate Constant @20 °C ($day^{-1}$) | 0-0.09 | 0.09 |
| Denitrification Temperature Coefficient | 0-1.045 | 1.045 |
| Half Saturation Constant for Denitrification Oxygen Limit (mg O/L) | 0-1 | 1.0 |
| **Organic Nitrogen** | | |
| Dissolved Organic Nitrogen Mineralization Rate Constant @20 °C ($day^{-1}$) | 0-1.08 | 0.0001 |
| Dissolved Organic Nitrogen Mineralization Temperature Coefficient | 0-1.08 | 1.08 |
| Fraction of Phytoplankton Death Recycled to Organic Nitrogen | 0-1 | 0.5 |
| **Organic Phosphorus** | | |
| Mineralization Rate Constant for Dissolved Organic P @20 °C($day^{-1}$) | 0-0.22 | 0.0001 |
| Dissolved Organic Phosphorus Mineralization Temperature Coefficient | 0-1.08 | 1.08 |
| Fraction of Phytoplankton Death Recycled to Organic Phosphorus | 0-1 | 0.5 |
| **Phytoplankton** | | |
| Phytoplankton Maximum Growth Rate Constant @20 °C ($day^{-1}$) | 0-3 | 1.6 |
| Phytoplankton Growth Temperature Coefficient | 0-1.07 | 1.068 |
| Phytoplankton Carbon to Chlorophyll Ratio | 0-200 | 30.0 |
| Phytoplankton Half-Saturation Constant for Nitrogen Uptake (mg N/L) | 0-0.05 | 0.025 |
| Phytoplankton Half-Saturation Constant for Phosphorus Uptake (mg P/L) | 0-0.05 | 0.003 |
| Phytoplankton Endogenous Respiration Rate Constant @20 °C ($day^{-1}$) | 0-0.5 | 0.1 |
| Phytoplankton Respiration Temperature Coefficient | 0-1.08 | 1.068 |
| Phytoplankton Death Rate Constant (Non-Zooplankton Predation) ($day^{-1}$) | 0-0.25 | 0.1 |
| Phytoplankton Phosphorus to Carbon Ratio | 0-0.24 | 0.24 |
| Phytoplankton Nitrogen to Carbon Ratio | 0-0.43 | 0.15 |
| **Light** | | |
| Light Option (1 uses input light; 2 uses calculated diel light) | 1-2 | 1 |
| Phytoplankton Optimal Light Saturation | 0-350 | 300 |
| **CBOD1** | | |
| BOD (1) Decay Rate Constant @20 °C ($day^{-1}$) | 0-5.6 | 0.09 |
| BOD (1) Decay Rate Temperature Correction Coefficient | 0-1.07 | 1.047 |
| BOD (1) Half Saturation Oxygen Limit (mg O/L) | 0-0.5 | 0.1 |

The wind-driven reaeration option used in this study, as well as the sources and sinks of oxygen, resulted in high variation in DO values. This option is utilized due to the shallow depth of the water. In such lakes, the impact of wind driven turbulence on reaeration can be significant (Havens et. al, 2001). As depicted in Figure 4.18, Chl-a concentrations increase in spring as a result of abundance of nutrients and temperature increase. However, in summer, potential nutrient limitations is the potential cause of limitation of the growth.

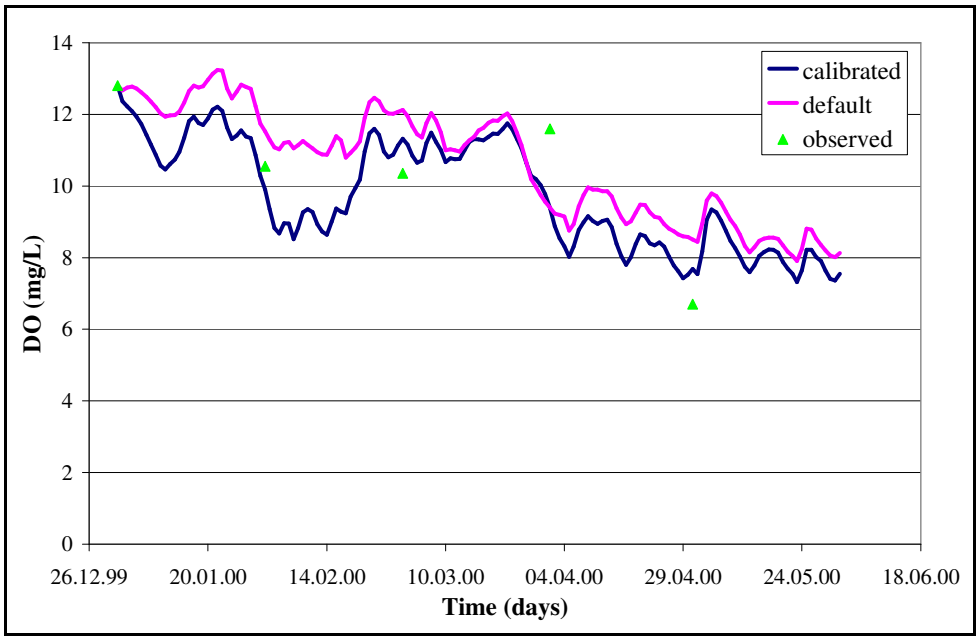


**Figure 4. 15** Comparison of measured, predicted and default values for DO at Segment 2

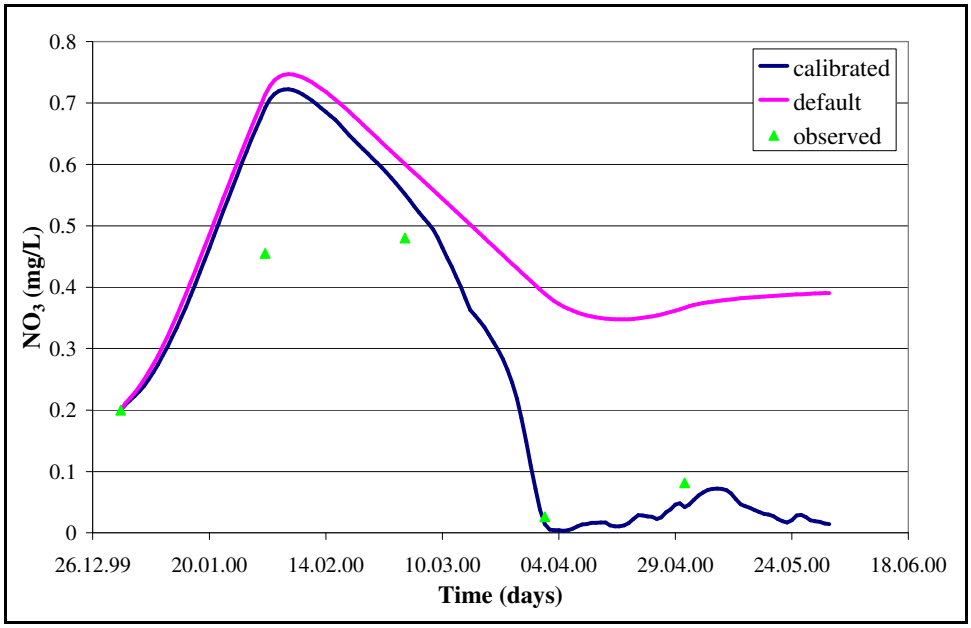


**Figure 4. 16** Comparison of measured, predicted and default values for $NO_3$ at Segment 2

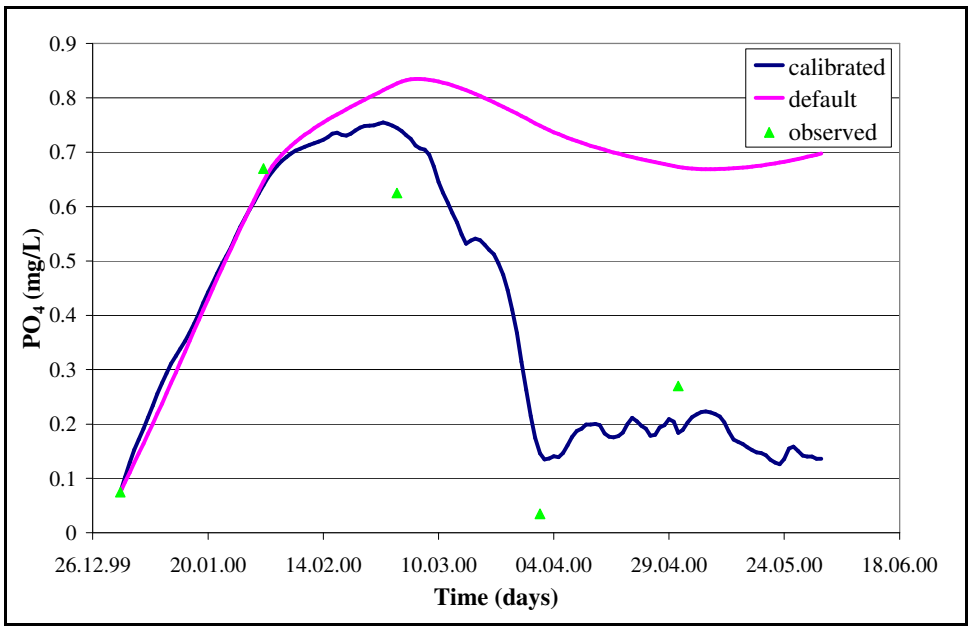


**Figure 4. 17** Comparison of measured, predicted and default values for $PO_4$ at Segment 2

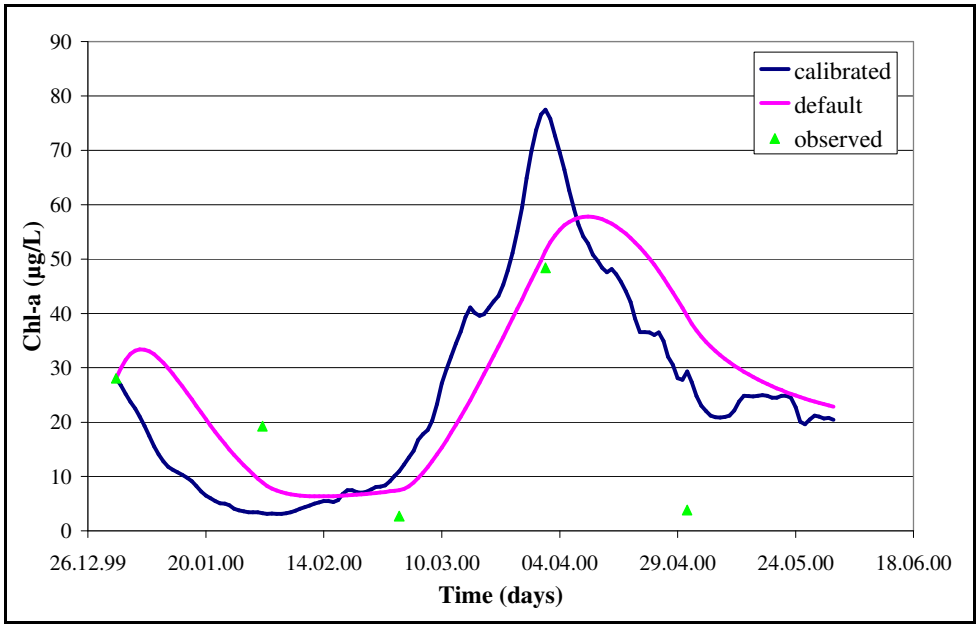


**Figure 4. 18** Comparison of measured, predicted and default values for Chl-a at Segment 2

When the $NO_3$ and $PO_4$ concentration profiles are compared with the ones for the typical shallow eutrophic lakes (Figures 4.19 and 4.20, respectively), similar trends are observed. In summer, $NO_3$ and $PO_4$ concentrations are minimized. In winter, these concentrations are high. This difference is caused by the algal productivity in summer which results in the depletion of $NO_3$ and $PO_4$ at the same time. It must be noted that, January-April season is the period in which fertilizer application is predominant in the agricultural areas around the lake. Therefore, during winter and spring, agricultural based phosphorus and nitrogen species can reach to the lake and boost these concentrations in addition to the spring turnover derived increase. One of the problems associated with the Uluabat Lake is the transport of SS via Mustafakemalpasa Stream. Phosphorus strongly binds to SS. Therefore, SS carried by the river can act as a mechanism for the transport of phosphorus to Uluabat Lake, as well as the sediment derived source.

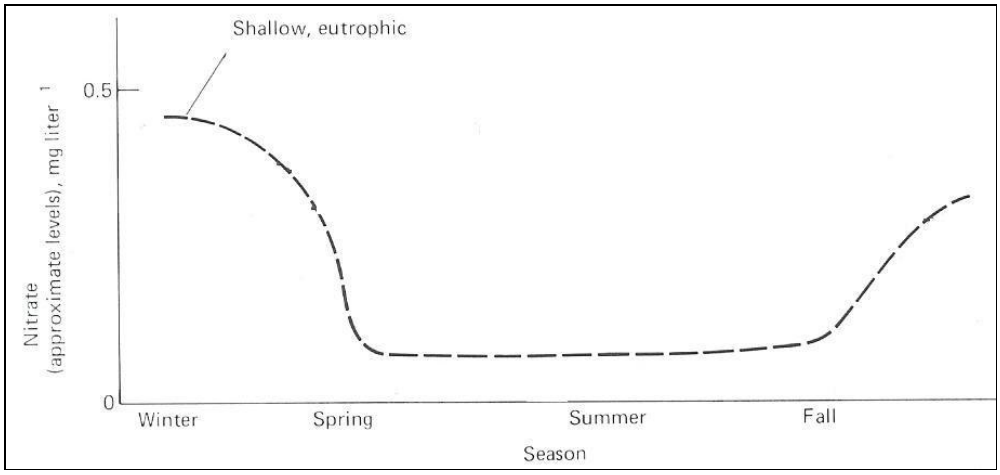


**Figure 4. 19** Typical seasonal $NO_3$ concentrations for shallow eutrophic lakes (Horne and Goldman, 1994)

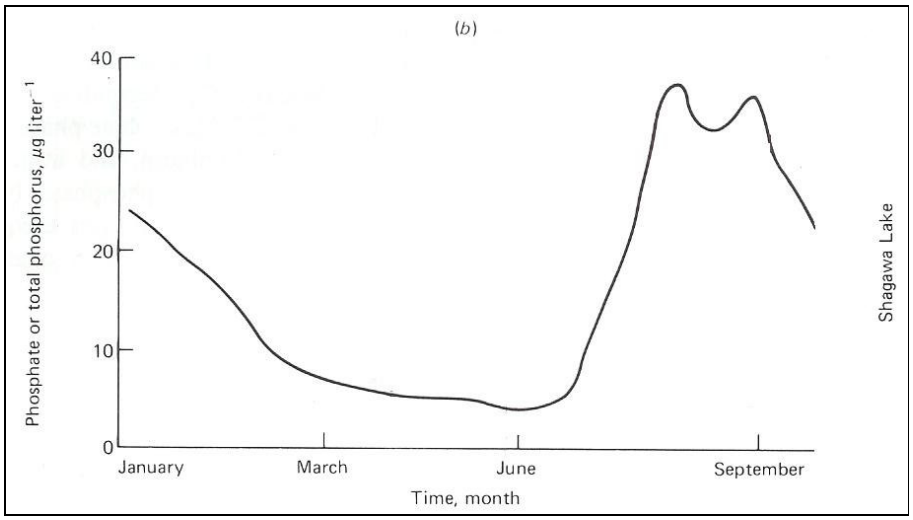


**Figure 4. 20** Typical Seasonal phosphate change for shallow eutrophic lakes (Horne and Goldman, 1994)

Following the calibration, the model was verified for another data set as described in the Methodology section. The comparison of the predicted values for the verification conditions with the observation and default data for DO, $NO_3$, $PO_4$ and Chl-a at Segment 2 are depicted in Figures 4.21, 4.22, 4.23 and

4.24, respectively. Higher DO values were calculated compared to the observation values for summer months. This may be due to the wind driven reaeration option used in this study. For $PO_4$ and $NO_3$, acceptable results were obtained. However, it was not possible to capture the increase in the Chl-a concentrations in summer of 1999. This may be due to the presence of water lilies on the surface of the water. The observation data obtained from the Ministry of Environment mostly represents the surface concentrations. In summer time, the deviation between the mean and surface concentrations can be high due to the eutrophic nature of the lake and presence of algae or water lilies on the surface. Surface observation data may not be representative of the mean concentrations. In addition, compared to the calibration conditions, relatively limited input data was available for the verification conditions. For example, changes in the water depth could not be accommodated due to lack of data. In addition, verification period mostly represents the summer conditions and there are studies in literature where the model kinetic parameters, such as carbon to Chl-a ratio (Wool *et al*, 2001), are changed with season. Such an approach is not followed in this study. Rather, average kinetic parameter values that would give a reasonable fit for both verification and calibration conditions were preferred.

