ABSTRACT

KIM, HYUN WOO. Evaluation of Hydrology in an Agricultural Watershed and Nitrogen Removal by Constructed Wetlands. (Under the direction of Dr. Stephen W. Broome and Dr. Devendra M. Amatya.)

Excessive nitrogen loading has been considered a major cause of water quality problems in eastern North Carolina. There has been a particular concern in coastal watersheds because agricultural and forested lands are located adjacent to recreational and environmentally sensitive waters. The key to nutrient management at the watershed scale is the understanding and quantification of the fate of nutrients at the field scale and after nutrients enter the aquatic environment. The constructed wetland, located at the outlet of a watershed, is recognized as an efficient and environmentally safe wastewater treatment system. There is no accepted method to describe and predict nitrogen transformation in canals and streams. Many studies have focused on the efficiency of constructed wetland systems for wastewater treatment, but the results still vary. The objectives of this research were to investigate the effects of in-stream processes in agricultural watersheds of the lower coastal plain, to determine nitrogen transformations in constructed wetlands, and to propose a modeling approach for quantifying nitrogen transport and in agricultural watersheds and constructed wetlands.

The first step was a review of the literature on nitrogen retention in agricultural streams, nitrogen transformation in constructed wetlands, and design criteria of constructed wetlands for efficient nitrogen removal. In the second step, the daily outflow from a 1037 ha agricultural watershed at Open Grounds Farm was evaluated using the DRAINWAT, DRAINMOD for Watershed, in the lower Coastal Plain, Carteret County, North Carolina. As a third step, vegetation succession and efficiency of nitrogen removal was monitored, and a sequential model of nitrogen transformations in wetlands was evaluated.

In order to estimate nitrogen loading from the agricultural watershed, accurate simulation results of hydrology are required. However, the watershed was not intensively instrumented for hydrologic simulation at the period of our study. Mainly due to

problems with flow meters and lack of sufficient measured data, the hydrology simulation results were not adequate to estimate nitrogen loading. For this reason, nitrogen loading was introduced only in the methodology section, and this study focused on the sensitivity analysis using four input parameters: Manning's roughness coefficient, maximum depressional storage, calculation methods of potential evapotranspiration (PET), and channel bedslope. PET calculation methods were found to be most sensitive among those four parameters. Two Thornthwaite based methods using different correction factors (Thorn 1 and Thorn 2) were tested. Thorn 2 correction factors obtained from the agricultural watershed were more accurate than Thorn 1 correction factors, which were averaged from two agricultural and one forested watershed.

Vegetation succession in the wetland cells and nitrogen removal efficiency of constructed wetland system were monitored in chapter 3. Three hydrophyte species including *Juncus roemerianus*, *Cladium jamaicense*, and *Spartina alterniflora* were successfully established in the constructed wetland, but the growth rate of *J. roemerianus* and *C. jamaicense* was significantly faster than that of *S. alterniflora* in this new environmental condition. Good removal efficiencies of constructed wetland system for all nitrogen species were observed. A simple, sequential model of nitrogen transformation model was tested at the inlet of constructed wetlands using limited measured flow data in selected periods. Simulation results were not yet reasonable due to the limited number of sample data, a long waterway before the first cell of the constructed wetland, and non-calibrated average rate constant of the model. This model may predict the concentration of nitrogen species successfully with proper calibration with more measured data.

Evaluation of Hydrology in an Agricultural Watershed
and Nitrogen Removal by Constructed Wetlands

by
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BIOGRAPHY

Hyun Woo Kim was born in Seoul, South Korea on June 14, 1969 (April 30, 1969 for lunar calendar) and grew up in the same city. After high school, he went to the Korea University, in Seoul in 1988. He got his Bachelor's and Master's degree at the Department of Forest Resources in Korea University from 1988 to 1998. After that, he came to Raleigh to continue his study at Soil Science Department in North Carolina State University under the supervision of Dr. Stephen W. Broome. Right after he passed the final oral exam of his Ph.D. degree conditionally, he was accepted as a postdoctoral research associate in the Institut Nationale de la Recherche Agronomique (INRA), Centre de Nancy in January 2004. After four year's research experiences in France, and Korea without completing his dissertation, he decided to complete his dissertation and came to USDA-Forest Service, Santee Experimental Forest, Center for Forested Wetlands Research, South Carolina due to the invitation of Dr. Devendra M. Amatya, Co-Chairman of his Ph.D. Advisory Committee.

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Finally… I arrived in the end of this long tunnel! It has been longer than I expected. At all painful times that I have had to change the completion schedule, I have remembered this word in the Bible.

"In his heart a man plans his course, but the Lord determines his steps."
(Proverbs 16:9, New International Version)

With this word in my mind, I could have waited for this exciting moment!! Sometimes I could not understand why these things happened to me, but now I believe Lord has a purpose for this delay even though I don't know right now. I am sure God has been working with me in ways that I cannot see. Anyway, thanks God! It is faster than Noah's Ark (=120 years)!!

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ABSTRACT

Excessive nitrogen drained from agricultural field has resulted in various water quality problems. Widespread evidence from scientific studies has stated that nitrogen can be removed from water in the agricultural watershed during its downstream transport. Many scientists have also used constructed wetlands to remove nitrogen from water drained from adjacent watersheds more efficiently. However, the efficiency of nitrogen removal by constructed wetlands varies due to many factors including water temperature, nitrate concentration in the water column, dissolved oxygen, organic carbon content in sediment and benthic infauna. This literature review was proposed 1) to understand the fate of nitrogen loaded from agricultural watershed, 2) to understand the nitrogen transformations in constructed wetlands, and 3) to discuss the design considerations of constructed wetlands for efficient nitrogen removal. This review provides the efficient information to understand the fate of nitrogen drained from agricultural watersheds, nitrogen transformation in constructed wetlands. General design principles of wetland construction for wastewater treatment and design approaches to nitrogen removal by constructed wetlands from waters near agricultural watersheds are also discussed.

1.1 Introduction

Agricultural activities in North Carolina have resulted in several environmental problems, particularly excessive nitrogen from agricultural watersheds. Nitrogen compounds play an important role in augmenting plant growth and stimulating the production of wildlife, but they are also the principal concern due to their role in eutrophication, their effect on the oxygen content of receiving waters, and their toxicity to aquatic invertebrate and vertebrate species (Kadlec and Knight, 1996).

Watershed can be simply defined as the area of land upon which water from direct precipitation, snowmelt, and other storage collects in surface channels and flows down to the outlet at which water goes into another water body such as streams, rivers, wetlands, lakes, and oceans (Black, 1990). Therefore, an agricultural watershed can be defined as a watershed including agricultural fields such as croplands and farms. The drainage from an agricultural watershed where inorganic nitrogen fertilizers are applied can contaminate surface and ground water. A high level of nitrate in drinking water supplies can be toxic to infants, (Lee et al., 1992) and cause the eutrophication of surface waters (Wolfe and Patz, 2002).

Wetlands are transitional areas between terrestrial and aquatic zones. They have many functions, and constructed wetlands have been designed and operated to simulate the water quality improvement function of natural wetlands, which frequently are sinks for large amounts of nutrients (Mitch and Gosselink, 1993). This technology for wastewater treatment has been gradually emerging since the 1950's (Watson, 1992) because constructed wetlands have been considered as an economical and environmentally-safe alternative treatment systems that are simple and easy to operate (Bhamidimarri et al., 1991). Wetlands are also beneficial for esthetics and wildlife utilization and production.

Many constructed wetlands have been used in order to treat nitrogen from adjacent watersheds. This overview examines the ways in which nitrogen is loaded from nearby watersheds and in which nitrogen is transformed in constructed wetlands. In addition, design considerations for efficient wastewater treatment with constructed wetlands were also discussed.

1.2. Nitrogen loading from agricultural watersheds

1.2.1. Nitrogen removal mechanisms in agricultural watersheds

Nitrogen is removed through plant uptake, denitrification, nitrogen immobilization in sediments, decomposition and sedimentation of organic matter, and dissimilatory nitrate reduction under extremely reduced conditions as nitrogen is transported through the watershed (Birgand, 2000; Butcher, 1933; Howard-Williams et al., 1982; Janse and Van Puijenbroek, 1998)

1.2.1.1. Macrophytes and algae

Nitrogen uptake by macrophytes in agricultural watersheds is closely related to the ability of plants to remove nutrients from the water column as water flows through. In addition, the importance of nitrogen removal through plant assimilation varies greatly in the literature because of the variations in macrophyte densities, nitrogen affinity, and intrinsic uptake rates.

Howard-Williams et al. (1982) reported that high uptake rates in a stream dominated by watercress may be due to high nitrate affinity and high density of this plant.

Macrophytes living in a stream are commonly rooted in the sediments to endure the stress from water currents (Butcher, 1933), but floating plants such as duckweed (Lemnaceae) are present in stagnant water in eutrophic agricultural ditches (e.g., Janse and Van Puijenbroek, 1998).

Even though this review is mainly focused on nitrogen transformation and transport in stream, the role of phosphorus on eutrophication in streams cannot be ignored. Human activities often result in large fluxes of P to streams. P tends to be retained efficiently in streams, and then leads to higher primary production, which sequentially leads to high rates of decomposition and depletion of dissolved oxygen in bottom waters and surface waters at night in calm weather (Correl, 1998). Phosphorus is commonly the growth-limiting nutrient for phytoplankton production in freshwater lakes; however, nitrogen is usually limiting in coastal marine systems (Blomqvist et al., 2004).

The rooted macrophytes can be classified with three groups according to the degree of shoot submersion: Submersed, emergent, and floating-leaved macrophytes. Submersed macrophytes have their stems and leaves almost submersed in the water, and almost all nutrient removal processes occur under the water and in the sediments. Emergent macrophytes are those whose stems leaves are clearly above water, and they get their nutrients mostly from the sediment. In the case of floating-leaved macrophytes, their leaves are usually floating at the water surface and the rest of the shoot is under the water. Therefore, the shoots and roots may assimilate nutrients (Birgand, 2000).