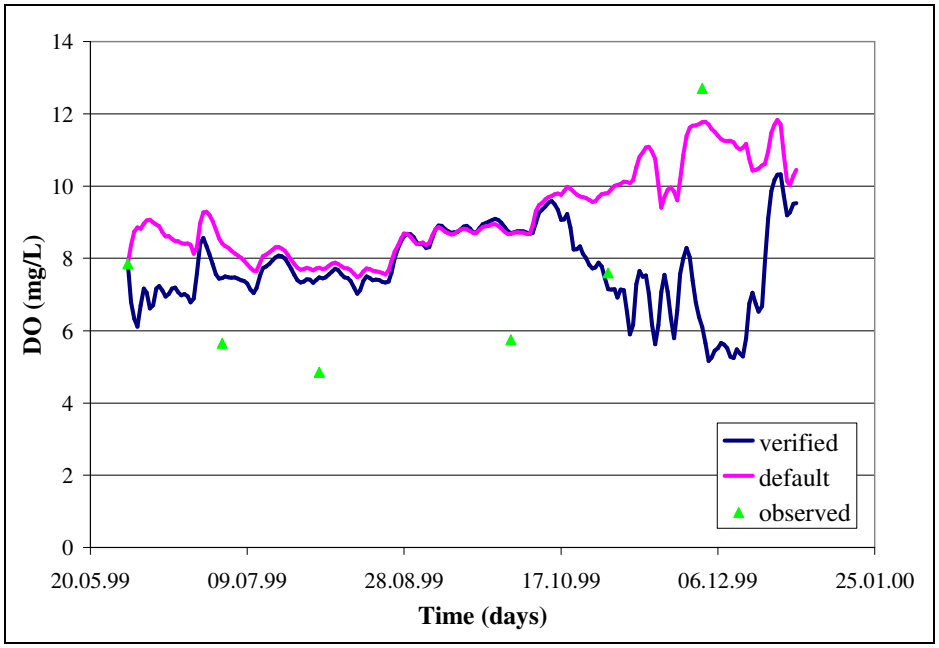


**Figure 4. 21** Comparison of measured, predicted and default values for DO at Segment 2

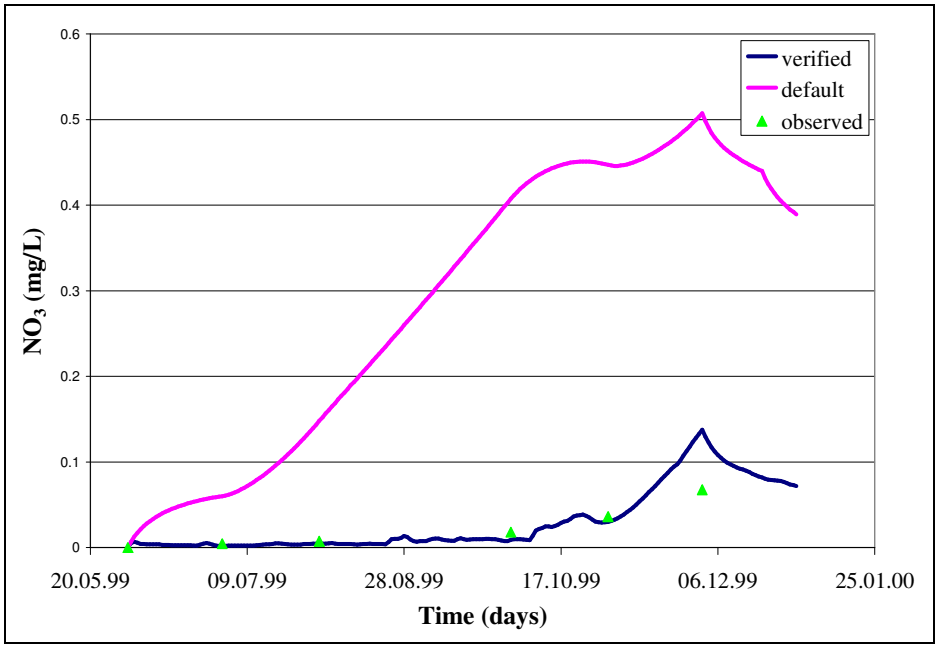


**Figure 4. 22** Comparison of measured, predicted and default values for $NO_3$ at Segment 2

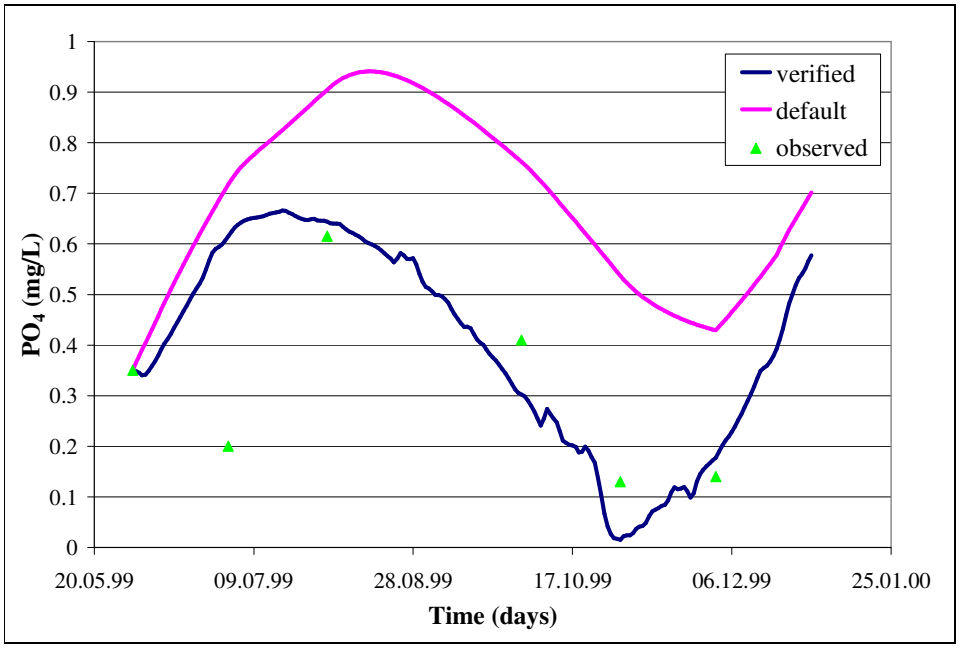


**Figure 4. 23** Comparison of measured, predicted and default Values for $PO_4$ at Segment 2

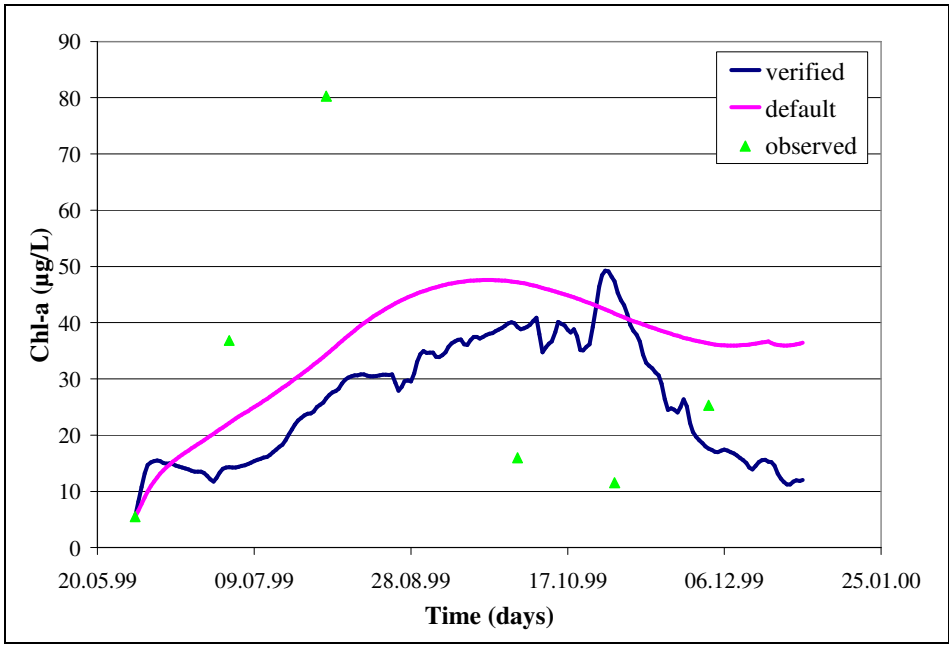


**Figure 4. 24** Comparison of measured, predicted and deafult values for Chl-a at Segment 2

### 4.3 Sensitivity Analysis

Following the calibration and verification of the model, sensitivity analysis was performed on the model outputs for the kinetic parameters used in this study by changing one variable at a time to show that lake is nitrogen limited thorugh the simulation period (given in Table 4.3). Below are the ones which the model exhibited relatively higher sensitivity compared to other kinetic parameters. Comparisons are based on the kinetic parameter values obtained following calibration and verification, which will be referred to as the "base case". Simulation period was taken as January 2000 to January 2001.

The analysis is performed such that the kinetic parameter values given in Table 4.3 were increased and decreased by 25% and 50%, providing that the modified values were within the ranges given for that parameter in the model. In the cases where this condition could not be met (for example the minimum value or the absolute value of 0 was used in the calibration), the average and maximum values were tested. Root Mean Square Error (RMSE) is used to evaluate the difference between the results for the base case and the other simulations. RMSE has a value of zero when base case values coincide with the simulation results. The increase in RMSE value for this study indicates larger difference between the base case and the simulation of concern, therefore, higher deviation from the base case.

#### 4.3.1 Sensitivity on Dissolved Organic Nitrogen Mineralization Rate

Dissolved organic nitrogen (DON) mineralization is a mechanism in which dissolved organic nitrogen is converted to dissolved inorganic nitrogen (DIN). DON mineralization rate used in the base case was replaced with the average and maximum values in the range provided in the model for this parameter to determine the effect of this constant on the water quality parameters of interest. When the model results were compared, noticeable differences were observed in

$PO_4$ and Chl-a concentration profiles (Figures 4.26 and 4.27, respectively). RMSE for $PO_4$ at Segment2 when average and maximum DON mineralization rates used were 0.1993 and 0.2028, respectively. As DON mineralization rate increased, the average available dissolved inorganic nitrogen concentration increased in the water column (Figure 4.28). Therefore, phytoplankton production was enhanced, which resulted in the usage of $PO_4$ as well (Figure 4.26). The average $PO_4$ concentration for Segment 2 decreased from the base value of 0.38 mg/L to 0.24 mg/L and 0.23 mg/L for the average and maximum mineralization rates, respectively. Increase in phytoplankton production due to rise in nitrate concentration shows that the lake is nitrogen limited during simulation period. Nitrogen limitation, phosphorus limitation and light limitation values through the simulation period given in Figure 4.25 confirm that the lake is nitrogen limited for most of the simulation period. A value less than one shows increase risk of limitation.

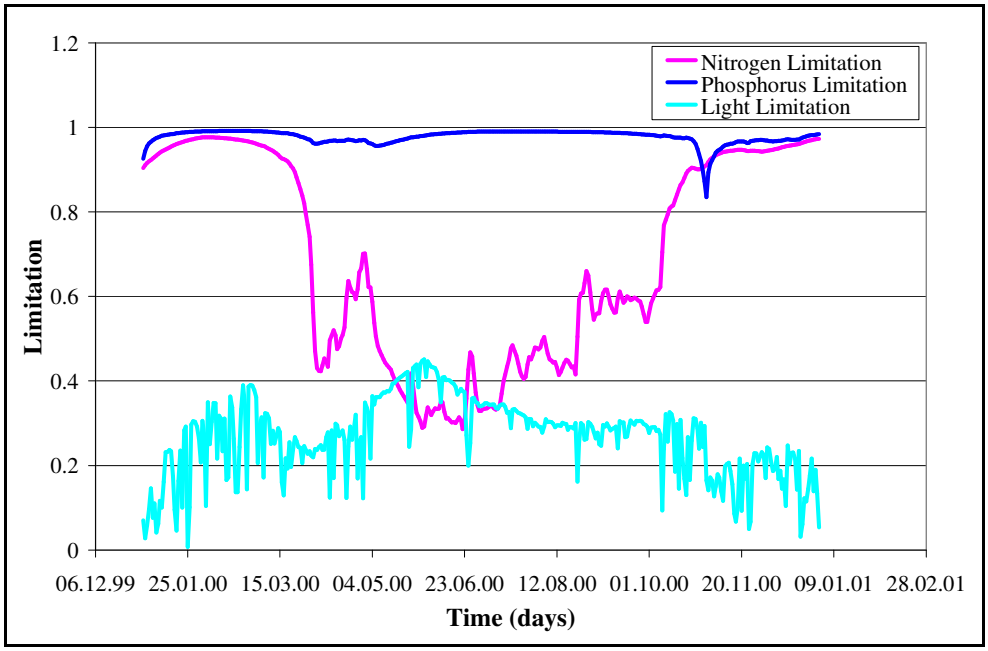


**Figure 4. 25** Nitrogen, Phosphorus and Light Limitations during simulation period

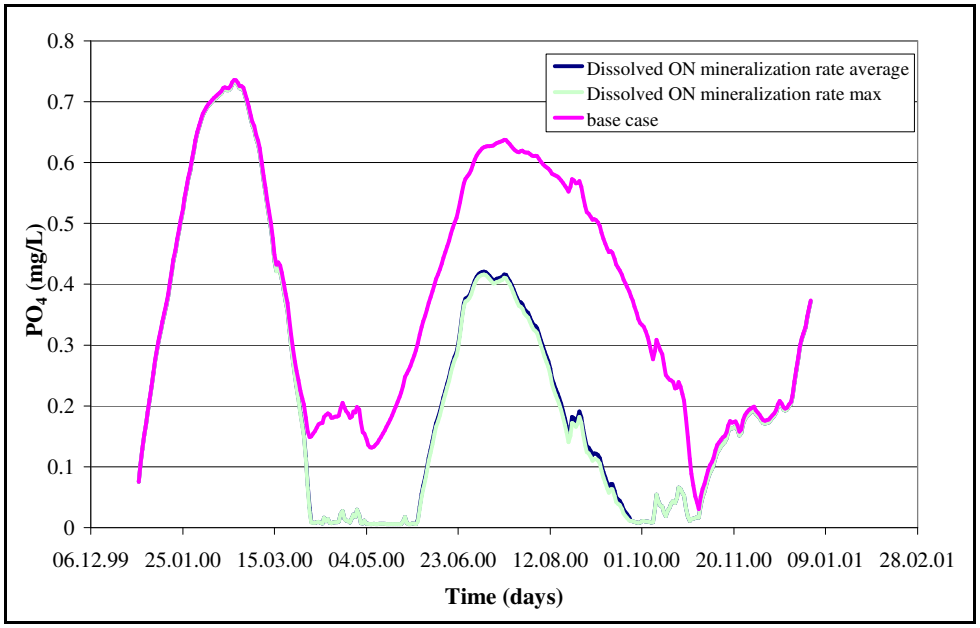


**Figure 4. 26** Effect of DON mineralization rate on $PO_4$ for Segment 2

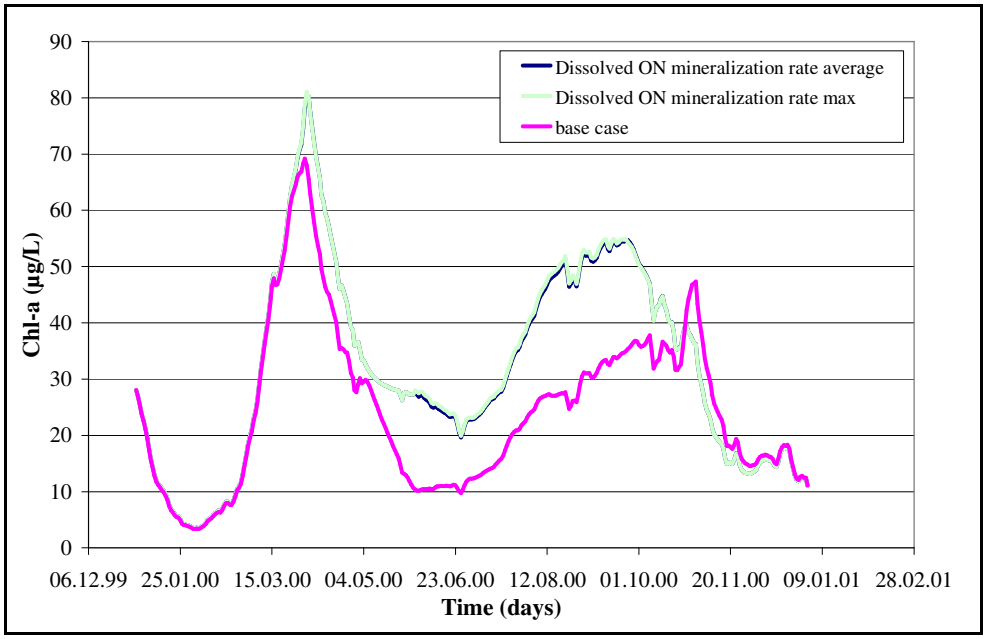


**Figure 4. 27** Effect of DON mineralization rate on Chl-a for Segment 2

The average Chl-a concentrations for the base and maximum rates were 24.41 µg/L and 31.80 µg/L, respectively (Figure 4.27). RMSE for Chl-a for average and maximum rates were 11.0848 and 11.3178, respectively. These results are indicative of nitrogen limitation at least in summer months. As $NO_3$ or DIN become available, $PO_4$ was used and algal growth was increased. Although a significant change in Chl-a concentrations was attained for different DON mineralization rates, the impact of this increase was not that significant on DO values as depicted in Figure 4.29. For average and maximum DON mineralization rate RMSEs for DO were 0.2757 and 0.2809, respectively. This is probably due to wind driven reaeration of the system.

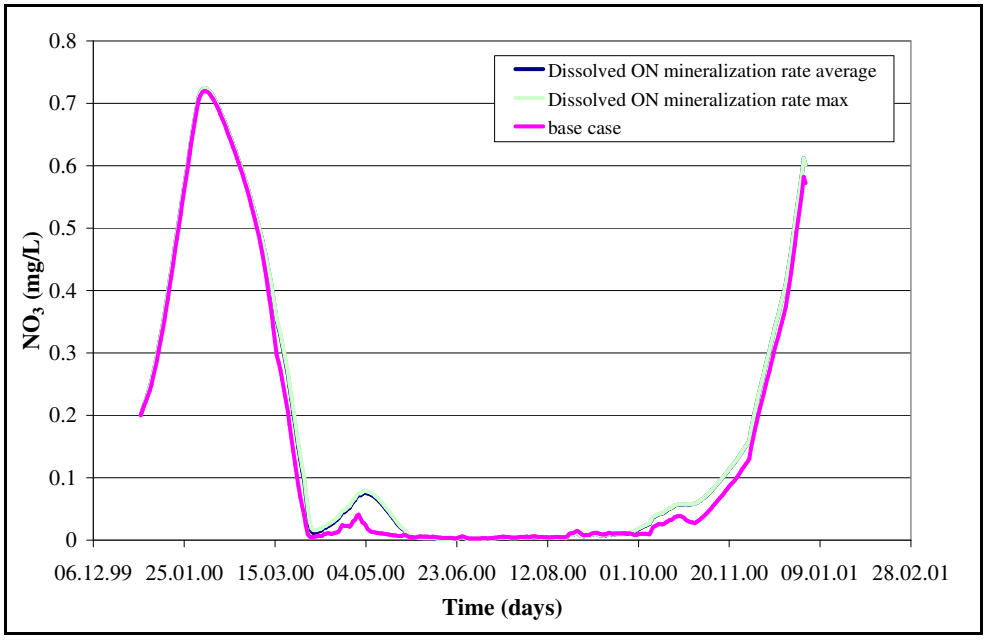


**Figure 4. 28** Effect of DON mineralization rate on $NO_3$ for Segment 2

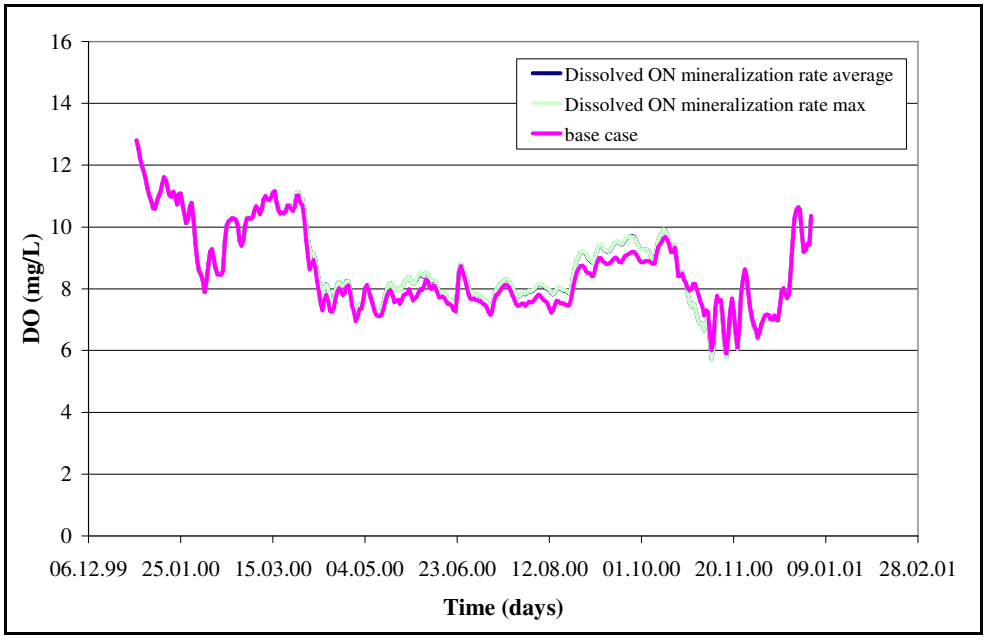


**Figure 4. 29** Effect of DON mineralization rate on DO for Segment 2

### 4.3.2 Sensitivity on Phytoplankton Maximum Growth Rate

Phytoplankton growth rate was increased and decreased by 25% and 50% to see the effect on water quality parameters. Chl-a RMSEs for modifications of +25%, +50%, -25% and -50% in growth rate were 5.5082, 9.3784, 6.4029 and 12.1333, respectively. As depicted in Figure 4.30, when the maximum growth rate was enhanced, Chl-a concentrations increased as expected. While the average Chl-a concentration was 24 µg/L for the base case, it was 30 µg/L when maximum growth rate was increased by 50%. As a result of phytoplankton uptake, $PO_4$ was decreased (Figures 4.32).

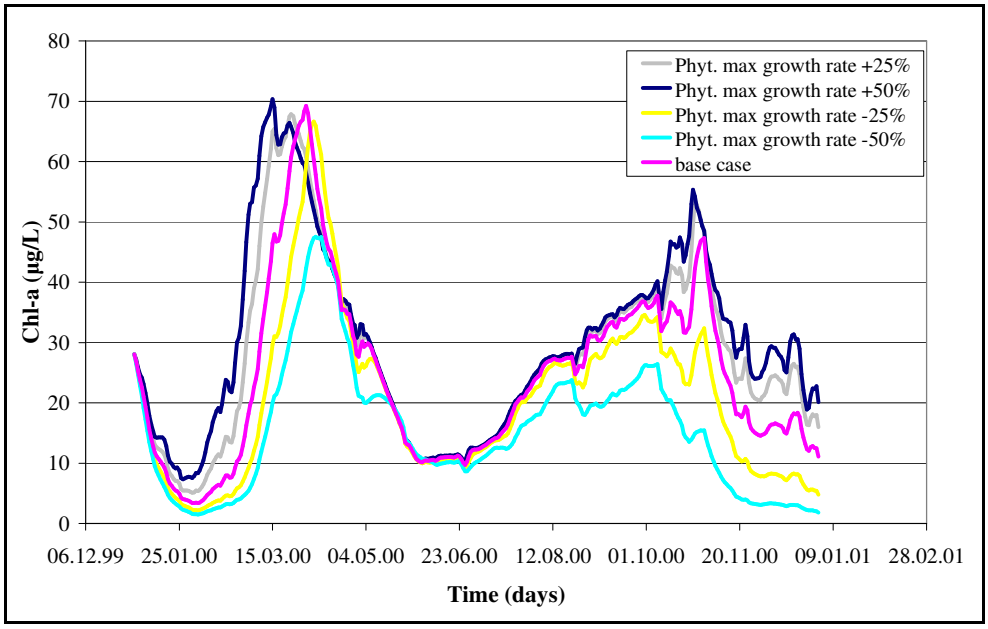


**Figure 4. 30** Effect of Phytoplankton Maximum Growth Rate on Chl-a for Segment 2

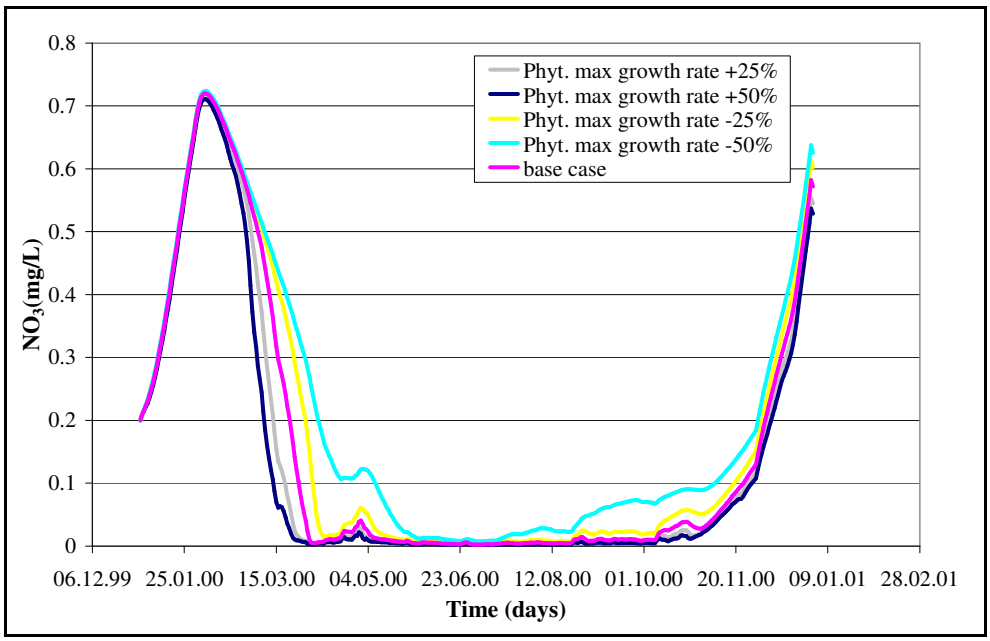


**Figure 4. 31** Effect of Phytoplankton Maximum Growth Rate on $NO_3$ for Segment 2

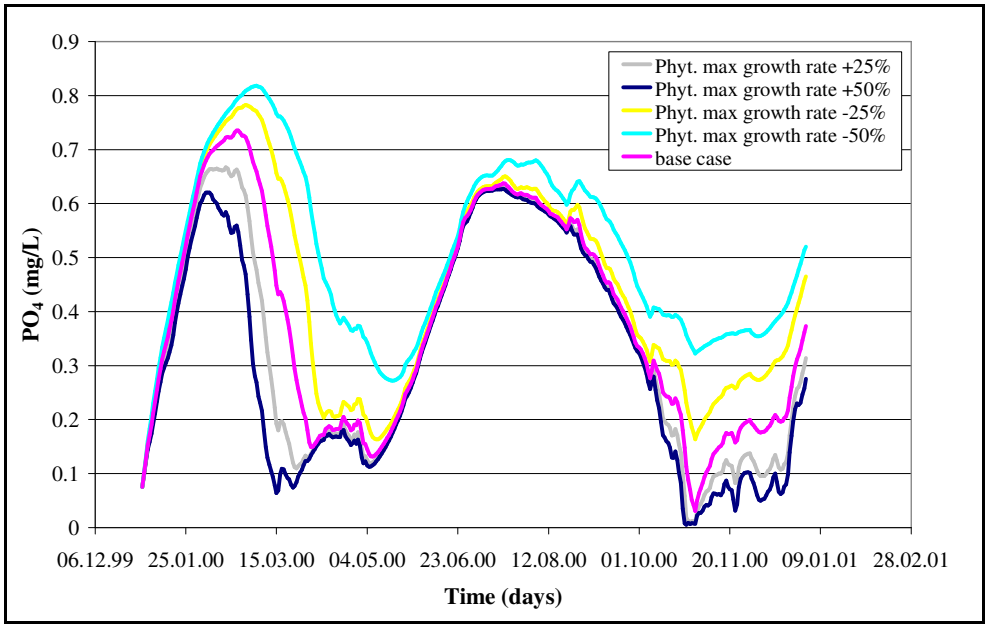


**Figure 4. 32** Effect of Phytoplankton Maximum Growth Rate on $PO_4$ for Segment 2

The decrease in $NO_3$ (Figure 4.31) was not as significant as it was for $PO_4$ since $NO_3$ was already scarce. When the growth rate was decreased and increased by 50%, RMSEs for $PO_4$ were 0.1642 and 0.1173. For nitrate these values were 0.0721 and 0.0597, respectively. This showed that the change in $NO_3$ and $PO_4$ concentrations were more pronounced when the growth rate was decreased rather than increasing it. When the maximum growth rate was decreased by 50%, $PO_4$ concentration increased from 0.38 mg/L to 0.51 mg/L.

The DO profiles obtained for different phytoplankton growth rates are depicted in Figure 4.33. Average DO concentration increased from 8.6 mg/L for the base case to 8.8 mg/L for 50% increase in growth rate. When the growth rate was decreased and increased by 50%, RMSEs for DO were 0.5257 and 0.4070, respectively. These results show that phytoplankton maximum growth rate is one of the important parameters that affect the concentrations of the water quality parameters for this system.