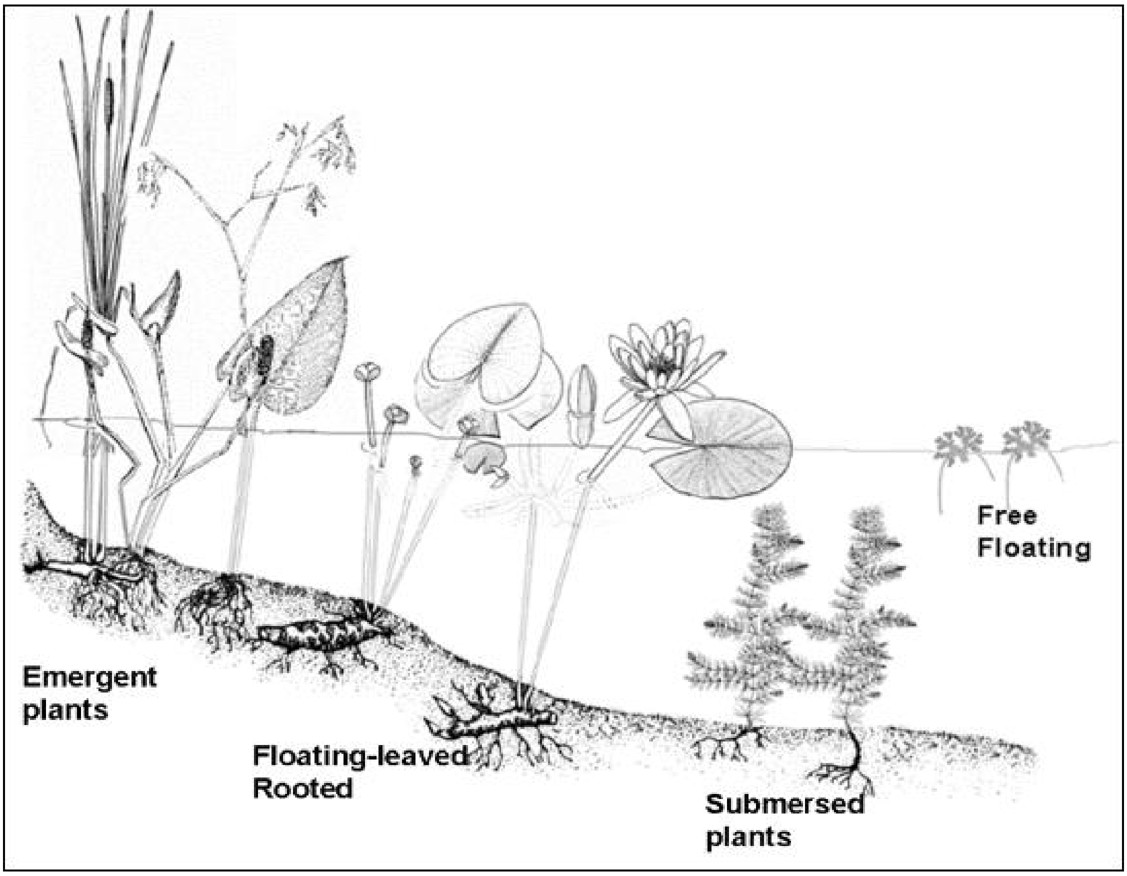

Fig 1.1. Types of plants in the watershed (Main et al., 2007, used under the permission of authors).

Several studies have also reported that algae, mainly periphyton (attached algae) in streams, potentially remove a large amount of nitrogen from agricultural streams due to their extremely high rate of primary production (Bachmann et al., 1988; Delong and Brusven, 1992). Goldman et al (1974) used this potential of algae to remove ammonia and nitrate from wastewaters with some success. They combined algal and seaweed nutrient stripping processes with a marine aquaculture. During the 11 weeks study period in summer of 1972, 95 per cent of the influent inorganic nitrogen was removed by algal assimilation. Overall, the number of studies on algal nitrogen removal is limited compared with that of plant uptake, and the potential of nitrogen removal by algae has probably been insufficiently documented so far. (Birgand et al., 2007).

1.2.1.2. Denitrification

Denitrification is "an energy requiring reduction process where electrons are added to nitrate or nitrite nitrogen, resulting in the production of nitrogen gas, nitrous oxide, or nitric oxide" (Kadlec and Knight, 1996). Nitrate is utilized by facultative anaerobic microbes as electron acceptors in their respiratory chain in the absence of oxygen. The presence of oxygen is inclined to inhibit the activity of nitrate reductase allowing denitrifiers to switch from an aerobic to an anaerobic metabolism. Denitrification usually does not occur, but rarely has been observed in aerobic conditions (Birgand et al., 2007; Kadlec and Knight, 1996).

Sylvia et al. (1998) reported that anaerobic conditions, nitrate availability, and carbon availability are the most important factors affecting denitrification in the environment. However, it seems that nitrate presence and availability are not the limiting factors in agricultural systems because it usually is abundant, although nitrate could be limiting at certain times of the year in some streams receiving agricultural drainage. Therefore, anaerobic conditions and carbon availability may be the major limiting factors.

The major site of denitrification in streams is the sediments. Biofilms in the water column are also potentially important. Nitrate is supplied to the stream sediments through diffusion from the water column, and nitrification of mineralized ammonia within the sediment after the decomposition of accumulated organic matter (Birgand et al., 2007).

1.2.1.3. Storage or burial of nitrogen

Storage or burial of nitrogen is also one of the main mechanisms that can remove nitrogen from the water column (Birgand, 2000). It includes four steps: sedimentation of organic matter onto the sediment, decomposition of the organic matter which leaves refractory organic matter, burial of the refractory organic matter into the sediment, and accretion of the sediment. However, relatively large amounts of nitrogen can be released back to the water column through decomposition of the organic matter (Webster and Benfield, 1986).

Sedimentation of organic mater mostly depends on the amount of organic mater in the stream (Birgand, 2000). Organic matter can be supplied into the stream through autochthonous (produced in the stream) and allochthonous (transported to the stream) origin. Because of the extremely high primary production of agricultural streams, thus a substantial amount of settling organic matter can be produced autochthonously in the stream. However, part of the sedimentation of organic mater may be allochthonously transported from out of the stream such as surrounding forests due to the intrinsic connectivity of streams with upstream communities (Entrekin et al., 2008).

These organic matters in the stream are decomposed through three steps at first: the initial rapid loss through leaching, microbial decomposition, and mechanical fragmentation (Webster and Benfield, 1986). After the mechanical fragmentation of organic matter, nitrogen seems to be sequentially released to the surrounding environments in different forms. Howard-Williams et al. (1983) suggested that ammonia is released into the water first, and then followed by nitrite, then nitrate, and finally by dissolved organic nitrogen (DON) based on their macrophyte decomposition study in laboratory condition. They insisted that sequential release of nitrogen species were mainly caused by nitrification which does not occur in the beginning due to the time taken by nitrifiers to colonize the decomposing organic matter (Howard-Williams et al., 1983).

Svendsen and Kronvang (1993) reported that emergent macrophytes were more efficient than submerged macrophytes in retaining nitrogen through trapping sediments and organic matter. They also showed that the rate of sedimentation was negatively correlated with the flow velocity. Sand-Jensen (1998) and Schulz et al. (2003) also observed similar results.

1.2.1.4. Immobilization in sediment

Other studies have reported that a substantial amount of nitrogen could be removed by immobilization in sediment. For example, Qualls (1984) reported that approximately

25 % of nitrogen flowing through a swamp system draining agricultural lands during high flow conditions was immobilized in leaf litter.

Some studies reported that nitrate enrichment increased decay rate of organic matter because nitrogen can be limiting for the microbial biomass (Carpenter and Adams, 1979). However, others reported that nitrate enrichment didn't increase decay rate because most macrophytes have relatively high levels of nutrients in their tissues, and they can sustain microbial growth during the decomposition phase (Howard-Williams et al., 1988).

All plant material is not decomposed completely, and undecomposed material remaining in soil is called as refractory material. Jewell (1971) found that the refractory portion ranged from 10 to 50 % and averaged 24% of the original plant material. Howard-Williams et al. (1983) reported that 56% of the initial nitrogen was regenerated as nitrate, 21% as DON, and 23% remained as refractory particulate nitrogen in their experiment. They also found that DON and refractory nitrogen may not be directly available to downstream plant communities.

1.2.2. Factors affecting nitrogen retention in agricultural stream

Researchers have studied the factors affecting nitrogen removal rates in agricultural streams. Five factors including temperature, nitrate concentration, dissolved oxygen, sediment organic carbon content, and benthic infauna are considered as the most important (Birgand et al., 2007).

1.2.2.1. Temperature

As mentioned above, nitrogen removal in agricultural watersheds is closely related to complicated biological processes such as decomposition, mineralization, nitrification, and denitrification. Temperature affects all of these biological processes.

Downing (1966) suggested that temperature is one of the important factors which growth of nitrifying bacteria depends on. The influence of temperature on *Nitrosomonas* bacteria is explained by the empirical relationship shown below.

$$C_T = \exp \cdot \phi(T - T_0) \quad (1.1)$$

where T_0 is the reference temperature (15 °C) and ϕ is an empirical constant (0.098/°C).

Dawson and Murphy (1972) reported that temperature influences biochemical reaction kinetics. This effect follows Arrhenius' law and is proportional to the next term.

$$e^{-Ea/RT}$$

Where,

E_a = activation energy of the reaction, $kJ \cdot mol^{-1}$

R = the perfect gas constant,

T = temperature, K

According to this theory, nitrogen uptake by plants is higher at higher temperatures. However, there are only a few reports on the temperature effect on these processes, and most of those articles dealt with nitrification and denitrification (Dawson and Murphy, 1972).

1.2.2.2. Nitrate concentration

Many studies reported that high nitrate concentration in the water column stimulates the denitrification process, the main nitrogen removal mechanism. Van Kessel (1977) showed that the disappearance of nitrate was caused mainly by denitrification in the sediment through their laboratory experiments with undisturbed water-sediment profiles from the canal. They found a linear correlation between instantaneous nitrate disappearance rates and nitrate concentration in the overlying water while incubating undisturbed canal sediments and following nitrate concentrations through time.

However, the nitrate concentration is usually high in agricultural streams due to agricultural activities in surrounding areas, denitrification in agricultural streams may depend on the diffusion from the water column (Pind et al., 1997). Pind et al. (1977) quantified total denitrification (D_{tot}) and divided it into denitrification based on NO_3^- diffusing from the overlying water (D_w) and coupled nitrification-denitrification within the sediment (D_n). They found that seasonal pattern of D_w was very similar to that of D_{tot} and 75-90% of D_{tot} can be explainable with D_w.

1.2.2.3. Dissolved oxygen

Oxygen levels in watersheds can influence nitrification, denitrification, and decomposition of organic matter and thus nutrient regeneration and release, even though they do not affect nitrogen uptake from macrophytes and algae directly. As oxygen level increases, decomposition and nitrification (or coupled nitrification-denitrification) activities increase, but denitrification activity decreases. Many studies have been supporting this basic supposition, and few articles are selected and briefly discussed here.

In spite of few conflicting reports on decomposition speed under aerobic and anaerobic conditions for freshwater macrophytes (Webster and Benfield, 1986), decomposition generally occurs faster under aerobic conditions than anaerobic conditions. Ponnameperuma (1972) suggested that inorganic nitrogen under anaerobic conditions is released in bigger quantities than under aerobic ones, mainly due to less microbial immobilization in anaerobic media. Yamamoto et al. (2007) recently examined whether replantation of benthic microalgae (BMA) can remediate shallow organically enriched sediment. They isolated *Nitzschia* sp., the dominant species in Hiroshima Bay, Japan, and mass cultured, then replanted in the same area. During the study period, they observed an increase in redox potential (ORP) and a decrease in acid-volatile sulfide (AVS) in the experimental area, indicating that the sediment condition changed from anaerobic to aerobic. They found organic nitrogen (ON) in the sediments decreased significantly as oxygen level increased in the sediment. Oxygen produced by the replanted BMA may have enhanced aerobic bacterial activity and accelerated the decomposition of organic matter.