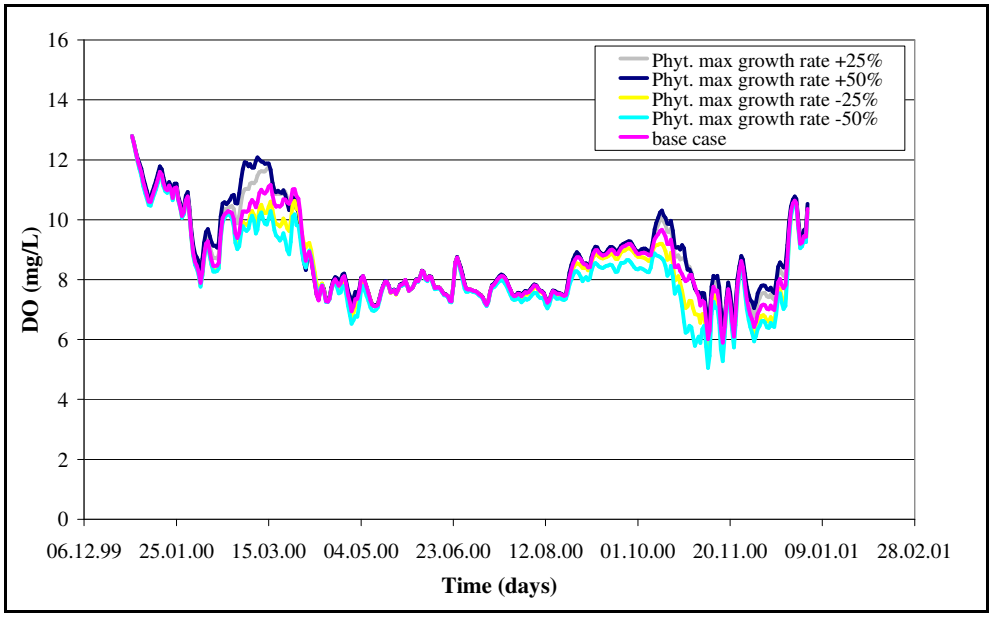


**Figure 4. 33** Effect of Phytoplankton Maximum Growth Rate on DO for Segment 2

### 4.3.3 Sensitivity on the Phytoplankton Nitrogen to Carbon Ratio

In WASP, the phytoplankton biomass is expressed as the Chl-a concentration. Using either a fixed or variable carbon to Chl-a mechanism, phytoplankton Chl-a is computed. Internal concentrations of phytoplankton nitrogen and phytoplankton phosphorus are calculated from the user-specified nitrogen to carbon (N/C) and phosphorus to carbon (P/C) ratios (Wool et al., 2001).

In this sense, phytoplankton N/C ratio was modified by ±25% and ±50% to see its impact on the water quality parameters. As phytoplankton grew, dissolved inorganic nitrogen is taken up and incorporated into the biomass. For every mg of phytoplankton carbon produced, inorganic nitrogen was consumed by N/C ratio (Wool et al., 2001). Figure 4.34 illustrates that when N/C decreases, the amount of inorganic nitrogen taken up by the phytoplankton decreases and, as a result, nitrate concentration increases in the system. Chl-a concentration in the

water column increases upon the decline in the N/C of the phytoplankton (Figure 4.35). This is due to the limited nitrogen species in the system at least for a portion of the simulation time. Since there is abundance of phosphorus, decreasing the required nitrogen content for growth gives rise to production. When the phytoplankton N/C ratio was decreased and increased by 50%, RMSEs for Chl-a concentrations were 11.8088 and 6.005, respectively.

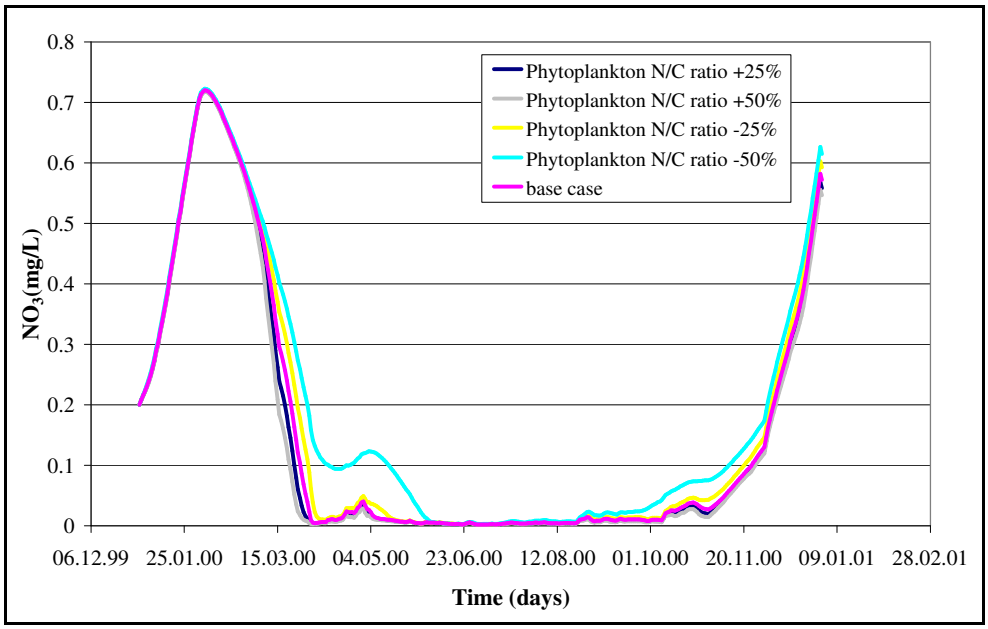


**Figure 4. 34** Effect of Phytoplankton N/C ratio on $NO_3$ for Segment 2

Raise in phytoplankton concentration as a result of decline in N/C ratio improved the DO concentrations as depicted in Figure 4.36. However, this in turn, decreased the $PO_4$ concentrations as a result of phytoplankton uptake (Figure 4.37). Average $PO_4$ concentration for Segment 2 decreased from 0.38 mg/L to 0.25 mg/L after N/C ratio was reduced by 50%. RMSEs for $PO_4$ for 50% reduction and 50% increase in N/C ratio were 0.1844 and 0.1007, respectively. Results showed that phytoplankton N/C ratio had a significant effect especially on Chl-a and nutrient concentrations.

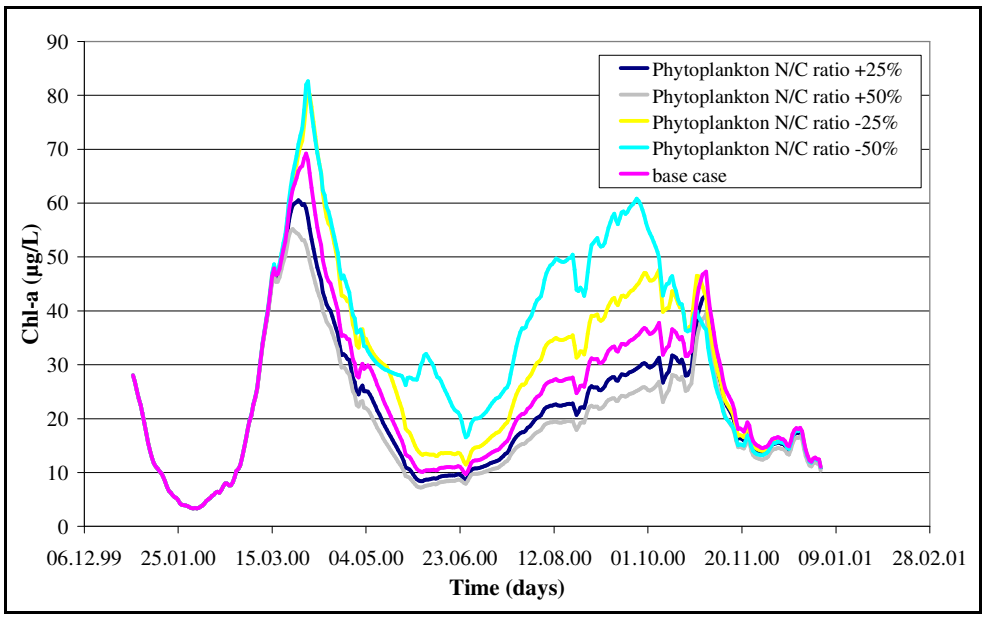


**Figure 4. 35** Effect of Phytoplankton N/C ratio on Chl-a for Segment 2

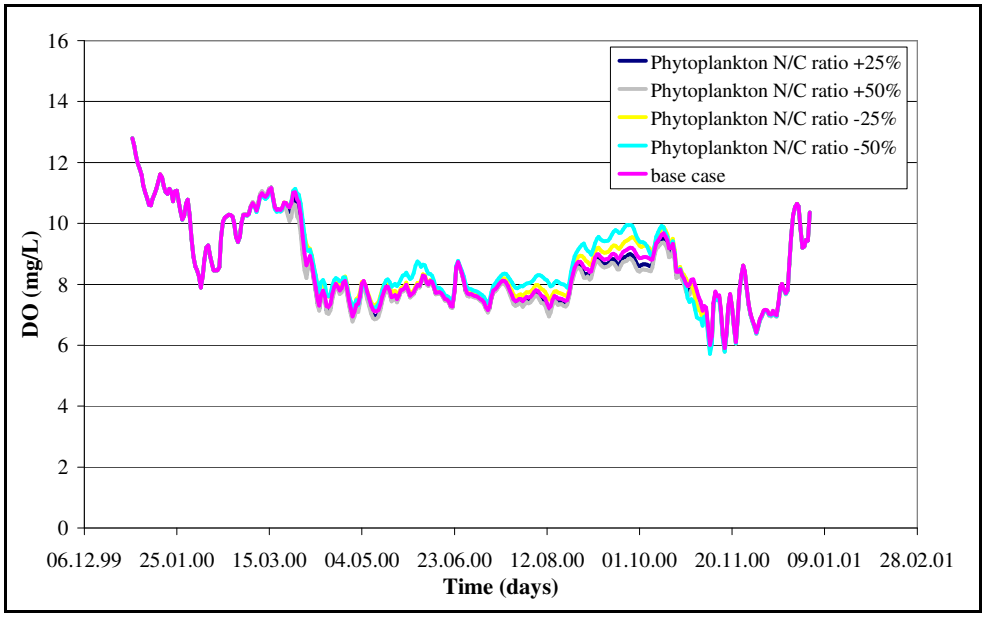


**Figure 4. 36** Effect of Phytoplankton N/C ratio on DO for Segment 2

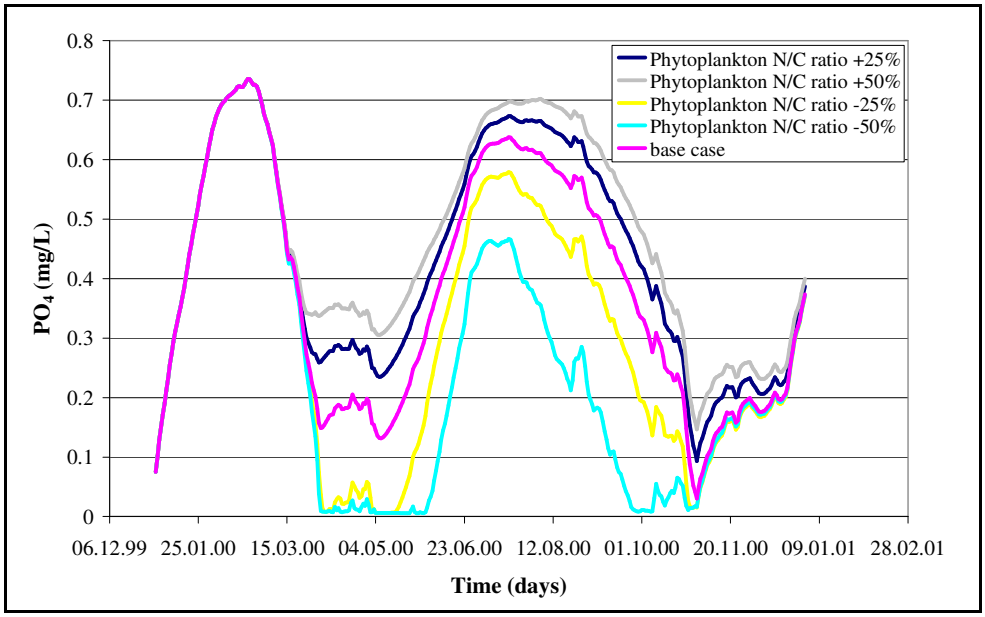


**Figure 4. 37** Effect of Phytoplankton N/C ratio on $PO_4$ for Segment 2

### 4.3.4 Sensitivity on Fraction of Phytoplankton Death Recycled to Organic Nitrogen

The fraction of dead and respired phytoplankton nitrogen that is recycled to the organic nitrogen pool can be specified in EUTRO. The effect of this constant on the system was analyzed by decreasing and increasing the constant by 25% and 50%. As given in Figure 4.38, increase in $NO_3$ concentration was observed when the fraction of phytoplankton death recycled to organic nitrogen was decreased. This situation can be attributed to the fact that, in the model, fraction of phytoplankton nitrogen recycled to inorganic nitrogen increases when the fraction of phytoplankton death recycled to organic nitrogen decreases. Therefore, Chl-a concentrations increased as $NO_3$ increased in the system as can be seen in Figure 4.39. For this case, as a result of enhanced production, decrease in $PO_4$ concentrations (Figure 4.40) and increase in DO (Figure 4.41) were observed as the fraction of phytoplankton death recycled to organic nitrogen was reduced. The change in $NO_3$, $PO_4$, Chl-a and DO concentrations with respect to

base case were more obvious when the growth rate was decreased. RMSEs for $PO_4$ for 50% reduction and 50% increase in fraction of phytoplankton death recycled to ON were 0.1801 and 0.1142, respectively. When 50% reduction and 50% increase were applied in recycle ratio, RMSEs for Chl-a concentrations were 10.2937 and 6.0820, respectively. For the same cases in order, RMSEs for $NO_3$ concentrations were 0.0275 and 0.0141, respectively.

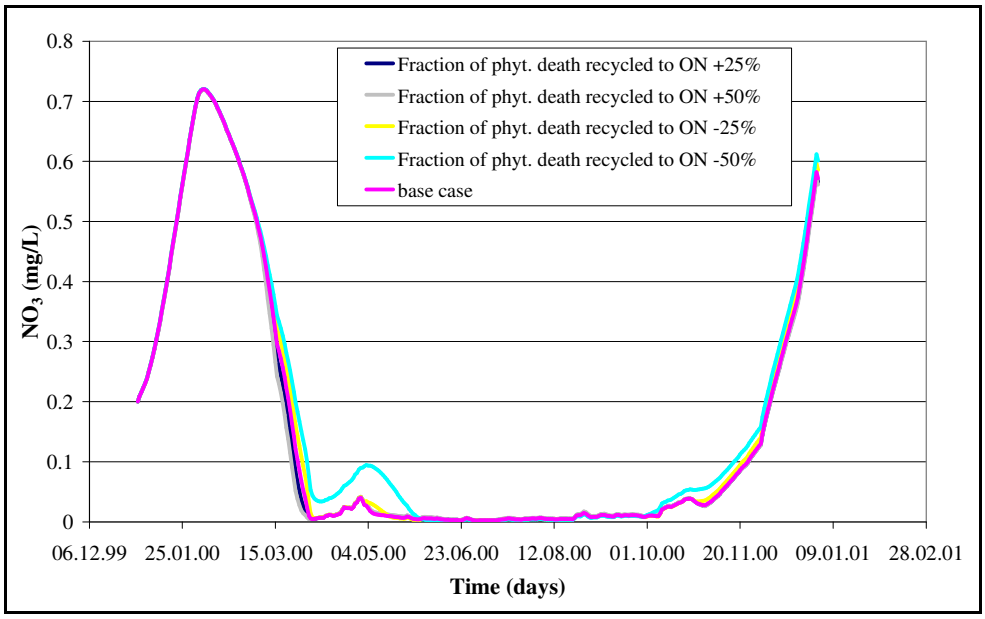


**Figure 4. 38** Effect of Fraction of Phytoplankton Death Recycled to Organic Nitrogen on $NO_3$ for Segment 2

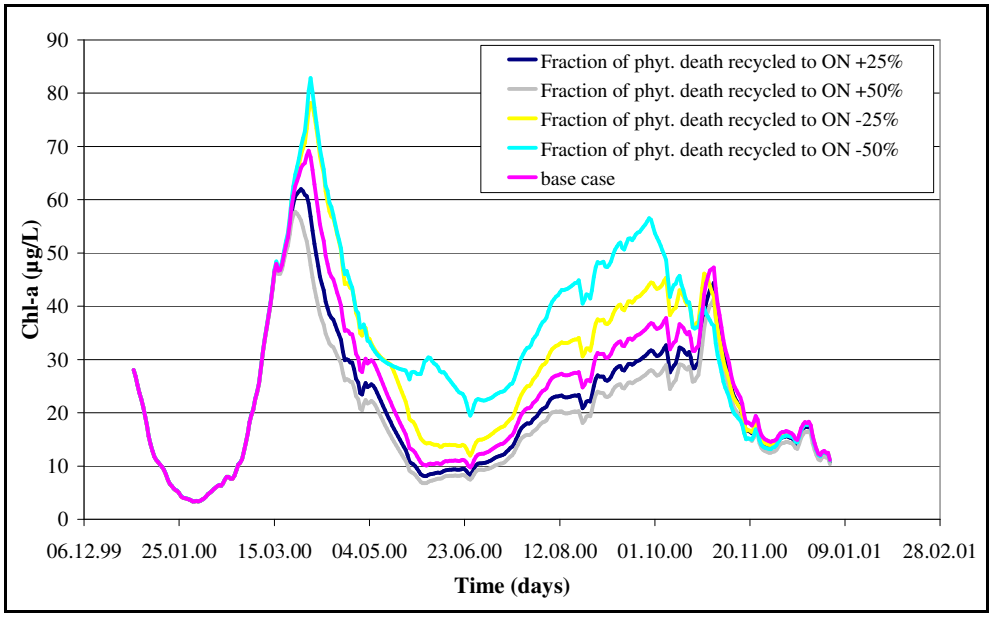


**Figure 4. 39** Effect of Fraction of Phytoplankton Death Recycled to Organic Nitrogen on Chl-a for Segment 2

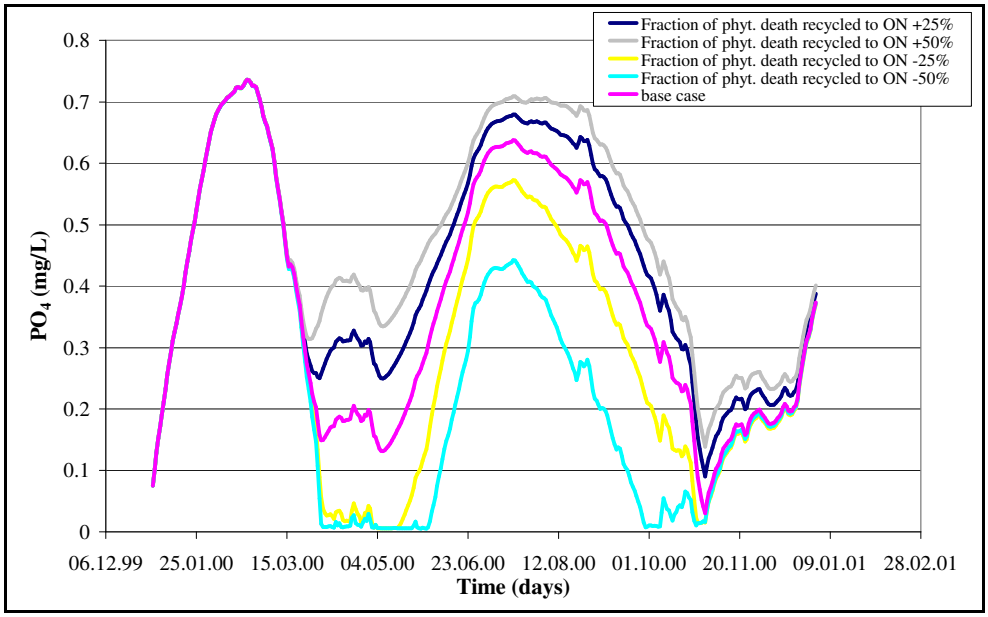


**Figure 4. 40** Effect of Fraction of Phytoplankton Death Recycled to Organic Nitrogen on $PO_4$ for Segment 2

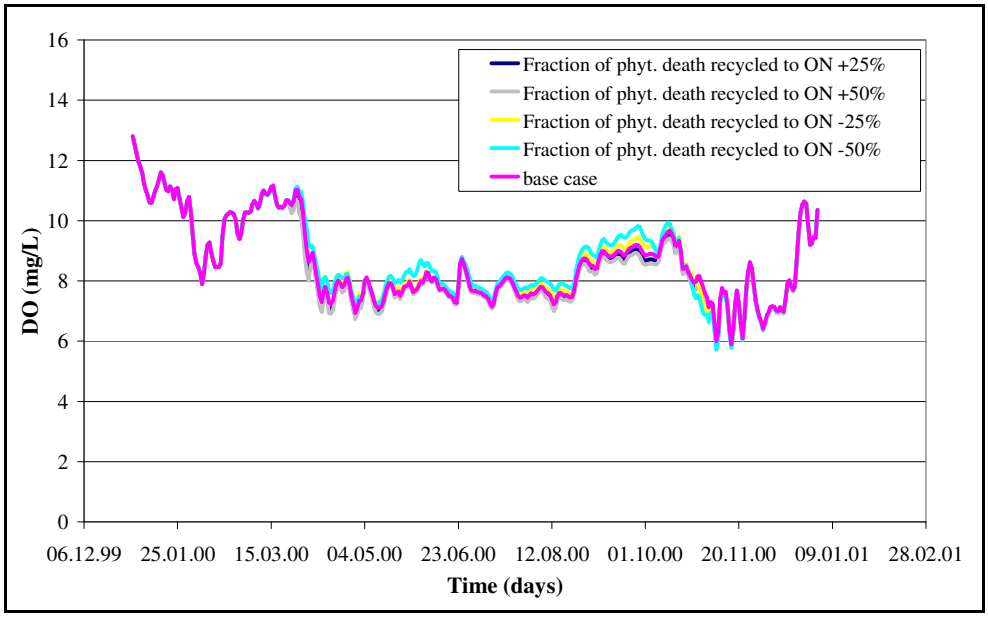


**Figure 4. 41** Effect of Fraction of Phytoplankton Death Recycled to Organic Nitrogen on DO for Segment 2

### 4.3.5 Sensitivity on the Sediment Oxygen Demand

The concentrations of oxygen in the overlying waters can be affected significantly by the decomposition of organic material in benthic sediment (Wool et. al., 2001). In general, sediment oxygen demand (SOD) can be determined by sediment analysis in laboratory, at site or through model calibration (Muhammetoğlu, 1998). As discussed before, in this study, few data were available to account for benthic activities on the water quality of the lake. Therefore, a typical SOD value was selected from literature (Wool et al., 2001) and defined as a uniform average value of 1.5 g/m$^2$.day for the lake. In order to examine the impact of this assumption, a sensitivity analysis was performed on SOD by modifying the quantity used in the model by ±25% and ±50%. The results are depicted in Figure 4.42. Average DO concentration for Segment 2 changed from the base value of 8.60 mg/L to 8.33 mg/L, 8.07 mg/L, 8.86 mg/L, and 9.12 mg/L for the adjustments of +25%, +50%, -25%, and -50%,

respectively. As SOD is decreased and increased by 50%, RMSEs for DO were 0.5624 and 0.5622, respectively. These values are close to RMSE value when phytoplankton maximum growth rate was decreased by 50%. Although SOD is an important parameter for DO levels, RMSE results show that its impact was not significantly more than the uncertainties in kinetic constants. This is probably due to the wind-driven reaeration. However, it is clear that determination of site specific SOD values are important to obtain better results.

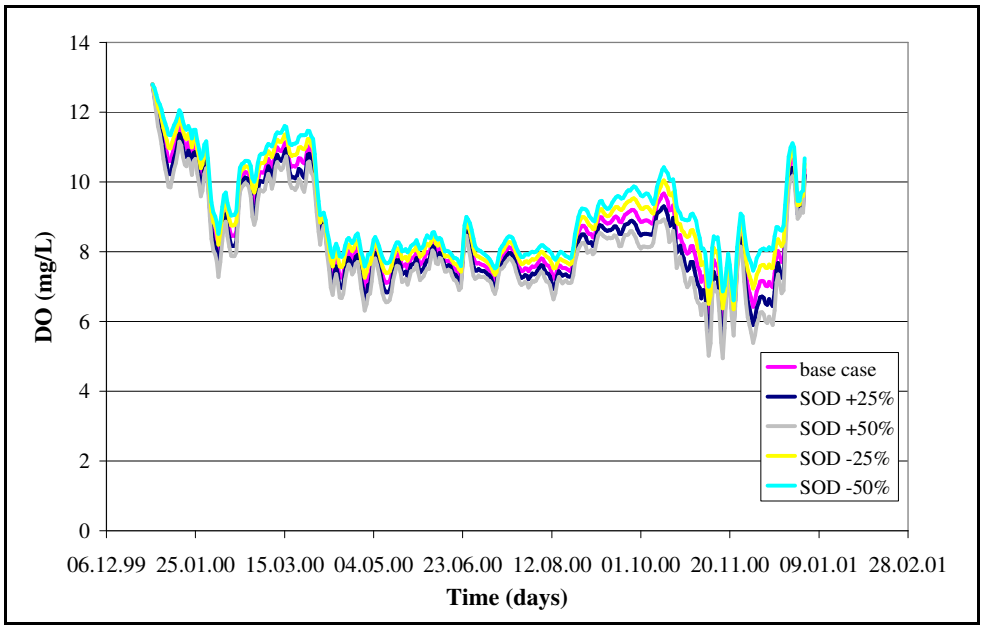


**Figure 4. 42** Effect of Sediment Oxygen Demand on DO for Segment 2

### 4.3.6 Sensitivity on the Wind Speed

One of the important constituents in the model is dissolved oxygen which has many complex reactions with all the other parameters. Main mechanism which affects DO is reaeration. Reaeration rate coefficient can be calculated using flow-induced or wind-induced rates (Muhammetoğlu, 1998). In this study, wind-induced rate was used. Wind speeds were decreased and increased by 50% to see

the effect of change of reaeration rate on DO (Figure 4.43). Average DO concentration for Segment 2 changed from the base value of 8.60 mg/L to 9.14 mg/L and 7.38 mg/L for the adjustments of +50% and -50%, respectively. When wind speed was decreased by 50%, RMSE for DO at segment 2 is 1.4943 which is the highest RMSE for DO compared to other examined constants. This result shows that wind speed is the most effective parameter on DO concentration for this system. Some variations between predictions and observations were observed during calibration and verification. These discrepancies can be minimized with better wind data collection. In this study, wind data was supplied from Bursa Meteorological Station which does not completely describe wind conditions at the site.

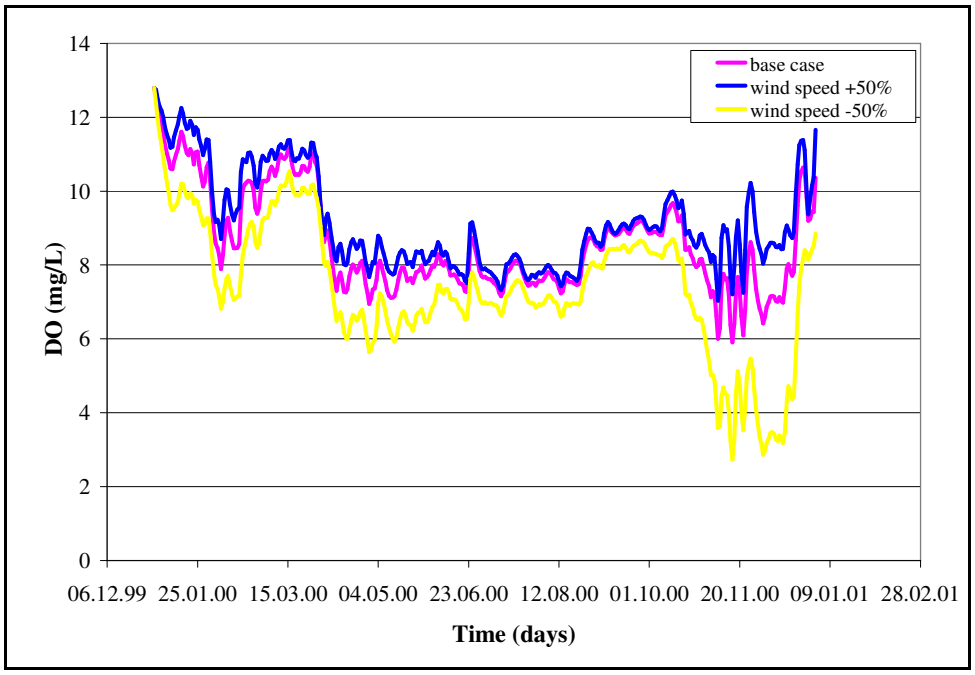


**Figure 4. 43** Effect of Wind Speed on DO for Segment 2

## 4.4 Impact of Management Scenarios on the Water Quality of the Lake

Water quality models are widely used in the world to understand and predict the water quality responses to natural phenomena, pollution stresses and remedial activity. The model used in this study is proven to produce fairly good predictions under different environmental conditions particularly for $NO_3$ and $PO_4$, as presented in the calibration and verification stages. Therefore, it can be concluded that, at least for these parameters which are very important in determination of the productivity and therefore eutrophic state of the lake, the model can be used to assess the impact of different management scenarios.

The management scenarios are based on limiting the phosphorus species. This is the preferred approach in eutrophication control since options related with nitrogen limitation cannot avoid the production of nitrogen fixing algae such as the blue-green ones. Most of the scenarios are based on a previous study (Bulut, 2005) that determined the phosphorus loads and their sources reaching to the lake. In that study, various scenarios were analyzed to reduce the phopshorus loads originating at the watershed scale. In this study, the impact of those reductions on the water quality of the lake was investigated. Results were compared to the ones obtained following the calibration and verification of the model, which is referred to as the base case.

### 4.4.1 Sediment Dredging

If there is a sediment layer which acts as a nutrient source for the lake, sediment dredging can be a method to decrease the nutrient inputs from the sediment layer to the overlying water column (Muhammetoğlu, 1998). Besides minimizing the sediment-oriented nutrient loads, sediment dredging can aid in the removal of SOD.

Sediment dredging was applied for a simulation period from 01.01.2000 to 01.01.2001 by removing phosphorus benthic flux and SOD for this study. The model results for DO, $NO_3$, $PO_4$ and Chl-a are presented in Figure 4.44. With the application of dredging, significant improvement was attained in the DO values. Especially, for the conditions set by the fall and winter of 2001, significant improvement was observed. While the average and the minimum of DO concentrations in Segment 2 throughout the simulation period for the base case was 8.6 mg/L and 5.9 mg/L, respectively, these values were 9.6 mg/L and 7.6 mg/L, respectively, following the sediment dredging. RMSE for DO at segment2 due to sediment dredging is 1.3116. According to this result, sediment dredging is important in terms of DO parameter.

Depending on the reduction of algal biomass as a result of nutrient reduction, the average $NO_3$ concentration was slightly reduced as well. The average nitrate concentration in Segment 2 decreased from 0.15 mg/L to 0.13 mg/L after sediment dredging. RMSE for $NO_3$ at segment 2 is 0.0727. Quantity of death phytoplankton recycled to organic nitrogen decreased due to low phytoplankton concentration. This caused decrease both in $NH_3$ and $NO_3$ concentrations because of diminished mineralization. Consequently, sediment dredging was also helpful to reduce $NO_3$ concentration in the lake.

The average $PO_4$ concentration in Segment 2 decreased from 0.38 mg/L to 0.13 mg/L. Similarly, the minimum concentration changed from 0.03 mg/L to 0.0038 mg/L and the maximum concentration altered from 0.74 mg/L to 0.61 mg/L. RMSE for $PO_4$ at segment 2 is 0.2902. Results show that sediment dredging has enormous impact on $PO_4$ concentrations.