Increasing oxygen concentration in the water column may enhance the processes such as nitrification and coupled nitrification-denitrification (Houweling et al., 2008; Rysgaard et al., 1994). Houweling et al. (2008) showed that aeration intensity is a significant factor in interactions between the water column and sediments through their nitrification modeling study using a lagoon model that includes a modified activated sludge biokinetic model. Using the isotope pairing technique, Rysgaard et al. (1994) showed that coupled nitrification-denitrification increased as oxygen level in the water column increased. However, they concluded that the overall denitrification activity in the sediment decreased as the oxygen concentration in the water column increased because Dw (in the water column) decreased more than coupled nitrification-denitrification (in the sediment) increased.

From the early studies (i.e., Triska and Oremland, 1981) many studies have shown that denitrification is inhibited by enhanced oxygen levels in the sediment. Triska and Oremland (1981) studied the role of periphyton in stream denitrification using the acetylene inhibition technique, and proposed the importance of oxygen concentration near potential denitrifying sites. They concluded that denitrification was inhibited by the incubation of periphyton scrapings in the light because of algal photosynthetic O_2 production. Cey et al. (1999) also reported that patterns of dissolved oxygen concentrations in the subsurface coincided with the pattern defined by groundwater nitrate and these patterns indicated that oxygen level within the riparian zone in a watershed were conducive to denitrification.

1.2.2.4. Sediment organic carbon content

Organic carbon in sediment is an essential factor of nitrogen removal in a watershed (Knowles, 1982), and has a significant relationship with denitrification activity (Hill and Sanmugadas, 1985). Its presence in the sediment can provide the electron donors for denitrifying organisms, even though the magnitude of the specific denitrification activity may depend on the quality of the carbon source, and labile organic carbon may be the most favorable substrate. In addition, oxygen demand resulting from organic carbon catabolism may decrease oxygen penetration in the sediment, thus increasing rates of denitrification by nitrate diffusion from the water column.

The organic carbon content in the sediment is dependent on the stream productivity (Fiebig et al., 1990), but can be favored by the presence of macrophytes (Clarke and Wharton, 2001). Fiebig et al. (1990) compared organic chemistries of soil waters in the riparian zone with an adjacent stream at an upland site in mid-Wales from August,1986 to August, 1987. They concluded that the riparian zone can contribute substantial amounts of DOC to a stream ecosystem. Clark and Wharton (2001) exploring spatial variability of sediment characteristics at 17 lowland rivers in southern England. They characterized total and inorganic phosphorus, total nitrogen, organic carbon, silt-clay fraction, and organic matter content, and investigated their relationships with aquatic macrophytes. They gave some indication of the organic carbon in sediment associated with five macrophyte species.

Swank and Caskey (1982) showed that denitrification rates determined on undisturbed sediment cores were positively correlated with organic carbon content. However, they found better correlation between denitrification rates and the TKN content in the sediment, and similar results were reported by Wyer and Hill (1984). Swank and Caskey (1982) indicated that the organic matter contained a large amount of carbon, and this is usually not available for microbes as an energy substrate because it is resistant to microbial decay.

1.2.2.5. Benthic infauna

Benthic infauna is defined as “any of a diverse group of aquatic animals that live within marine and fresh water sediments” (Storm water glossary, 2009) and may increase nitrogen removal in agricultural streams by enhancing both denitrification and nitrification-denitrification processes. Chatarpaul et al. (1979) found that nitrate concentration in the water decreased much faster in the columns with worms than in the ones without through their incubation experiment on undisturbed sediment with and without worms. In addition, they also found that nitrate concentration in the overlying water increased again after 12 days of incubation. They explained that the first decrease was due to denitrification and the increase after 12 days incubation was due to nitrification.

The next year, the same authors showed that initial enhanced denitrification in the presence of worms was partly due to the mechanical transfer of nitrate as worms draw current into the tunnels formed while they search for food. They also explained that the tunnels could provide supplementary surface area for the nitrate diffusion to actual denitrification sites (Chatarpaul et al., 1980). They showed that the worms themselves could carry out denitrification because of their denitrifying bacteria on their body through their experiments incubating the worms in artificial sediment made of glass beads.

With another hypothesis, they suggested that the burrowing activity of the worms could accelerate the upward movement of sediment ammonium to the aerobic layer enhancing the nitrification process. They proposed that this nitrification enhancement could be partly due to the oxidation of ammonia released by the worms and their fecal pellets.

Hansen and Kristensen (1997) studied potential effects of a possible recolonization of benthic infauna to previously defaunated organic-rich sediment, and found similar results. Burrow construction and ventilatory activities following macrofaunal recolonization in the long-term defaunated organic-rich sediment resulted in a massive pulse release of accumulated porewater NH^{+}_{4} to the overlying water. Their results pointed out that macrofaunal recolonization will have a pronounced long-term influence on benthic metabolism and nutrient exchange, leading to a reduction of the large internal pool of dissolved and particulate nitrogen in the sediment.

1.2.3. Spatial and temporal variability of nitrogen removal

Many studies have shown that nitrogen removal efficiency or rate varies spatially with reach in the watershed, organic carbon in the sediment, stream channel size, and stream morphology (Birgand, 2000).

Cooke and White (1987) found no statistically significant differences in the *in situ* denitrification activity (IDA) in different core samples from the left bank to the right of different cross-sections along an 800m reach of the River Dorn in England. Their IDA study used the acetylene inhibition technique.

Several studies on the effects of sediment type on denitrification have found that denitrification rates are highest in fine organic sediment and lowest in gravel ones (Wyer and Hill, 1984; Hill and Sanmugadas, 1985; Wyer and Kay, 1989; Garcia-Ruiz et al., 1998a). As stream order increases, sediment types are inclined to be finer because of the decrease of flow velocity (Garcia-Ruiz et al., 1998b). These results propose that denitrification process may increase as stream order increases.

However, a counteracting effect has been found between organic carbon and nitrate concentration for the magnitude of denitrification rates as stream order increases. As nutrients are taken away from the water in the downstream direction, nitrate concentrations diminish and may become a limiting factor for denitrification. For this reason, the denitrification rate is generally lower in estuaries and receiving water bodies than in headwater agricultural streams (Seitzinger, 1988), but surely exceptions still exist (Garcia Ruize et al., 1998b).

Several studies have also reported that spatial variation of primary productivity in streams can influence spatial variability of nitrogen removal from water. Vannote et al. (1980) suggested that stream productivity is not expected to be significant in streams with three and lower order, but extremely high primary productivity was found in first order streams. This River Continuum Concept was taken over by Minshall et al. (1983).

Distribution of macrophytes can also affect the spatial variation of nitrogen removal. Canfield and Hoyer (1988) showed that plant growth in Florida streams is primarily light limited. Butcher (1933) proposed that water current is the major factor of macrophyte distribution. A literature review by Carr et al. (1997) confirmed that high currents usually inhibit growth. Agricultural lands are not located in steep areas where water current may be a limiting factor. Water depth can be another factor to influence the distribution of nitrogen removal in higher order streams even though submersed macrophytes can grow in deeper waters (Carr and Chambers, 1998).

The spatial distribution of nitrogen removal efficiency is also associated with the stream channel size and stream morphology. Nitrogen removal efficiency depends on the ratio of the amount of nitrogen stripped from the water to the total amount of water flowing into a stream. As stream size increases, flow rate increases, the mass of nitrogen in the water column also increases, and so does the stream bottom area. Therefore, the efficiency of nitrogen removal is based on the proportion of the stream bed area versus flow rate. Leopold and Maddock (1953) showed that the mean stream depth D (m) and the average stream flow Q (m^3/s) can be correlated using the following equation.

$$D = 0.2612 \times Q^{0.3966} \qquad (1.2)$$

Stream depth therefore increases with increasing stream order and the ratio of stream bed area to flow rates decreases. According to this concept, nitrogen removal efficiency is inclined to decrease with increasing channel size, but several researchers found opposite results due to the counteracting effects such as decrease in water velocity and water residence time increase, which occur concurrently as stream order increases (Garcia-Ruiz et al., 1998b; Howarth et al., 1996).

Stream morphology can also influence the spatial variability of nitrogen removal. Rosgen (1996) suggested that stream morphology depends on longitudinal slope, underlying substrate, bank full flow, and riparian zone management. Even when longitudinal slope and underlying substrate are exactly same, flood flow and riparian zone stability can be influenced by management and anthropogenic factors.

Temporal variation of nitrogen removal efficiency or rate has also been presented by many studies. Even though there were few exceptions (Cooper and Cooke, 1984; Oehler et al., 2007), nitrate concentrations in agricultural watershed are generally highest in winter and lowest in summer. In spite of the highest nitrogen concentration in winter, nitrogen removal efficiency is low in winter mainly due to the low plant metabolism of macrophytes (Birgand et al., 2007).

Howard-Williams et al. (1982) reported that nitrogen removal by watercress uptake in the reach in the watershed was much greater in summer because the stream area covered by plant mat was larger. They insisted that watercress is an ideal plant for nitrate removal from stream due to the fast growth rates and high nitrogen affinity. However they also suggested that nitrate removal is only a summer phenomenon, and much of the stripped nitrogen will be returned to the water after growing season.

Alexander et al. (2000) proposed that nitrogen removal rates in the stream expressed as temporal decay coefficient ($time^{-1}$) decreased as the mean stream water depth increases. They analyzed the data from several related articles and converted nitrogen loss presented as a fraction of external inputs to loss rates expressed in $time^{-1}$ using the water time of travel for each study. Few data used for their analysis suggested a clear trend.

1.3. Nitrogen transformations in constructed wetlands

1.3.1 Nitrogen removal by constructed wetlands

Natural wetlands were used for wastewater treatments a long time ago. However, it was not the first option, but the last resort due to some restrictions. For this reason, constructed wetlands were developed to mimic the water quality function of natural wetlands.

Three basic types of constructed wetlands have been developed and tested for wastewater treatment. These are surface flow, subsurface flow, and vertical flow wetlands. Surface flow wetland systems are most similar to natural wetlands with standing water on the surface usually 0.15 to 0.3 m deep (Watson, 1992). In subsurface flow systems, wastewater flows beneath the surface in the root zone of marsh vegetation. The substrate is usually coarse gravel that is 0.3 to 0.9 m deep. In vertical flow wetland systems, water is applied to the surface in timed intervals and flows down through the plant roots, which are growing in porous media (Watson, 1992).

The first work utilizing constructed wetlands for wastewater treatment was considered to be at Max Planck Institute, Krefeld, Germany, in 1953 (House et al., 1993). Through this study, researchers found that plant species most desirable for wastewater treatment need to have a large rooting zone, grow rapidly, have a high transpiration rate, and have adventitious roots. *Phragmites australis* was effective in treating brewery effluents, nuclear laboratory and municipal sludges. Dramatic improvement in effluent quality was observed in all cases. The sludge was reduced to nearly one thousandth of its initial volume. The bacterial levels of Salmonella species and Escherichia coli were also lowered (Siedel, 1976).

After this research, many studies have focused on using constructed wetland systems for treating wastewater such as municipal and nuclear laboratory sludges, and agricultural runoff contaminated by heavy metals, pesticides, fertilizers, or excessive nutrients. Constructed wetlands have been useful for removal of excessive nutrients as the nutrients increased growth of wetland plants (House et al., 1993).

It was reported that excessive nitrogen from animal wastes or agricultural runoff has been successfully removed by constructed wetlands. Gersberg et al. (1983) reported that total nitrogen removal efficiency was from 25% to 95% in their experiment of surface flow wetlands. They insisted that the removal efficiency of total nitrogen was related to the supplemental carbon addition such as methanol and mulched plant biomass because carbon addition can stimulate denitrification.

Frankenbach and Meyer (1999) estimated the annual denitrification rate of a surface flow treatment wetland using a N mass balance approach, accounting for measured influx and efflux of N, measured plant uptake of N, measured uptake of N by soil accretion into the sediment, and estimated NH_3 volatilization. The removal efficiencies of their constructed wetlands vary from 17% to 37% according to the season. On an areal basis, net N removal was 1.84 g N/m^2/day from mid-May to November, and the mean nitrogen removal rate was 1.07 g N/m^2/day in entire year.

Huang et al. (2000) evaluated the impact of residence time on nitrogen removal in subsurface wetlands constructed for treating domestic wastewater. They insisted that species did not influence nitrogen removal. Instead, the concentrations of each nitrogen species in the wetlands significantly decreased with increased residence time. The efficiency of nitrogen removal in this study varied according to nitrogen species. Removal of NH_4 and TN ranged from 18 to 73% and 31 to 68%, respectively. The NO_3 concentration in the influent and effluent at study site were low and no differences with residence time were observed.

The removal of dissolved nutrients from surface waters is known as a two-step process, delivery and consumption. The main mechanisms of nutrient delivery in wetlands are convective mass transfer within surface waters, sedimentation, and downward infiltration. In the case of consumption, four processes-biomass expansion, adsorption on peat, soil building, and microbial activity-collectively provide the mechanisms at the surface of the soil, litter, plant stems and the algal mat (Kadlec and Hammer, 1988).

Various microbial activities were closely related to nitrogen removal processes. Through those microbial processes such as mineralization, nitrification, and denitrification, nitrogen is transformed sequentially and eventually removed from wastewater. After Mulder et al. (1995) found a new pathway from ammonia to nitrogen gas in anaerobic condition and named it ANAMMOX, several researchers have studied the potential of this new process for improving nitrogen removal in constructed wetlands with some success (i.e., Dong and Sun, 2007). However, more published results are needed to invite ANAMMOX as one of the conventional nitrogen removal processes in wetlands.

Howard-Williams (1985) suggested that nutrient spiraling concept was useful in explaining nutrient transport and transformation in wetland ecosystem. Through literature review, he indicated that a nutrient cycle in wetlands is displaced and becomes a spiral because of the continuous downstream movement of water and particulate materials. He also pointed out spiraling rather than cycling is possibly useful concept to describe nutrient dynamics in systems such as wetlands where a continuous unidirectional transport of materials is a major feature of them.

This concept implies that a single atom may be used over and over as it flows into downstreams, and the degree of utilization depends on the downstream displacement from one cycle to the next, spiraling length (Newbold, 1992). Prior and Johnes (2002) monitored total nitrogen concentrations in the southeastern England over 2 years to investigate the influence of instream nutrient spiraling and wetland transformation processes on surface water quality. They found the small variations of total nitrogen concentrations which were consistent with the nutrient spiraling theory. However, the nutrient spiraling theory is not yet extensively applied to nitrogen removal by constructed wetlands, at least in our knowledge.

1.3.2. Nitrogen cycle in wetlands

The nitrogen cycle in constructed wetlands is intricate and many factors interact. It is described by many studies including Brooks (1989), Kadlec and Knight (1996), Lee et al.(1999), and Widener and Wolfe (1994) (Fig 1.2).

Most of the nitrogen in wetlands is present in the sediment in organic forms. Concentrations of available inorganic nitrogen in the sediment are generally two orders of magnitude less than the sediment organic nitrogen. Nitrogen is also present in the form of plant biomass. The major nitrogen input into wetlands is sediment bound nitrogen (Widener and Wolfe, 1994). Nitrogen in wetlands is subject to transformation through the sequential processes such as ammonification, nitrification and denitrification, and almost all of these processes are controlled by microorganisms. All microbial activity is influenced by temperature because temperature affects enzyme activity (Kadlec and Knight, 1996).

Ammonification is the process which changes organic nitrogen in the wetland plants into inorganic nitrogen in the water or adjacent soil (Brooks, 1989; Kadlec and Knight, 1996; Schalls, 1989). It proceeds more slowly in anaerobic than aerobic conditions because of the reduced efficiency of heterotrophic decomposition in anaerobic environments, but ammonium nitrogen is more likely to accumulate in anaerobic systems because of decreased nitrification rates. Kinetically ammonification is faster than nitrification. The rate of ammonification in wetland soils depends on temperature and pH. The optimum temperature ranges from 40 to 60 ^oC and the optimum pH is from 6.5 to 8.5 (Kadlec and Knight, 1996).

In aerobic zones of wetlands, nitrification of ammonium occurs in a two-step microbial process that can be summarized by the two equations below (Kadlec and Knight, 1996).

$NH_4^+ + 1.5O_2 \rightarrow 2H^+ + H_2O + NO_2^-$ (mediated by *Nitrosomonas* spp.)

$NO_2^- + 0.5O_2 \rightarrow NO_3^-$ (mediated by *Nitrobacter* spp.)

One of the most unique characteristics of wetlands is the oxidized zone at the water and sediment interface, which occurs mainly due to the shallow water depth. Oxygen concentration in the oxidized zone is enhanced by the rhizosphere, a region around the root hairs of wetland plants. The plant effectively transports oxygen to the root hairs, some of which goes into the soil matrix forming the oxidized zone where a significant amount of nitrification occurs (Widener and Wolfe, 1994).

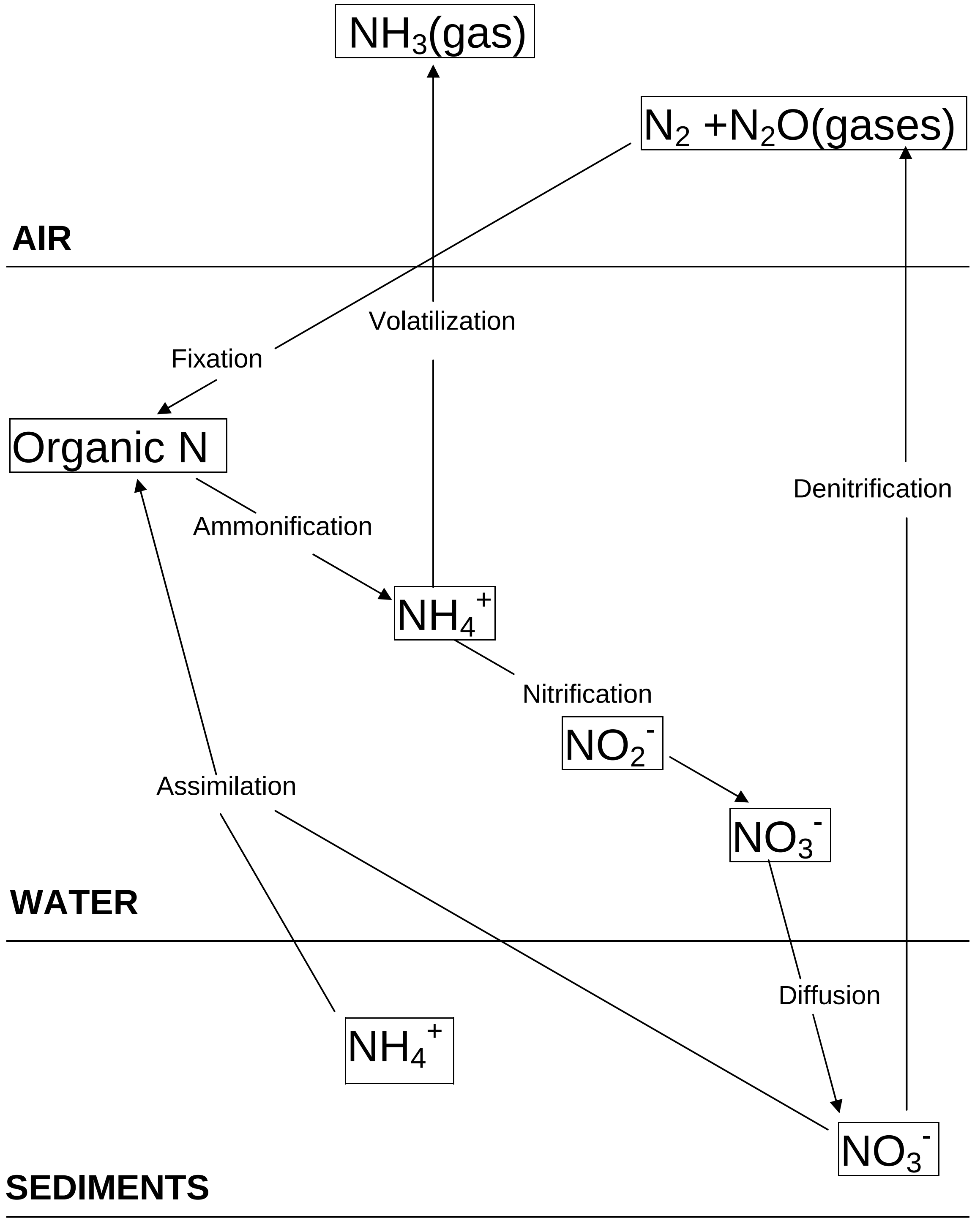

Fig 1.2. Nitrogen cycle in treatment wetlands (based on Kadlec and Knight, 1996. Reproduced under the permission of authors).

Denitrification occurs in anaerobic microzones which form primarily in the sediment layer beneath the oxidized zone. Microzones can also form on substrate surfaces and suspended particles where dissolved oxygen (DO has been depleted through biological oxygen demand (BOD) exertion. Ammonium, produced during organic decay in anaerobic zones, is stable, but is driven by diffusion to aerobic zones, where nitrification occurs. Nitrate diffuses into the anaerobic zones where denitrification takes place.