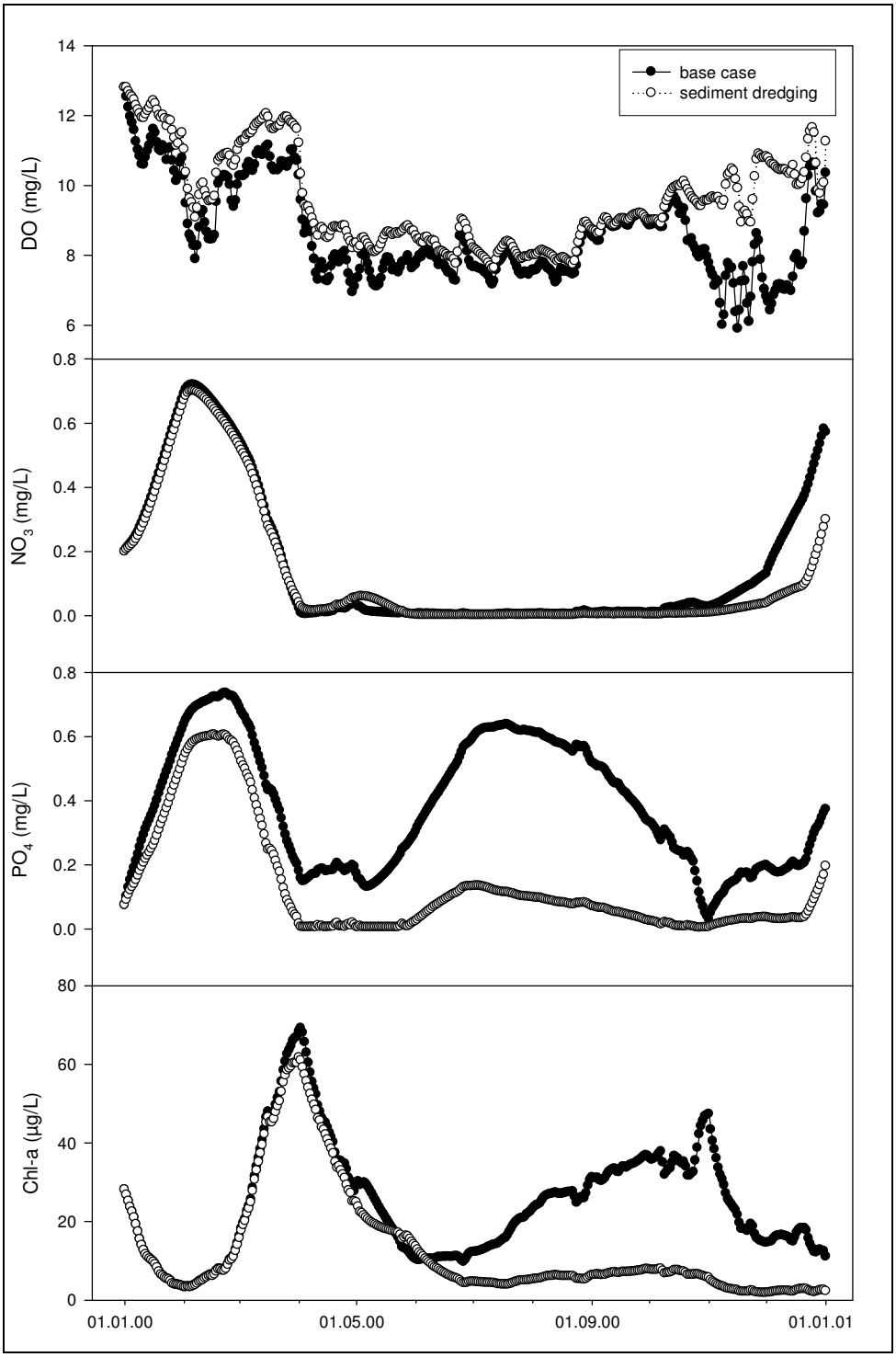


**Figure 4. 44** Effect of sediment dredging on DO, $NO_3$, $PO_4$ and Chl-a for Segment 2

Chl-a values were declined due to the reduction primarily in $PO_4$ and $NO_3$ concentrations. While the average Chl-a concentration for the base case was 24.41 μg/L, it was 13.09 μg/L after the application of sediment dredging. RMSE for Chl-a at segment 2 is 15.6911. Figure 4.44 showed that the decrease in Chl-a was mainly as a consequence of $PO_4$ reduction by sediment dredging.

Sediment dredging may not be a feasible action to take since the lake is a RAMSAR site and dredging may have a negative impact on the fishing activities. However, it is a fact that the depth of the lake is declining due to sediment transport. Therefore, in future, dredging can be an alternative at least when applied partially. This study indicated the importance of loads arising from the sediment in terms of the protection of the water quality of the lake. As a result, at least any further sediment load to the lake should be prevented.

### 4.4.2 Control of Phosphorus Load from Agricultural Sites Around the Lake

The fertilizer applied on agricultural sites over the watershed area was another source of nutrients for the Uluabat Lake. When the amount of fertilizer applied on agricultural sites were decreased by 20%, 30% and 50%, soluble phosphorus load (SOLP) to the Lake Uluabat decreased by about 6.5%, 9.7% and 16.2%, respectively (Bulut, 2005).

When SOLP load to Lake Uluabat was decreased by about 6.5%, 9.7% and 16.2%, $PO_4$ concentrations in the lake decreased with respect to base case as depicted in Figure 4.45. When the reduction in SOLP load increases, orthophosphate concentration decreases in the lake. When SOLP load to the Lake Uluabat decreased by about 6.5%, 9.7% and 16.2%, average orthophosphate concentration decreased from 0.38 mg/L to 0.35 mg/L, 0.34 mg/L and 0.31 mg/L, respectively. When SOLP load to the Lake Uluabat decreased by about 6.5%, 9.7% and 16.2%, RMSEs for $PO_4$ were 0.0319, 0.0476 and 0.0793, respectively. There was no notably impact of mentioned SOLP load

reductions on other water quality parameters of interest. For example, the average Chl-a concentrations were 24.4 µg/L for no SOLP reduction, 24.3 µg/L for 6.5% SOLP reduction, 24.3 µg/L for 9.7% SOLP reduction and 24.2 µg/L for 16.2% SOLP reduction. Additionally, respective RMSEs for Chl-a, $NO_3$ and DO at segment2 were 0.2517, 0.0005, 0.0142 for 6.5% SOLP reduction, 0.4072, 0.007, 0.0233 for 9.7% SOLP reduction and 0.7727, 0.0015, 0.0446 for 16.2% SOLP reduction. The average DO and $NO_3$ concentrations were 8.6 and 0.15 mg/L, respectively, for the base case and fertilizer usage reduction scenarios. Therefore, for the conditions studied, reducing the amount of fertilizer applied on agricultural fields around the lake did not have a significant impact on the water quality parameters despite the reduced in lake $PO_4$ concentrations.

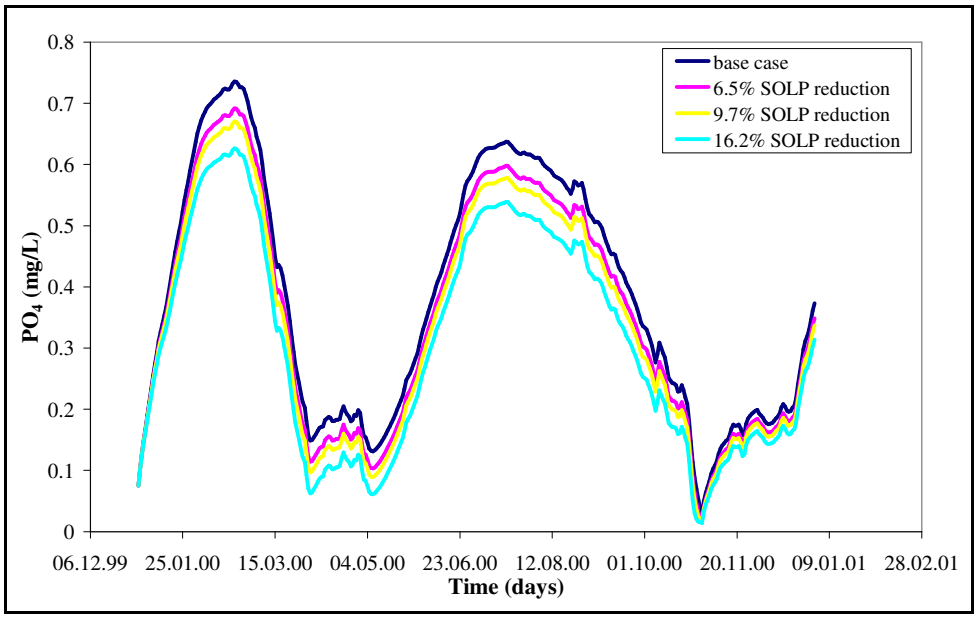


**Figure 4. 45** Effect of reduction in fertilizer application on $PO_4$ concentrations in Segment 2 (simulation period: 01.01.2000-01.01.2001).

### 4.4.3 Sediment Dredging and Phosphorus Control from Agricultural Sites

In order to test the synergistic impact of both fertilizer application reduction and sediment dredging, scenarios applied in Sections 4.4.1 and 4.4.2. The resulting water quality profiles for DO, $PO_4$, $NO_3$ and Chl-a are given in Figures 4.46, 4.47, 4.48, and 4.49, respectively. As can be observed from these figures, fertilizer reduction on top of sediment dredging slightly improved the results obtained for sediment dredging alone. The impact was mainly on the $PO_4$ concentrations. Average $PO_4$ concentration in Segment 2 was reduced from the base value of 0.38 mg/L to 0.12 mg/L, 0.11 mg/L, 0.10 mg/L by sediment dredging and 6.5%, 9.7%, 16.2% SOLP reductions, respectively. With only sediment dredging, this value was 0.13 mg/L. Respective RMSEs for DO, $PO_4$, $NO_3$ and Chl-a at segment 2 were 1.3038, 0.3010, 0.0730, 15.8576 for 6.5% SOLP reduction and sediment dredging, 1.2995, 0.3065, 0.0733, 15.9483 for 9.7% SOLP reduction and sediment dredging and 1.2908, 0.3179, 0.0744, 16.1607 for 16.2% SOLP reduction and sediment dredging.

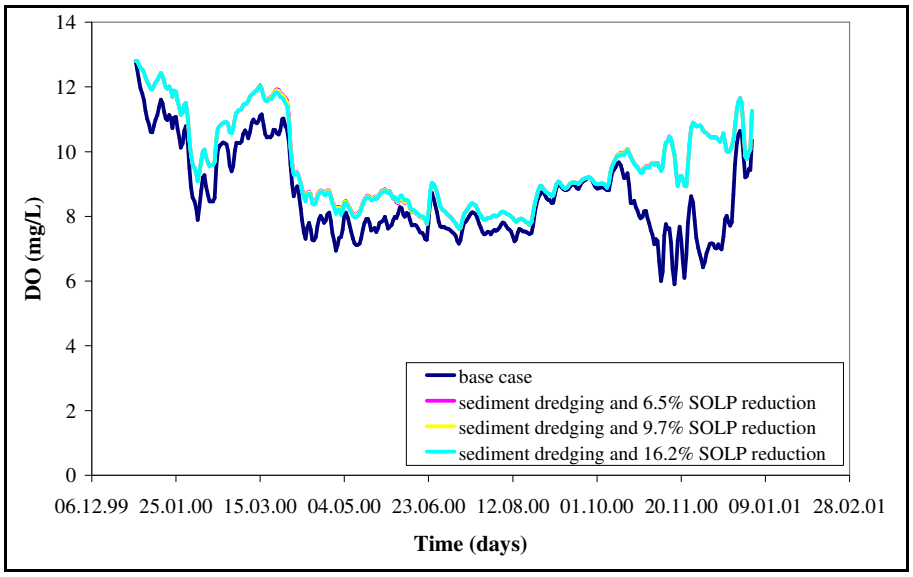


**Figure 4. 46** Effect of sediment dredging and phosphorus control from agricultural sites on DO for Segment 2

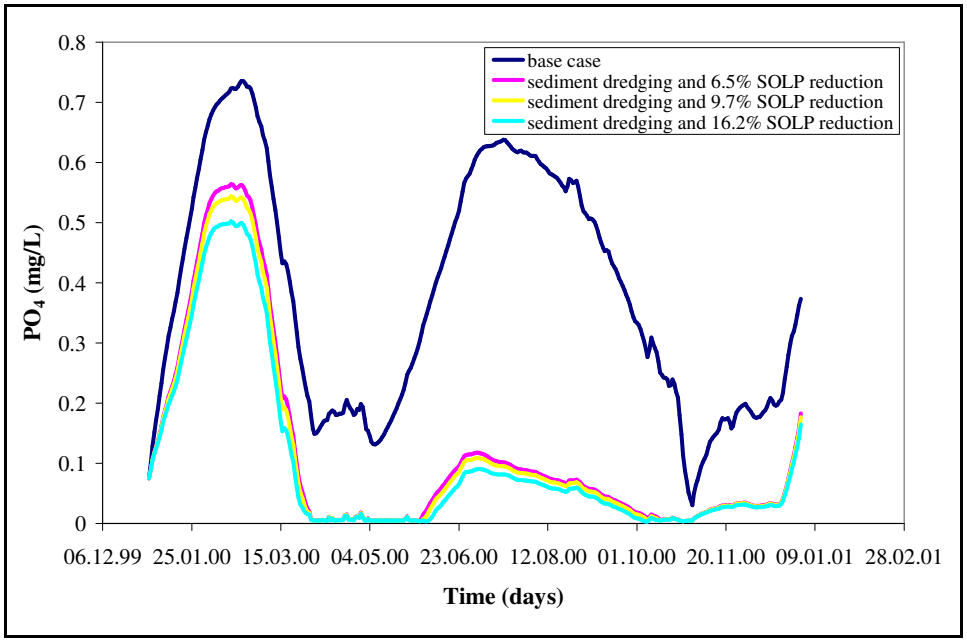


**Figure 4. 47** Effect of sediment dredging and phosphorus control from agricultural sites on $PO_4$ for Segment 2

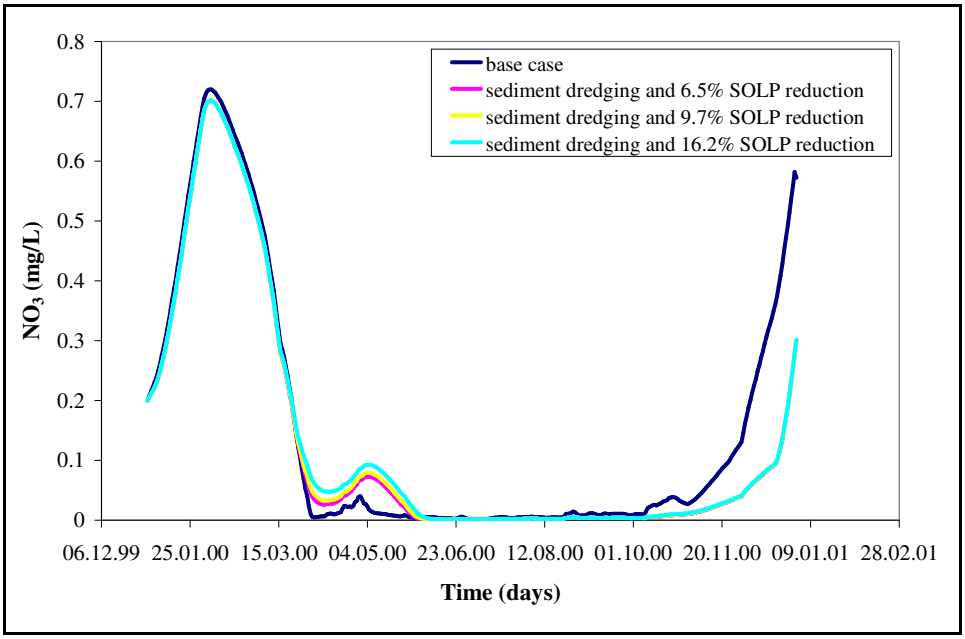


**Figure 4. 48** Effect of sediment dredging and phosphorus control from agricultural sites on $NO_3$ for Segment 2

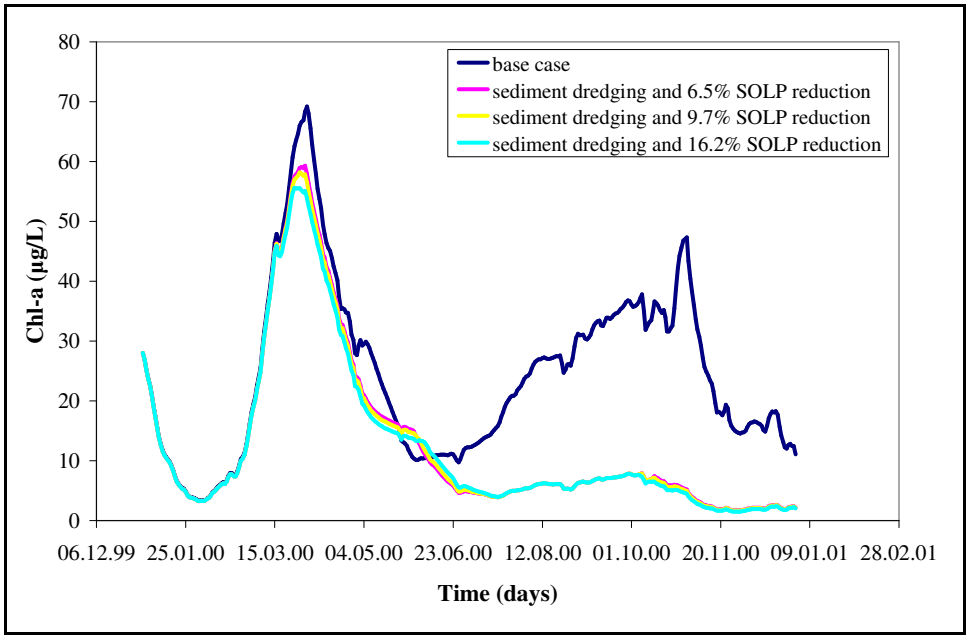


**Figure 4. 49** Effect of sediment dredging and phosphorus control from agricultural sites on Chl-a for Segment 2

### 4.4.4 Buffer Zone Application

In the study of Bulut (2005) the agricultural landuses on the areas surrounding the Lake Uluabat were removed and set with landuse of range-grasses to create a buffer zone around the lake. This approach resulted in 10% reduction in the overall SOLP amount transported to the lake. Simulation was performed that reflected this decrease in the total load entering to the lake in order to analyze the impact of this option on the water quality of the lake.