The diffusivity of the different N species and the existence of zones depending on redox potential cause spatial variation of concentrations and drives the dynamics of a wetland system (Widener and Wolfe, 1994). As mentioned above, denitrification usually occurs in anaerobic condition. However, Widener and Wolfe (1994) insisted that the denitrifying microbes are facultatively anaerobic and they can also subsist in oxygen rich waters.

In some natural wetlands, atmospheric nitrogen may be fixed by wetland plants and algae, but that process is usually assumed to be negligible in treatment wetlands. Anaerobic nitrogen oxidation process, ANAMMOX (Mulder et al., 1995), is possibly one of the processes in nitrogen cycle, but this process is not yet extensively investigated. These processes related to nitrogen cycle in wetlands were summarized in Fig. 1.2.

1.3.3. Mechanisms and factors affecting nitrogen transformations

To remove nitrogen from agricultural wastes through wetland system, denitrification and plant uptake have been regarded as two primary mechanisms (Frankenbach and Meyer, 1999).

Denitrification is the bacterially mediated reduction of nitrate and nitrite to nitrogen gases which occurs in anaerobic soils as mentioned earlier (Starr and Gillham, 1993). This process has been regarded as one of the most important nitrogen removal process from agricultural wastewater because nitrogen can be transformed to N_2 gas and removed permanently from wastewater (Frankenbach and Meyer, 1999; Gersberg et al., 1983; House et al., 1994; Huang et al., 2000; Hunt et al., 1994). Denitrification in soil is influenced by several factors such as available organic carbon, NO_3-N supply, anaerobic condition, vegetation, temperature, pH, redox potential, moisture content, and the presence and activity of denitrifiers (Reddy et al., 1978; Starr and Gillham, 1993). In a wetland system, the amount of organic carbon, nitrate supply, oxygen level and temperature have been regarded as the most important factors (Starr and Gillham, 1993; Frankenbach and Meyer, 1999). In addition to plant uptake and denitrification, nitrogen loss can occur by ammonia volatilization (Howard-Williams, 1985). In the case of ammonia volatilization, pH and temperature are the most important factors. This process is important in specific cases when NH_4-N concentration is greater than 20mg/L and pH is greater than 8 (Kadlec and Knight, 1996).

Wetland vegetation plays an important role in nitrogen removal from wastewater. Using several microcosm wetlands unplanted and planted with five macrophytes including *Phragmites australis*, *Commelina communis*, *Penniserum purpureum*, *Ipomoea aquatica*, and *Pistia stratiotes*, Lin et al (2002) found that Planted wetlands exhibited significantly greater nitrate removal than unplanted wetlands ($P<0.01$). In addition, a wetland planted with *Penniserum* showed consistently higher nitrate removal than those planted with the other four macrophytes, suggesting that all of the macrophyte species have specific nitrate removal efficiency possibly due to their ability to produce carbon for denitrification. Plants not only take up the nitrogen directly, but also provide an oxidized microenvironment for microorganisms in the rhizosphere that stimulates both the decomposition of organic matter and the growth of nitrifying bacteria.

The five main factors affecting nitrogen removal in the watershed - temperature, nitrate concentration in the water column, dissolved oxygen, organic carbon content in sediment, and benthic infauna - may be applicable to wetlands since natural wetlands are intrinsically part of watersheds. However, due to the unique characteristic of wetlands as the transitional area between terrestrial and aquatic ecosystems, application of these five factors to wetlands may be different from that of watersheds.

As temperature affects all of the biological processes in agricultural watersheds, it is also closely related to complicated biological processes in wetlands. The term of activation energy of the reaction can be applicable to wetlands, and temperature influences biochemical reaction kinetics in wetlands. Using Mass balance calculations, Bachand and Horne (2000) demonstrated that bacterial denitrification was the main mechanism for nitrate removal, and water temperature significantly affected denitrification rates. Poe et al. (2003) also showed that water temperature is one of the important factors in denitrification rates in constructed wetland.

Poe et al. (2003) showed that Increase of nitrate concentration in water column due to episodic storm events can substantially increase denitrifying activity in the constructed wetland with nitrate addition and rainfall sampling. However, Bachand and Horne (2000) found confronting results. They reported that nitrate concentration in the water column may not be important because many constructed wetlands are increasingly being used for treating nitrogen-rich wastewaters.

It is not difficult to imagine that denitrification rate decreases as dissolved oxygen (DO) level in sediment increases since denitrification, one of the most important nitrogen removal mechanisms by constructed wetlands, usually occurs in anaerobic condition. Gebremariam and Beutel (2008) evaluated nitrate removal rates and dissolved oxygen levels in small batch-mode wetland mesocosms with two different macrophytes, cattail (*Typha* spp.) and bulrush (*Scirpus* spp.) They showed that carbon supply from highly degradable plant litter can stimulate heterotrophic microbial activity, which in turn depress DO levels, and eventually enhance denitrification.

Oostrom (1995) studied the effect of influent organic matter on nitrogen removal in four experimental surface-flow wetlands treating a nitrified meat processing effluent. He reported that organic carbon from influent organic matter was each responsible for about 50% of nitrogen removal.

At least in my knowledge, no studies have been focused on the relationship between benthic infauna and nitrogen removal efficiency of constructed wetlands. However, several studies have indicated the close relationship between benthic infauna and other crucial factors of nitrogen removal by constructed wetlands. Using nitrogen isotope paring method with undisturbed cores from intertidal wetland sediments, Svensson et al. (2000) showed that oxygen and nitrate consumption, release of ammonium and denitrification in the sediment was correlated to the biomass of benthic infauna. About 30% of the denitrification was explained by the presence of benthic infauna. In wetlands, the actual nitrogen removal is also dependent on the hydrologic, geochemical, and redox condition and soil structure and vegetation (Dørge, 1994).

1.4. Design considerations for efficient nitrogen removal with constructed wetlands from water loaded from an agricultural watershed

1.4.1. General design principles

Many case studies on constructed wetlands with various purposes have been reported worldwide recently. For example, Babatunde et al. (2008) reported that 1107 cases on constructed wetlands including 551 from the US, 96 from UK, 70 from Australia, and 56 from Germany were published into 67 international scientific journals from 1992 to 2002. We need to have some fundamental design principles for constructed wetlands to minimize costs. As Hammer (1992) pointed out, wetland design criteria should be updated continuously based on case studies. We went back to classical reviews including Mitsch's (1992), Hammer's (1992), and Mitsch and Wilson's (1996) for this purpose. Latest review articles including Babatunde et al. (2008), Carty et al (2008), and Lee et al. (2009) based on recent case studies were also used in the discussion of this section on design considerations for constructed wetlands.

The general principles for wetland design from those articles mentioned above can be summarized as follows. 1) Understand wetland function and design for it, not for form; 2) Give system enough time to be functional; 3) Design for minimum maintenance and self-design; 4) Design the system with the landscape, not against it; 5) Design with multiple objectives; 6) Design as an ecotone, in other words, a buffer system between upland and aquatic systems. Discussions in details for these topics can be found in other articles (Mitsch, 1992; Mitsch and Wilson, 1996).

Based on their general suggestions applicable to the relationship between constructed wetlands and watersheds, our discussions are more focused on landscape position, positions within the watershed, size, and wetland features.

1.4.1.1. Landscape position

Location of constructed wetlands in the landscape is an important factor in determining their role of water quality improvement, and it affects the source and amount of water (Osmond et al, 1995). For example, wetlands that are near a topographical high in a landscape, such as a mountain bog, are supposed to receive more intermittent and less reliable water supplies than those of a marsh in a low area such as the lower costal plain in North Carolina.

Osmond et al. (1995) categorized wetlands as precipitation-dominated, groundwater-dominated, or surfaceflow-dominated based on the source of water, and their description can be summarized as follows. Precipitation-dominated wetlands are on local topographic high in a watershed or in flat or slightly elevated areas in the landscape, where they receive little or no surface runoff. Groundwater-dominated wetlands form in landscape positions at which the water table actively discharges, particularly at the base of hills and in valleys. Such wetlands also receive overland flow but they have a steady supply of water from and to groundwater. Surfaceflow-dominated wetlands are usually located in low points on the landscape or within other water resources. Such riverine, fringe (marsh), and tidal wetlands actively play a role in the landscape since they come in contact with, store, or release large quantities of water and act upon sediments and nutrients. These wetlands are recharged by ground water as well, but surface water provides the major source of water (Osmond et al, 1995).

Mitsch and Gosselink (2000) mentioned the hydrogeomorphic position in the landscape. According to their definition, hydrogeomorphic position is ‘the degree to which a wetland is open to hydrologic and biological fluxes with other systems, including urban and agricultural landscapes’, and they suggested that wetland values depended on this position. In this sense of hydrogeomorphic positions, they classified wetlands as in-stream systems, riparian systems, isolated basins, and coastal fringe systems.

In-stream wetlands process large amounts of water, and they are advantageous to have a potential to treat a significant portion of the water that passes the wetlands in the stream. Riparian wetlands are mainly fed by flooding water and sediments, and slowly release the water back to the stream after the flood passes. Isolated basins may provide flood control and habitat for waterfowl and amphibians even though their exchange of water with neighboring basins is limited. Coastal or fringe systems are important to productivity in the off-shore waters, and they are generally found along coastlines (Mitsch and Gosselink, 2000).

1.4.1.2. Positions within the watershed

Wetland designers have debated on locating whether several small wetlands in upstream or fewer big wetlands in downstream (Ogawa and Male, 1986; Mitsch, 1992; Van der Valk and Jolly, 1992). The advantage of positioning several small wetlands in upstream is that less runoff and erosion might occur in the whole watershed as a result of storing water and sediments in the upstream. Individual small wetlands are also easier to establish. On the contrary, the advantage of few big watershed in downstream (i.e., only one at the outlet of the watershed) may be the simple maintenance and monitoring. Van der Valk and Jolly (1992) suggested that more sediments might be retained by several small wetlands in upstream. On the contrary, Ogawa and Male (1986) reported that the usefulness of wetlands in decreasing flooding and thus water pollution increased as the distance the wetlands is downstream (Mitsch and Gosselink, 1993), and few big wetlands in downstream were more useful.

However, more case studies on this topic still need to be done, and the optimal placing of wetlands in agricultural watersheds should be thoroughly examined based on topography, land use pattern, surface layouts, and subsurface drainage networks (Van der Valk and Jolly, 1992).

1.4.1.3. Size

Compared with conventional treatment plants, requirement for large and flat land area is one of the major disadvantages of constructed wetlands (Hammer, 1992), and many review articles and case studies have discussed on this factor (Hammer, 1992; Mitsch and Gosselink, 1993; Larson et al., 2000; Gerke et al., 2001; Mitsch et al., 2001; White and Fennessy, 2005). Increasing the size of constructed wetlands is conceptually one of the simplest ways to increase nutrient removal efficiency (Gerke et al., 2001). However, constructed wetlands larger than necessary may result in high evaporation rates in the arid region like Arizona, and they need to be constructed as small as possible to meet the treatment objectives while minimizing evaporation aside from the additional costs in overdesigning wetlands.

Wetlands should be restored and constructed with a suitable size ensuring their integration at the landscape scale. Mitsch and Gosselink (1993) proposed that the size of treatment wetlands for nitrogen removal should be at least 5% of the watershed area. Several studies also have reported proper wetland size for water quality improvement at the catchment scale. Hammer (1992) recommended 2% of the catchment area for wetlands establishment to virtually treat all runoff. Larson et al. (2000) showed a design for intercepting agricultural drainage with a watershed to wetland ratio of 22:1, meaning that approximately 4.7% of the catchment was covered by wetlands. Mitsch et al. (2001, 2005) suggested that a sufficient improvement of the water quality would be achieved if the treatment wetland area was between 3 and 5% of the Mississippi catchment area. Additionally, modeling incorporates different criteria for restoration suitability which provides a variety of results for wetland restoration potential at the watershed scale depending on the selected and weighted variables (White and Fennessy, 2005). More recently, Moreno et al. (2007) suggested that proper size of constructed wetlands ranged from 3.25 to 5.60% of total watershed areas to remove most nitrogen from wastewater.

1.4.1.4. Wetland features

Hunt and Doll (2000) suggested common features or internal topography of constructed wetlands for wastewater treatment, and those features are forebays, deep pools, shallow water areas, non-floodable areas, a retention device, and two outlet devices. These features were also mentioned by others (Hunt et al., 2007; Villanova Stormwater Wetlands, 2009; Iowa Stormwater management manual, 2009). These features were suggested to be designed within the wetland to establish a long, winding path for water to chase as it flows through the wetland (Fig 1.2). This flow path was suggested to prevent shortcircuiting, to increase stormwater retention time, and to improve the treatment efficiency (Hunt and Doll, 2000).

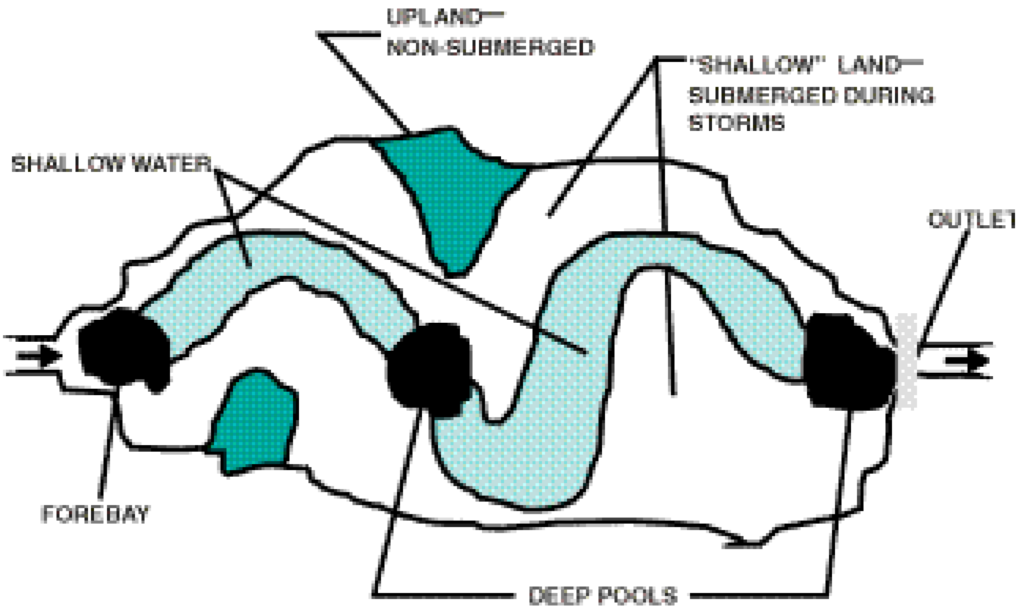

Fig. 1.3. Plan view of stormwater wetland (From Hunt and Doll, 2000. Used under the permission of authors).

1.4.1.4.1. Forebays

Forebays or sediment forebays are typically deep areas of the wetlands and placed where runoff enters the wetland to remove most of the sediment from the storm water before it is disposed into the wetland. These features can be considered as facilities of equivalent upstream pre-treatment (Hunt and Doll, 2000; Iowa Stormwater Management Manual, 2009). A fixed vertical sediment depth marker is installed in these facilities to measure sediment deposition over time. In order to make sediment removal easier, the bottom of the forebay may be hardened using concrete (Iowa Stormwater Management Manual, 2009). If sediment or litter is a concern, it is important to design the wetland so heavy equipment will have easy access to the forebay. Proper maintenance of the forebay will help keep the rest of the wetland from filling with debris. Designers from Mid-Atlantic States have found that the forebay's surface area needed to be 10 percent of the total wetland surface area in order to maintain adequate treatment efficiency (Hunt and Doll, 2000). The forebay is sized to contain 0.25 cm of rain over the watershed (Villanova University Stormwater Wetlands, 2009), and should be 120 ~ 180 cm deep (Iowa Stormwater Management Manual, 2009).

1.4.1.4.2. Deep Pools

Deep pools or deepwater zones are an important part of the wetland if fish habitat and mosquito control is a design requirement (Hunt and Doll, 2000; Iowa Stormwater Management Manual, 2009). Deep pools serve several functions in a stormwater wetland (Hunt et al., 2007). They may dissipate flow energy and trap the sediment flowing in with stormwater. They provide an anaerobic environment for enhanced nitrate treatment, additional water storage that increases both infiltration and evaporation, and then reducing outflow volumes in locations where water tables are low (Hunt et al., 2007). The 75 cm-minimum pools are designed to retain water even during drought. They support little emergent wetland vegetations since they are too deep for most wetland vegetation to grow in except submerged or floating vegetation (i.e., water lilies) which is capable of thriving in water this deep (Iowa Stormwater Management manual, 2009). Fish (i.e., gambousia) that eat mosquito larvae need deep waters to survive (Hunt and Doll, 2000). The surface area of deep pools can range from 1.5 m-diameter circles to much larger areas. Deep pools should occupy 5 to 10 percent of the wetland's surface area (Hunt and Doll, 2000).

1.4.1.4.3. Shallow Water

Shallow water or low marsh areas are part of the "bread and butter" of the stormwater wetland (Hunt and Doll, 2000; Iowa Stormwater Management Manual, 2009). Shallow water zone is where wetland vegetation thrives. This zone is important in nutrient removal since they are better oxygenated than deep pools and support nutrient transformations including nitration (Hunt et al., 2007). At low flow, water should follow the course of the shallow water area. Water is designed to be between 15 and 30 cm deep in the shallow water zone before a storm, with greater depths during rainfall (Hunt and Doll, 2000). Based on recent case studies, Hunt et al. (2007) pointed out most wetland plants could not tolerate this depth for extended periods. They recommended that the average depth of shallow water zone needed to be shallower (i.e., 5 to 10 cm) than that of Hunt and Doll (2000) for diversity of species in constructed wetlands. The shallow water zone occupies roughly 40 percent of the wetland's total surface area (Hunt and Doll, 2000).

1.4.1.4.4. Shallow Land

The land feature that dominates the wetland, “shallow land,” is typically dry except during storms when it is submerged, and acts as an internal floodplain (Hunt and Doll, 2000; Hunt et al., 2007). Two primary objectives of this feature are to stabilize the slop characteristic and to optimize pollutant removal (North Carolina Division of Water Quality, 2007). This land should be between 0 cm and 30 cm above the water at normal pool (Hunt et al., 2007). By having a variety of terrain, a wider variety of vegetation can be grown. Certain plants only like being wet some of the time. A wider variety of plants leads to a wider variety of animals, which leads to more mosquito predators. This “shallow land” typically accounts for 30 to 40 percent of the total surface area of the wetland. If bacterial die-off is an important part of the design, this feature could be larger (Hunt and Doll, 2000). Hunt et al. (2007) referred this zone as “temporary inundation zone” recently. This zone was also called high marsh zone by other wetland designers (North Carolina Division of Water Quality, 2007; Iowa Stormwater Management Manual, 2009).

1.4.1.4.5. Upland/non-floodable areas

Some regions of the wetland can be created not to be submerged normally (Hunt and Doll, 2000), and to tie the wetland into its surroundings (Hunt et al., 2007). These upland/non-floodable areas or upper banks can serve as observation points if the wetland will be used for educational or recreational purposes. Other varieties of vegetation more common to upland regions can grow on these upslope areas. The land can range from 60 to 120 cm or even higher above normal pool, with the dictating factor being the height of water during storms. Non-floodable areas technically are not included as part of wetland zones. They can comprise as much or as little of the total treatment system as desired (Hunt and Doll, 2000).

1.4.1.4.6. Outlets

Outlet need to be as simple as possible and avoid the concrete or other over-engineered structures (Mitsch, 1992; Carty et al., 2008). The wetland outlet serves three purposes. First, it detains water during smaller rain storms, allowing runoff to be slowly released by drawdown devices. Second, it successfully passes excess water through the wetland during big storm events. Third, it lowers the pool elevation inside the wetland for maintenance (Hunt and Doll, 2000; Hunt et al., 2007). Therefore, outlet design needs to be focused on maintaining the existing flood control functionality and still sustaining the wetland by restricting low flows (Villanova University stormwater wetlands, 2009).

The retention device (Fig 1.3) must be constructed so that water does not rise above the top of it during a design storm—such as the 25-year, 24-hour storm. Typically a deep pool is constructed immediately upstream from the weir. Water levels within the wetland are regulated by drawdown structures such as weirs, riser-barrels, and small orifices. For maintenance purposes it is best to install a pipe that can be used to drain the wetland of all its water, except for that in deep pools.

Fig. 1.4. Two prominent features of a typical retention device (Hunt and Doll, 2000). First is the principal spillway. In this photograph the principal spillway is the wooden weir. The second is a drawdown device. In this case the small riser pipe has holes drilled into it behind the protective rocks. The difference in elevation between the drawdown device and the principal spillway dictates the wetland’s temporary storage volume.

Another drawdown device is needed to slowly release water from the storage area of the wetland, which is the volume of water held between the elevation of the weir crest and the desired normal pool. A flashboard riser, originally used for controlled drainage systems in eastern North Carolina, has been adopted into several stormwater wetland designs as an effective outlet (Hunt et al., 2007). A flashboard riser is usually composed of wooden boards with orifice, an adjustable weir, and possibly trash rack or downturn pipe in a corrugated aluminum pipe cut in half with sleeves at either end into which the wooden boards are placed. More detailed description on this apparatus can be found in Hunt et al. (2007). A flashboard riser can be efficiently used when the wetland needs to be drained for maintenance purposes. This is also particularly useful for vegetation maintenance in the initial stage of wetland construction since less fluctuation in the water level during the initial growing season allows for a higher survival rate of planted vegetation (Hunt et al., 2007).