According to the output of the model, the only noticeable effect of buffer zone was observed on $PO_4$ concentrations in the lake (Figure 4.50). With the buffer zone around the lake, the average $PO_4$ concentration was decreased from 0.38 mg/L to 0.33 mg/L in Segment 2 for the simulation period. The maximum concentration decreased from 0.74 mg/L to 0.67 mg/L. The impact on other parameters was minimal. While the average Chl-a concentration in Segment 2

was altered from 24.4 µg/L to 24.3 µg/L due to phosphorus reduction, no noticeable change was observed in DO and $NO_3$ concentrations. When buffer zone was applied, RMSEs for DO, $PO_4$, $NO_3$ and Chl-a at segment2 were 0.0242, 0.0490, 0.0008 and 0.4226, respectively. According to these results, buffer zone application alone is not sufficient to improve the water quality of the lake.

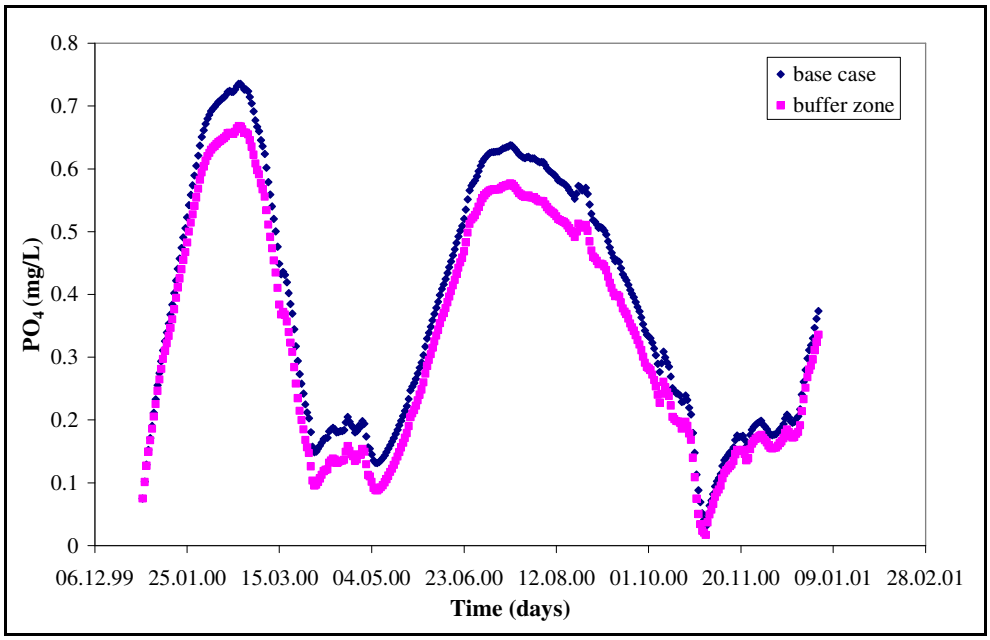


**Figure 4. 50** Effect of buffer zone around Uluabat Lake on $PO_4$ for Segment 2

### 4.4.5 Control of Phosphorus Coming from Emet and Orhaneli Streams

The pollutant loads originating from the Emet and Orhaneli Streams are carried through the Mustafakemalpasa Stream and reaches to the lake. When the phosphorus load coming from Emet and Orhaneli Streams was decreased by 20%, 30% and 50%, SOLP load to Lake Uluabat through the Mustafakemalpasa Stream was decreased by about 11.2%, 16.8% and 28.4%, respectively (Bulut,

2005). This scenario was employed to see the effect of phosphorus control in Emet and Orhaneli watersheds on the progression of the water quality in the lake. Results are given below.

As depicted in Figure 4.51, the average $PO_4$ concentration in Segment 2 for the period of simulation was reduced from 0.38 mg/L to 0.33 mg/L, 0.30 mg/L, and 0.25 mg/L with the decline in SOLP coming from Emet and Orhaneli Streams by 11.2%, 16.8%, and 28.4%, respectively. Compared to the change in $PO_4$, impact of phosphorus load reduction arising from Emet and Orhaneli Streams on DO, $NO_3$ and Chl-a were marginal as shown in Figures 4.52, 4.53, and 4.54, respectively. Respective RMSEs for $PO_4$, DO, $NO_3$ and Chl-a at segment2 were 0.0549, 0.0279, 0.0009, 0.4859 for 11.2% SOLP reduction, 0.0822, 0.0467, 0.0015, 0.8101 for 16.8% SOLP reduction and 0.1381, 0.0918, 0.0034, 1.6561 for 28.4% SOLP reduction.

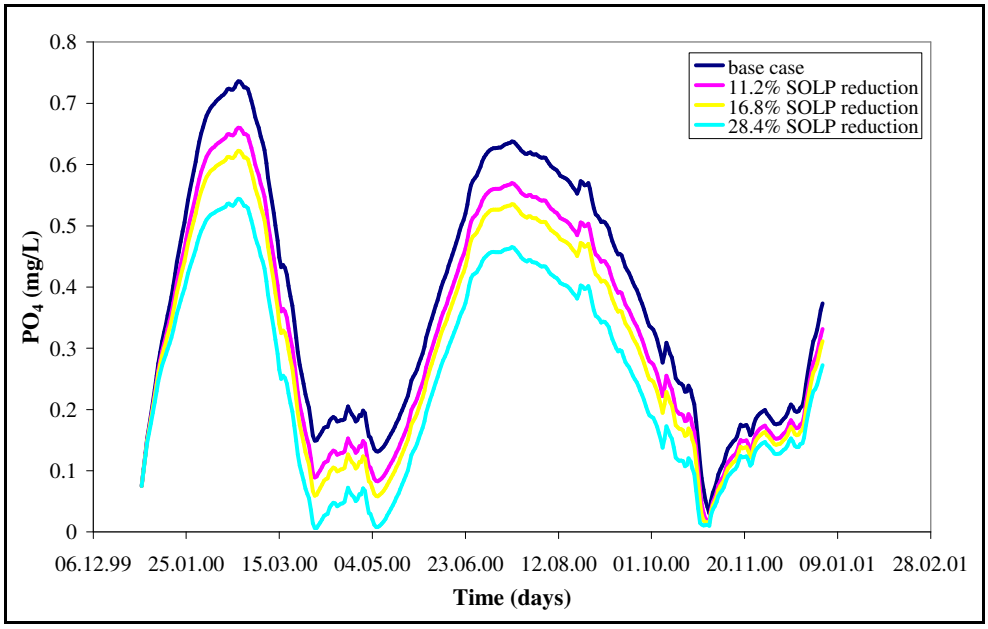


**Figure 4. 51** Effect of phosphorus load reduction from Emet and Orhaneli watersheds on $PO_4$ in Segment 2

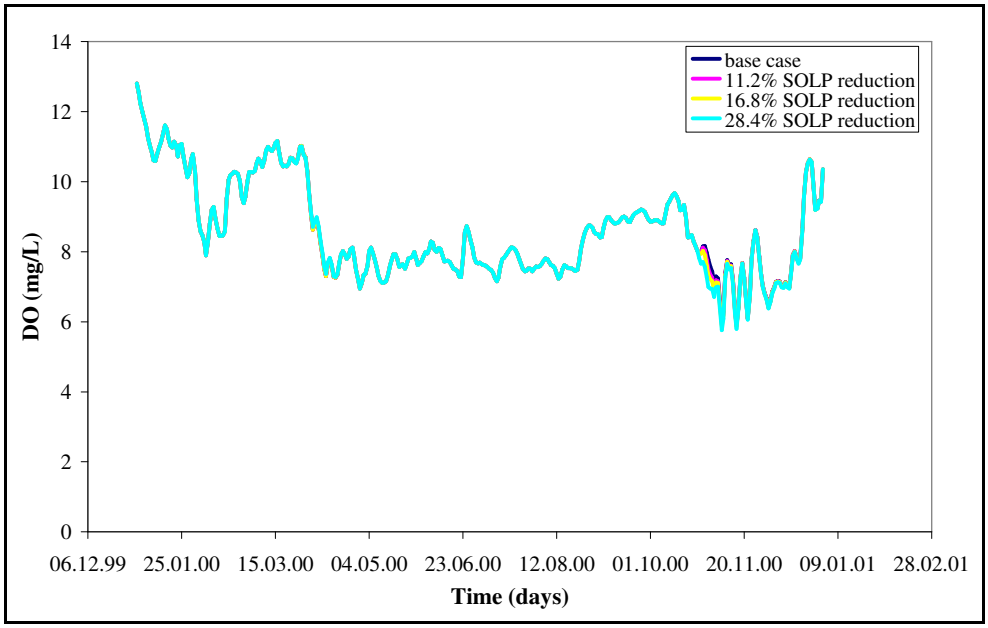


**Figure 4. 52** Effect of phosphorus load reduction from Emet and Orhaneli watersheds on DO in Segment 2

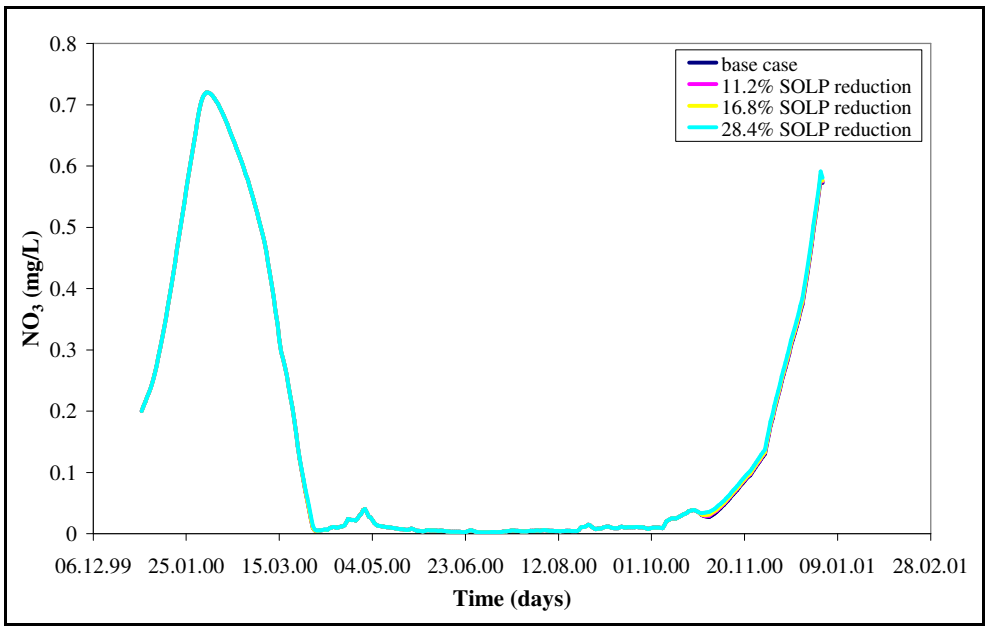


**Figure 4. 53** Effect of phosphorus load reduction from Emet and Orhaneli watersheds on $NO_3$ in Segment 2

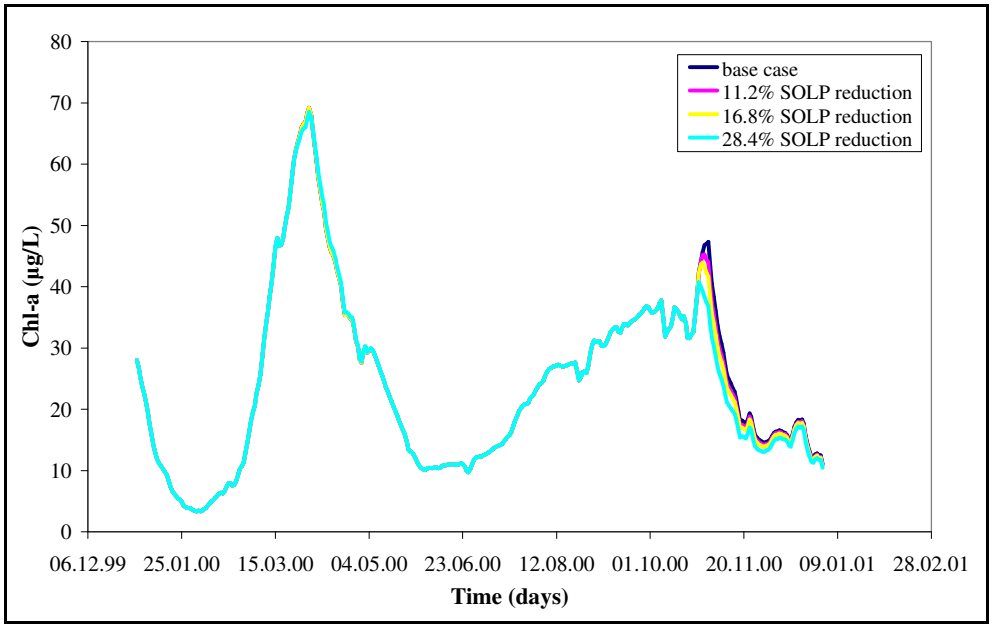


**Figure 4. 54** Effect of phosphorus load reduction from Emet and Orhaneli watersheds on Chl-a in Segment 2

### 4.4.6 Combination of Different Management Options

In this section the overall impact of a number of management options were considered. As in the study of Bulut (2005), the agricultural activities around the lake were replaced by range-grasses type landuse to form a buffer zone, phosphorus loads from the Mustafakemalpaşa District sewage system and Emet and Orhaneli Watersheds were decreased by 50%. In addition, 50% decrease was considered in the fertilizer usage through the agricultural lands. This scenario resulted in about 48% reduction in the overall SOLP load to Lake Uluabat (Bulut, 2005). On top of this, sediment dredging was applied.

According to the simulation results based on the above scenario, significant improvement was attainable in the water quality of the lake. Figures 4.55, 4.56, 4.57, and 4.58 depict the profiles for $PO_4$, $NO_3$, DO and Chl-a, respectively. In these figures results for the base case, combined management options and

sediment dredging alone are shown. In previous scenarios, sediment dredging had the most significant impact on the water quality. Therefore, the results for this scenario are depicted for comparison purposes.