1.4.1.4.7. Additional features

Additional features including emergency spillway, maintenance access, safety features, and runoff bypass were also suggested (Hunt, 2007; Iowa Stormwater Management Manual, 2009). Auxiliary spillway may be required to safely pass excessive water and to prevent from causing structural damage. Maintenance access must be provided to wetland features from a public or private road, and may be designed to allow vehicles to turn around. Safety features such as fences may not generally desirable for all constructed wetlands, but it is obvious that all wetland features must be constructed under the regional safety guidelines. If the deep pool areas need to be constructed, the contours need to be managed in order to reduce the potential for accidental drowning.

When a stormwater wetland cannot have enough size for proper treatment, the wetland may need to let runoff bypass it rather than flow directly through it to prevent from blowing out vegetation. If the available space of wetland is less than 2/3 of what is needed, a bypass is generally recommended (Hunt et al., 2007). More discussions on additional wetland features can be found at Iowa Stormwater Management Manual (2009) and Hunt et al. (2007).

1.4.2. Design approaches to remove nitrogen

General design principles were discussed in previous section, and this section is more focused on the constructed wetland design approaches for nitrogen removal. In order to improve nitrogen removal efficiency of constructed treatment wetlands, basically wetland designer has to choose between two methods; 1) extensive pretreatment such as solid removal, nitrification, and denitrification, or 2) fairly large wetland treatment areas (Kadlec and Knight, 1996). Conservative design with extensive treatment or/and fairly large wetland area is based on specific permit limits since the major nitrogen transformation mechanisms vary seasonally. Different assumptions have been used for wetland design to meet annual, monthly or daily limits. Wetland design approaches for nitrogen removal can be based on quick estimates, regression equation, or mass balance equations with first order transformations (Kadlec and Knight, 1996).

1.4.2.1. Quick estimates

Detailed design calculations for nitrogen reduction are complex, and quick estimates are often useful and convenient for basic screening of concepts and for quick comparison of alternatives. However, they are based on different data ranges and system variables in the databases and do not account for a multitude of potential site-specific parameters that might affect nitrogen removal. Other empirical and preferably rational design approaches should be used for detailed conceptual or final design for nitrogen loss in wetlands. Limiting conditions provide for quick, but overly conservative estimates of size requirements. When land area is not a major problem, total nitrogen loading rates and hydraulic loading rates can be conservative, and total nitrogen reduction is essentially certain to occur (Kadlec and Knight, 1996).

An example of this 'rule of thumb' design approach was the Lakeland, FL constructed wetland treatment system. This is a conservatively sized treatment wetland, operated at very low loading rates, and correspondingly large detention times (Jackson, 1989). However, in most cases, wetlands are designed to the smallest wetland area consistent with the desired degree of performance. Input/ output correlation approach may possibly an alternative tool (Kadlec and Knight, 1996).

1.4.2.2. Input/output

Regression equations derived from input-output data from treatment wetlands provide a better basis for description of water quality treatment performance and determination of proper wetland size. The advantage of these equations lies in their simplicity. However, the residual variability is large, in part due to the limitations on the available variables. These regressions typically do not maximum in the ammonium profile. They cannot be adjusted for environmental variables such as temperature, evapotranspiration, or precipitation. Regressions are also constrained to the limits of the independent variables in the generating data sets. Most regressions cannot be extrapolated beyond those limits without engendering serious errors (Kadlec and Knight, 1996).

1.4.2.3. Mass balance

Mass balance approach based on sequential interrelations among nitrogen species is the best way to estimate nitrogen transformation in constructed wetlands accurately. First order, area based nitrogen loss models provide a suitable method for design of wetland treatment systems in most circumstances. These have the advantage of correctly describing internal phenomena in flow through wetlands, as well as describing batch wetland operation. Studies on side by side wetlands confirm the effects of the principal variables of inlet concentrations and hydraulic loading rates. The parent mass balance equation for water movement may be adjusted to fit extreme environmental conditions of precipitation or evapotranspiration. The rate equations account for return fluxes from the wetland biomass and thus can fit the entire range of hydraulic loadings.

In parameter estimation, the sequential nature of the nitrogen transformations cannot be ignored. Apparent rate constants, which do ignore species generation, can be seriously misleading, even to the point of having the wrong sign. Their use in design is usually inappropriate, unless applied to exactly the same feed speciation from which they were derived. The quality of the first order rate constants for wetland treatment systems will improve with time as additional operation data are collected and analyzed. These rate constants incorporate effects of pH, dissolved oxygen, and other physical, chemical, and biological processes that affect rates of ammonification, nitrification, and denitrification in wetland treatment systems (Kadlec and Knight, 1996).

1.5. Conclusion

Agriculture has been considered as the main source of non-point pollution in the United States. High nitrogen loading drained from upland agricultural systems detrimentally affects downstream water quality. There is a need to understand nitrogen load, and transport from agricultural systems. In order to improve the quality of water drained from agricultural watersheds, constructed wetlands have been considered effective treatment systems.

This literature review examines nitrogen loading from agricultural area and nitrogen transformation in constructed wetlands. Four major mechanisms of nitrogen removal in watersheds including denitrification, vegetation uptake, storage/burial of nitrogen, and immobilization in the sediment were discussed.

Denitrification is the most important process of nitrogen loss in agricultural watersheds. Denitrification processes usually occur in anaerobic conditions such as wetlands, but this process is also important in well oxygenated environments due to the anaerobic nature of the sediments in agricultural streams.

Another major mechanism of nitrogen loss from agricultural watersheds is nitrogen uptake by hydrophytes or other primary producers. Most nitrogen is returned into the water through decomposition. More than a third of the immobilized nitrogen in plant tissue may not be used for microbial assimilation because it is left as refractory dissolved and particulate organic nitrogen.

Storage and burial of nitrogen and immobilization in the sediment were also briefly discussed. Nitrogen removal from agricultural watersheds depends on factors such as temperature, nitrate concentration in the water column, dissolved oxygen, organic carbon in the sediment, and benthic infauna.

Wetland is a part of watershed, and these four mechanisms and five factors discussed on nitrogen removal in agricultural watersheds are also efficient to understand nitrogen transformation and removal by constructed wetlands. However, based on the fact that wetlands are the unique ecosystems located between terrestrial and aquatic ecosystems, different approach to understand nitrogen removal by constructed wetlands is more useful. For this reason, discussion mainly focused on denitrification and plant uptake in constructed wetlands.

General design considerations of constructed wetlands were discussed. Based on these general design considerations, three design approaches-rule of thumb estimates, input/output, and mass balance- for nitrogen removal by constructed wetlands were also discussed. Among these three approaches, Mass balance method based on sequential interrelations among nitrogen species was the most efficient way to predict nitrogen transformation and removal by constructed wetlands due to the complexity and sequential interrelations of nutrient cycle in wetlands. This literature review may be useful to understand nitrogen cycle in watershed and wetland. More quantification of nitrogen transport and transformation in both systems is still required.

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2. Evaluation of Hydrology of an Agricultural Watershed Draining into Treatment Wetlands

ABSTRACT

Excessive amounts of nitrogen from agricultural fields may deteriorate surface and subsurface water quality. This study was conducted to evaluate the hydrology of a 1037 ha agricultural watershed draining into the treatment wetland at Open Grounds Farm in Carteret County, North Carolina. The DRAINWAT, DRAINMOD for Watershed model, was selected for hydrological modeling to obtain flow rates that can be applied to a lumped parameter model for estimating nitrogen loading from the watershed. This model and similar DRAINMOD-based models have been widely used and tested to adequately describe the hydrology of various agricultural and forested watersheds and watersheds with mixed land use in poorly drained coastal plain soils. Its outputs such as flow rates from individual fields and velocities in the drainage canal network have been successfully applied for estimating nitrogen loading by linking to the first order kinetics based lumped parameter nitrogen decay model. However, hydrological simulation results for drainage outflows (both daily, and annual) could not satisfactorily be calibrated in this study mainly due to limited measured data. Therefore, this study focused more on sensitivity analysis of four input parameters including Manning's roughness, surface depressional storage, channel bed slopes, and PET calculation methods to the drainage outflow. The PET by various calculation methods was found to be the most sensitive parameter in this study. The other three parameters were not as significantly sensitive to the cumulative outflow as expected. Even though the hydrological simulation results were limited due to unsatisfactory calibration, DRAINWAT model may be a useful tool if it can be calibrated properly with reliable measured data as was shown in other studies.

2.1. Introduction

Elevated levels of nitrogen are undesirable for surface or groundwater quality for several reasons. High nitrate concentration in drinking water is a potential public health hazard if consumed by infants (Sedlak, 1991). Many studies also indicate that nitrate in drinking water could contribute to the formation of nitrosamines in the body, many of which are carcinogenic. Nitrogen may stimulate excessive growth of algae in surface waters and consequently contribute to eutrophication (Sedlak, 1991).

Eastern North Carolina in the United States has suffered from a decline in water quality over the past 20 years mainly due to excessive nitrogen from both point and non-point source pollution (Poe et al., 2003). Non-point source pollution is considered as the main source of nutrient loading to surface water in rural areas, and is usually well distributed in a large portion of watershed. A large number of studies have been conducted in recent years to evaluate and reduce N loading into the watershed outlets (Amatya et al., 2003, 2004; Fernandez et al., 2005, 2007). Stow et al. (2001) compared the patterns of long-term watershed nutrient inputs with in- stream nutrient loads for the Neuse River, North Carolina. They concluded that nitrogen inputs increased tremendously in the Neuse River watershed during 1990-2000 due to intense animal production. The patterns of in-stream nitrogen loads vary by location and the nitrogen fraction considered, especially downstream where there were no clear nitrogen trends. They found less than 10 % of the nitrogen input to the watershed is exported to the estuary, which means that significant nitrogen removal was occurring in the stream and watershed (Stow et al., 2001)

Birgand (2000) investigated the biogeochemical processes affecting nitrogen loading in a 1125 m long agricultural canal reach of the lower coastal plain near Plymouth, North Carolina. He reported that 3% of total nitrogen entering the upstream end was retained within the reach in 14 months mainly due to the combination of nitrate retention and release of organic nitrogen. He also found that most of the nitrate removal was caused by denitrification after diffusion from the water column to the sediment through the measurements of algae and macrophyte biomass within the reach and the measurements of nitrogen and carbon concentrations profiles at the sediment water interface. Release of organic nitrogen resulted from flux of refractory organic nitrogen from the sediment into the water column. He indicated that approximately 20% of the retention of inorganic nitrogen was explained by the algal and macrophytic assimilation. Appelboom (2004) studied similar in-stream processes of a forested canal based on detailed experiments. He reported that decay coefficients along a forested canal ranged from 0.07/day to 0.16/day. Birgand (2000) found decay coefficients in agricultural canal were from 0.2/day to 1.6/day (Fernandez et al., 2007).