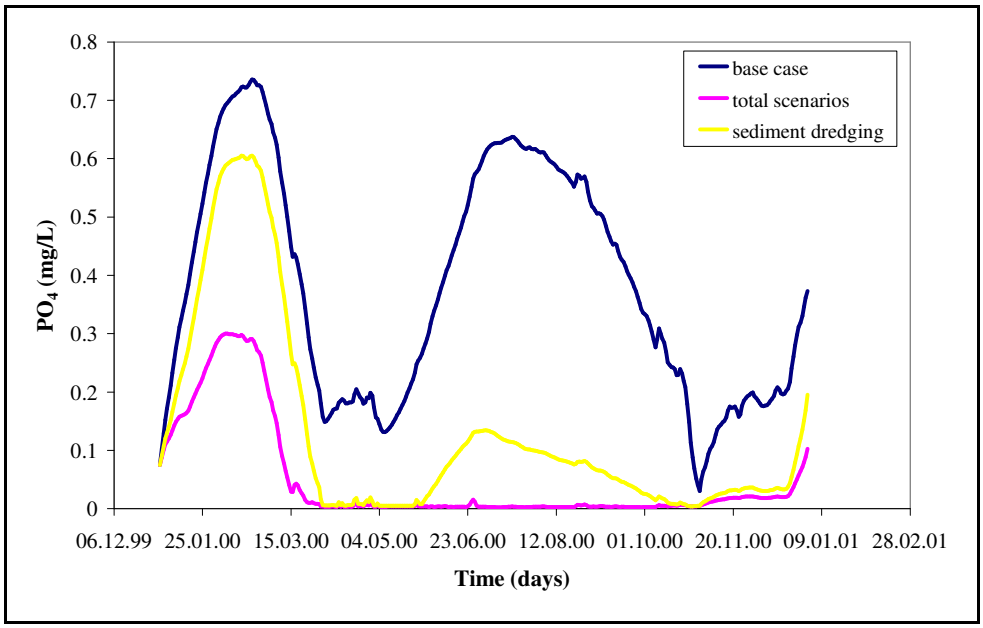


**Figure 4. 55** Effect of combined methods on $PO_4$

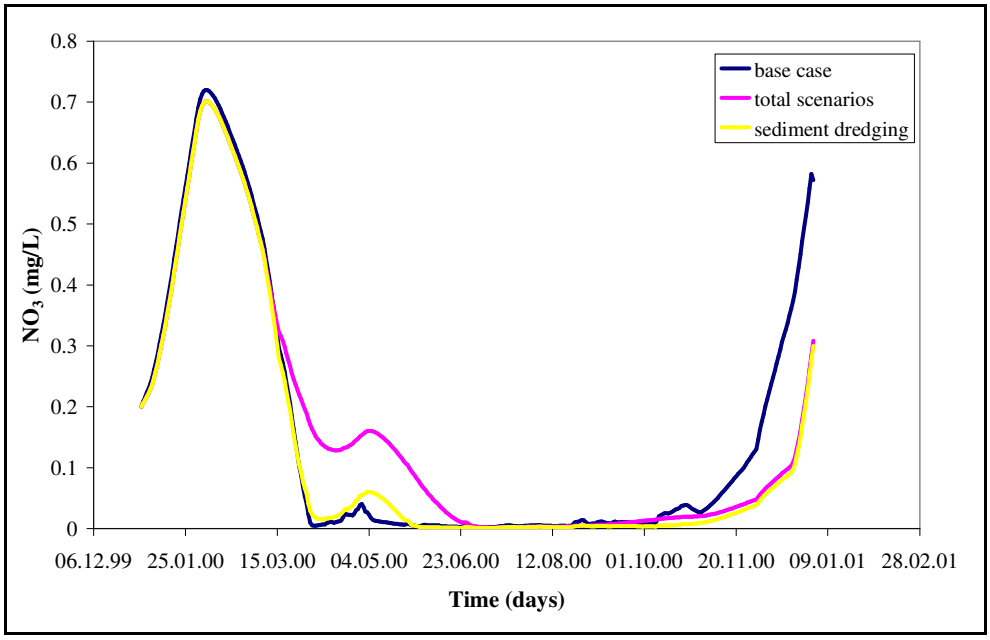


**Figure 4. 56** Effect of combined methods on $NO_3$

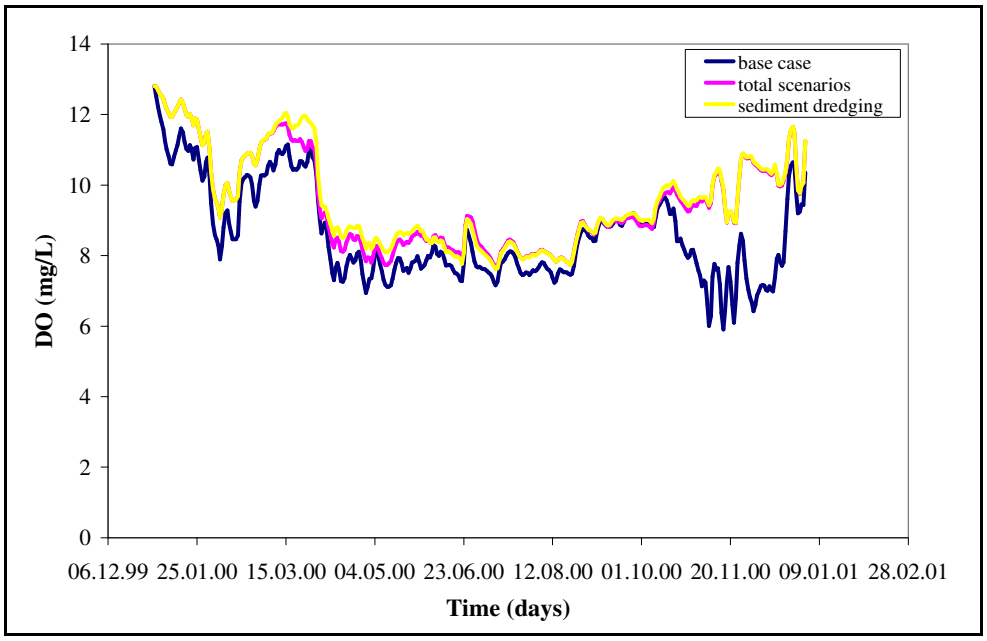


**Figure 4. 57** Effect of combined methods on DO

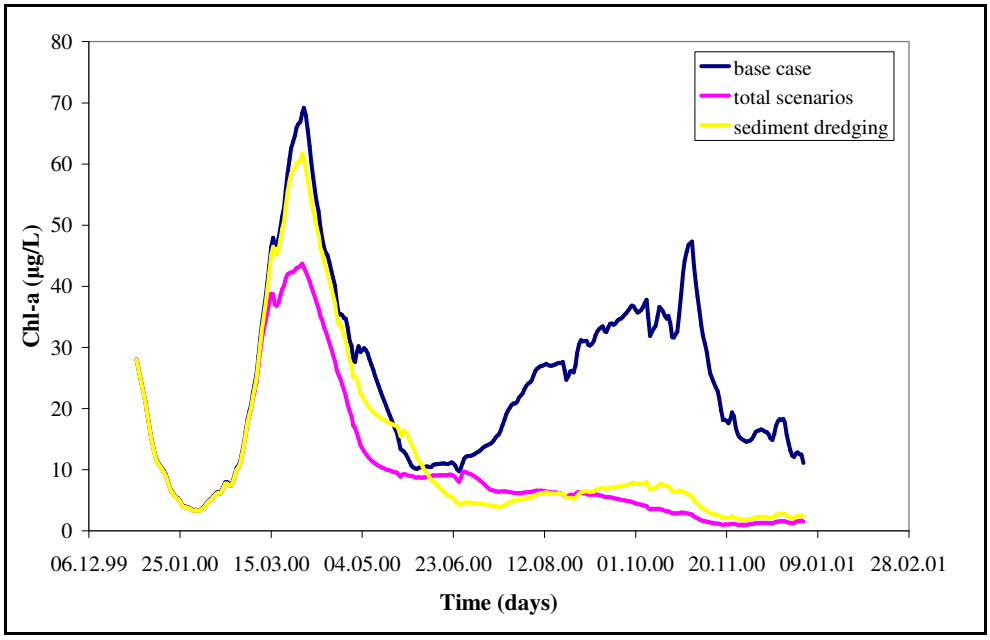


**Figure 4. 58** Effect of combined methods on Chl-a

As seen in Figure 4.55, combined methods for stress reduction in the lake resulted in significant $PO_4$ reduction in the lake. Average orthophosphate concentration is decreased significantly. The average $PO_4$ concentration in Segment 2 reduced from the base value of 0.38 mg/L to 0.05 mg/L. This value was 0.13 mg/L when sediment dredging was employed alone. Therefore, in this case, supporting the dredging with other methods had a positive impact on the decline of $PO_4$ concentrations.

Following the reduction in $PO_4$, algal production was minimized in relative to the base case (Figure 4.58). The average Chl-a concentration was reduced from 24.4 µg/L to 10.7 µg/L in Segment 2. As a result of reduction in algal production, $NO_3$ concentrations were decreased as well (Figure 4.56). Average nitrate concentration in Segment 2 was 0.154 mg/L. This is higher than the value (0.13 µg/L) for the case when only sediment dredging is applied. This may be due to the shift in the limiting nutrient as a result of reduced $PO_4$ concentrations.

Depending on the better quality conditions in the lake, DO concentrations were impacted as well (Figure 4.57). The average DO concentration for the base case was 8.6 mg/L. This value boosted up to 9.5 mg/L when combined methods were applied. As a result, in overall, combined methodologies had a very positive impact on the quality of the lake. When combined method was applied, RMSEs for $PO_4$, $NO_3$, DO and Chl-a were 0.3740, 0.0878, 1.2488 and 17.7242, respectively.

The concentrations reached by the combined methods were analyzed with respect to the Water Pollution Control Regulation put into force by the Ministry of Environment and Forestry. The average DO and $NO_3$ concentrations at the end of the simulation period satisfied the Class I criteria. The average $PO_4$ concentration met the requirement set by Class II. General Trophic Classification of Lakes and Reservoirs (Wetzel, 2001) was taken into consideration in order to evaluate the trophic state of the lake based on the average Chl-a concentration. According to this comparison, the lake was at the border of mesotropic and eutrophic range.

# CHAPTER 5

# CONCLUSIONS AND RECOMMENDATIONS

## 5.1 Conclusions

In this study, the water quality of Uluabat Lake was simulated by using Water Quality Analysis Simulation Program (WASP Version 7.2). Data used in this study was composed of meteorological data, flow data, water quality data belonging to segments and boundaries, segment properties and bathymetry of the lake. After model input setup, calibration and verification of WASP was performed for dissolved oxygen (DO), chlorophyll-a (Chl-a), ortho-phosphate ($PO_4$-P) and nitrate ($NO_3$-N). An acceptable similarity was generally obtained between the predicted and observed water quality values for both calibration and verification. Some mismatches of the model DO predictions in verification may be due to the wind driven reaeration option used in this study. Another reason can be underestimation of SOD. As a result, SOD should be evaluated well and in-situ measurements or lab experiments should be conducted for the improvement of model predictions. In verification process, it was not possible to capture the increase in the Chl-a concentrations in the summer of 1999. This can be attributed to the limitation of phytoplankton growth due to lacking of nitrogen. In this system, nitrogen fixation by algal species can be underestimated.

Following the calibration and verification of the model, sensitivity analysis was performed on the model outputs. As a result of the sensitivity analysis, dissolved organic nitrogen mineralization rate, phytoplankton maximum growth rate, phytoplankton nitrogen to carbon ratio, fraction of phytoplankton death recycled to organic nitrogen are the relatively higher sensitive parameters for water quality parameters interest.

Following the sensitivity analysis, the impact of different management scenarios on water quality conditions for the lake was assessed. For this purpose, sediment dredging, control of phosphorus load from agricultural sites around the lake, the synergistic impact of both fertilizer application reduction and sediment dredging, buffer zone application on the agricultural areas surrounding the Lake Uluabat, control of phosphorus coming from Emet and Orhaneli Streams and overall impact of already mentioned scenarios on water quality of the lake was evaluated. According to results of management scenarios, sediment dredging alone had the most significant impact on the water quality. Furthermore, supporting the dredging with other methods had a positive impact on the decline of $PO_4$ and Chl-a concentrations in the lake. However, sediment dredging can damage the biodiversity in the sediment and can cause depletion of fish in the lake. Fishery is an important economic supply for residents around the lake. Therefore, it is necessary to take public opinion regarding environmental problems and to solve them.

Subsequent to sensitivity analysis, the water quality status of Uluabat Lake in 2005 was determined by using monitoring data obtained from field work. Then, the progression of the general trophic status of the lake and the limiting nutrient for the system within years was evaluated. Results show that different nutrients became limiting in Lake Uluabat for different years. In 2000, the lake was nitrogen limited. Beginning by 2002, the limiting nutrient shifted to phosphorus. This can be due to implementation of the Lake Uluabat Management Plan conducted between 1998 and 2002 by the Turkish Ministry of Environment and Forestry and Association for the Protection of Nature (DHKD).

In lake restoration, it should be noted that whichever nutrient may be a limiting factor or has driven the process, the algal growth can be reduced by severely restricting the supply of only one of them. It is however, far easier to control phosphorus in external sources than nitrogen due to its low solubility and high

settling rate. In a lake receiving nutrients for many years, a much greater build up phosphorus is expected than nitrogen due to releases from sediments. Then practical solution is to attempt nutrient control first through phosphorus control (Celtemen, 1998). Main sources of phosphorus in Uluabat Lake are fertilizers used in farmlands around the lake and industrial discharges. To improve the water quality in the lake, amount of nitrogen and phosphorus coming to the lake should be reduced. For this purpose, application of some measures such as changes in agricultural practices, prevention excess fertilizer use by establishing farmer-training programs and control of direct effluent discharges from industrial facilities are necessary. However, a management strategy in a watershed scale is essential for control of P load to the lake. Although agricultural activities may have great impact on the pollution of the lake, other point and non-point sources should be considered as well (Bulut, 2005).

In conclusion, sediment dredging alone is the most effective alternative among the ones analyzed in this study for the conditions considered. In order to improve the water quality of the lake stated by oligotrophic class a number of management alternatives may be necessary in addition to sediment dredging and mentioned other managements. However, it should be noted that it was not possible to include the nitrogen fixing algae and sediment reactions in the model. So, the results obtained in this study are based on the conditions considered or assumed. But, on the average, it is deemed that fair results have been obtained especially for the nutrient concentrations.

### 5.2 Recommendations

In the model application of Lake Uluabat, a water quality network of 5 horizontal main segments and two layers (a total of 10 segments) was adopted. Benthic segments at the bottom merely acted as a convenient sink for settling of organic matter. Model calculations within this segment were ignored by using the intermediate eutrophication kinetics in WASP (Wool, 2001) due to the

insufficient data to simulate the benthic processes. However, sediment layer and the interactions of the layer with the overlying water column is an important part in the model setup for DO process, nutrient cycles and algal growth. Determination of site specific sediment characteristics such as sediment oxygen demand, benthic nutrient fluxes, deposition and uptake of nutrients are recommended to be determined either in situ or in laboratory for better model predictions (Muhammetoğlu, 1998).

The model could predict generally DO constituents well. However, some mismatches were observed in verification. This discrepancy is due to wind driven reaeration process. Wind is a very important parameter for DO reaeration process especially in shallow lakes. Wind velocities change widely depending on space and time. For this reason, wind data at field should be collected at the same time with all the water quality field measurements to obtain more accurate predictions.

In this study, simulations were performed with limited data. Therefore, more detailed water quality monitoring in Lake Uluabat can provide more data to the model. By this way, more precise results can be obtained and better management practices can be developed for Lake Uluabat.

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