Many modeling studies have also been carried out to describe nitrogen loading from agricultural and forested watersheds. The cumulative effects of land use and water management practices on downstream water quality of a watershed can be evaluated through the consideration of many factors such as climate, topography, geology, soils, land cover, and soil & water management practices (Fernandez, et al., 2002).

Because these cumulative impacts directly influence the nitrogen loads in receiving waters, and this prediction is dependent on successful modeling of the related watershed hydrology, it is important to predict the effects of drainage and land management practices from various fields on outflow from related watersheds (Amatya et al., 2001). Two kinds of models have been developed and evaluated for water quality planning and assessment. One is a comprehensive process-based model, and the other is a simplified lumped parameter model.

Comprehensive hydrology and water quality models are designed to simulate the impacts of the dynamics of various processes including surface runoff, subsurface flow, evapotranspiration, and channel flow. Depending on objectives, comprehensive models can be more complex both in hydrology of field and in-stream routing components and in water quality resulting from transport or transformation of nutrients. The runoff portion of a comprehensive model can be a simple coefficient of total rainfall or amount generated by using the SCS curve number method, or predicted by other complex process-based models like DRAINMOD. The degrees of complexity in transport and transformation components also vary from solving nonlinear multi parameter process-based in stream advective-dispersive transport and transformation equations to a lumped parameter equation with a first order kinetics (Amatya et al., 2003).

Many comprehensive models have been developed and applied in different spatial and temporal scales, e.g., ANSWERS (Beasley et al., 1980), AGNPS (Young et al., 1987), SWAT (Arnold et al., 1998), and DUFLOW (Aalderink et al., 1995). As an example, Chaplot (2005) recently studied the impact of the mesh size of the digital elevation model, DEM (from 20 to 500 m) and the soil map scale (1/25,000; 1/250,000 and; 1/500,000 scale) within the Soil and Water Assessment Tool (SWAT) to predict runoff, sediment, and NO_3–N loads at the outlet of an agricultural watershed. In this article, Chaplot showed that the upper limit of DEM mesh size is 50 m for simulating watershed loads. Decreasing the mesh size beyond this threshold does not substantially affect the computed runoff flux but generated prediction errors for nitrogen and sediment yields. Whatever the DEM mesh size considered, a detailed soil map has to be considered to accurately estimate the loads.

As decision tools, however, this type of model is difficult to use for assessing water quality because of their high data input requirements, problems in calibration and parameterization in large watersheds, and underlying uncertainties in the formulation processes and parameterization (Fernandez et al., 2002). Most decision makers working with watershed level management may only need planning level information that is obtained easily from the use of lumped parameter models. These models require minimal input data for simulation and are capable of accurate predictions on long temporal scales (Cooper and Bottcher, 1993; Fernandez et al., 2002). In addition, lumped parameter models require less time and effort than comprehensive models. If lumped parameter models can be properly coupled with error and uncertainty analysis, they can provide more information than the traditional comprehensive models (Fernandez et al., 2002).

For this reason, lumped parameter models have been more frequently used for watershed scale hydrological modeling (Haith and Shoemaker, 1987; Johnes, 1996; Wickham and Wade, 2002; Shrestha et al., 2008). A suite of simple lumped to comprehensive DRAINMOD-based models also have been developed to predict N loading at the outlets of poorly-drained coastal plain watersheds (Amatya et al., 2003, 2004; Fernandez et al., 2002, 2005, 2006, 2007; Skaggs et al., 2003).

DRAINMOD-N, a two-dimensional field scale model simulating the transport and transformation of N in soil profile, was developed based on the water balance calculations of DRAINMOD (Breve et al., 1997). Northcott et al. (2001) applied this model with surface runoff routing to simulate subsurface drainage and nitrate-nitrogen load in fields with irregular tile drainage systems in east central Illinois. This model treats only nitrate among all compartments of nitrogen. Youssef et al. (2005) improved DRAINMOD-N by adding detailed nitrogen compartments and simple carbon cycles on DRAINMOD-N. However, DRAINMOD-N and DRAINMOD-N II do not include detailed in-stream transport and process modules for a watershed scale modeling (Fernandez et al., 2005).

WATGIS, GIS based lumped parameter water quality model, was developed to estimate the nitrogen loading patterns (Fernandez et al., 2002). This model uses spatially distributed delivery ratio (DR) parameters to account for spatial and temporal nitrogen retention/loss along a drainage network. DR values are calculated from time of travel and an exponential decay model for in stream processes. DRAINMOD-DUFLOW, a process-based hydrology and hydraulics model, was used to develop relationships between travel time and daily flow depth, upstream drainage area and length of flow path from the field to the outlet. After these relationships are developed, WATGIS can be used to predict effects of land use and management practices on nutrient load at the outlet (Fernandez et al., 2002; Skaggs et al., 2003).

Fernandez et al. (2005) described two watershed-scale hydrology and water quality models (DRAINMOD-DUFLOW and DRAINMOD-W) that were used to evaluate the cumulative impacts of land use and management practices on downstream hydrology and nitrogen loading of a poorly drained coastal plain watershed. In both approaches, field hydrology was simulated using DRAINMOD (Skaggs, 1978). In DRAINMOD-DUFLOW, in-stream hydraulics and NO_3-N transport were predicted by DUFLOW (Aalderink et al., 1995). In the second approach with DRAINMOD-W, DRAINMOD was linked with a new one-dimensional canal and water quality model. These models were successfully tested using data from a 2950 ha managed forested watershed in the coastal plain of eastern North Carolina (Fernandez et al., 2005).

In the following year, the same authors developed DRAINMOD-GIS, a watershed scale, lumped parameter hydrology and water quality model based on DRAINMOD hydrology (Fernandez et al., 2006). This model links DRAINMOD field hydrology and a lumped parameter water quality model with a simplified drainage canal routing and in-stream process submodel. The model was tested with measured data from the watershed used in the previous study. Model predictions were in good agreement with measured data (within 1%), both measured outflows and NO_3-N loads. In a subsequent study, this model was applied to Chicod Creek watershed, a 11100 ha coastal plan watershed in North Carolina that is not intensively managed and documented. Hydrology predictions were within 5 % of the measured data and predicted mean monthly nitrate-nitrogen loads compared well with the measured data (Chescheir et al., 2004; Fernandez et al., 2007).

DRAINWAT-@RISK (Amatya et al., 2001) is a spreadsheet-based model where annual or seasonal field outflows simulated by another version of DRAINMOD-based watershed scale model, DRAINWAT, are used in @RISK decision tool system (Palisade Corporation, 1997; Skaggs et al., 2003). Other inputs including export concentration for each field, distance from the field outlet to the watershed outlet, and in-stream decay rate may be entered in the spreadsheet for simulating the nitrogen load at the watershed outlet. DRAINWAT (Amatya et al., 1997, 2003, 2004), the model used in this study, will be described in the section of 'Materials and Methods.' All of these models are listed and summarized in Table 2.1.

The objectives of this study are 1) to investigate the hydrology, primarily the drainage outflows, of the 1037 ha, poorly-drained agricultural watershed in the North Carolina coastal plain draining into a constructed wetland system using DRAINMOD based watershed scale model, DRAINWAT, 2) to analyze the sensitivity of the model input parameters, and 3) to explore the potential of its application to estimate nitrogen loading from the watershed using its outputs in a lumped parameter exponential decay model of first order kinetics. The simulation model was tested with 30 months (June 1st, 1999- December 31th, 2001) of outflow data collected by the UNC Institute of Marine Science on the study watershed located at Open Grounds Farm, Carteret County, North Carolina.

Table 2.1. DRAINMOD based models to predict hydrology and nutrient transport/transformation in watersheds with poorly drained soil.

Model #	*Model **	*Channel Flow*	*Feedback*	*Transport*	*Water quality*	*Uncertainty analysis*	*GIS*
1	DRAINMOD-N, N II	N/A	Yes (Hour)	ADR	ADR	Yes	No
2	DRAINWAT	St. Venant	Yes (Hour)	Slug flow	Exponential decay	No	No
3	DRAINMOD-DUFLOW	St. Venant	Yes (Hour)	ADR	Exponential decay	-	No
4	DRAINMOD-W	St. Venant	Yes (Day)	ADR	Exponential decay	-	No
5	DRAINMOD-GIS	Simplified Diffusion	No	Slug flow	Exponential decay	-	Yes
6	WATGIS	N/A	N/A	Slug flow	Exponential decay	Yes	Yes
7	DRAINWAT-@RISK	Average velocity	No	Slug flow	Exponential decay	Yes	No

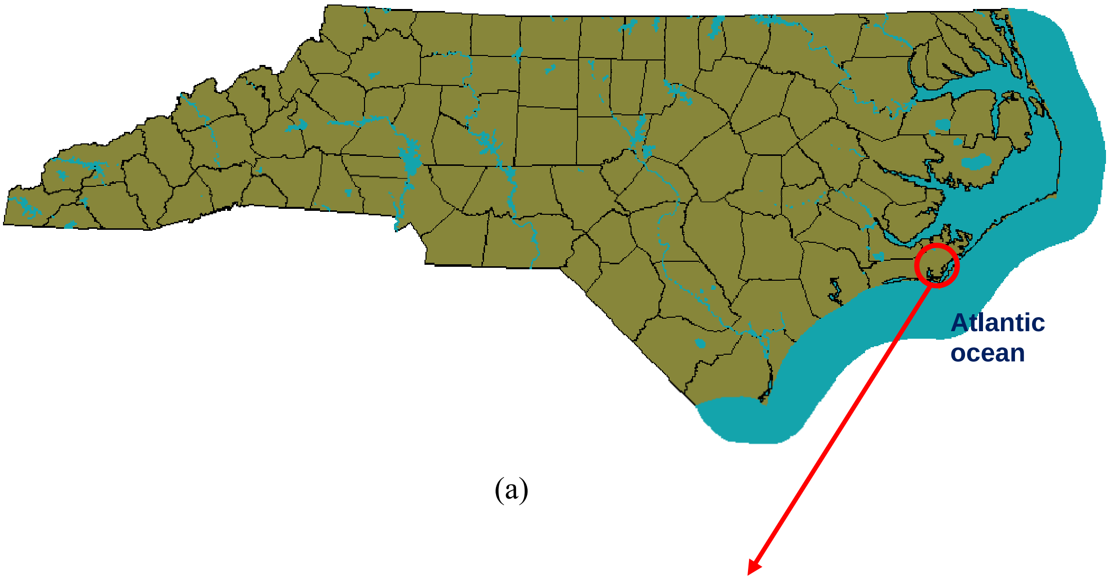
All models use DRAINMOD to predict both surface and subsurface drainage rates at the field scale (modified from Skaggs et al., 2003)

2.2. Materials and Methods

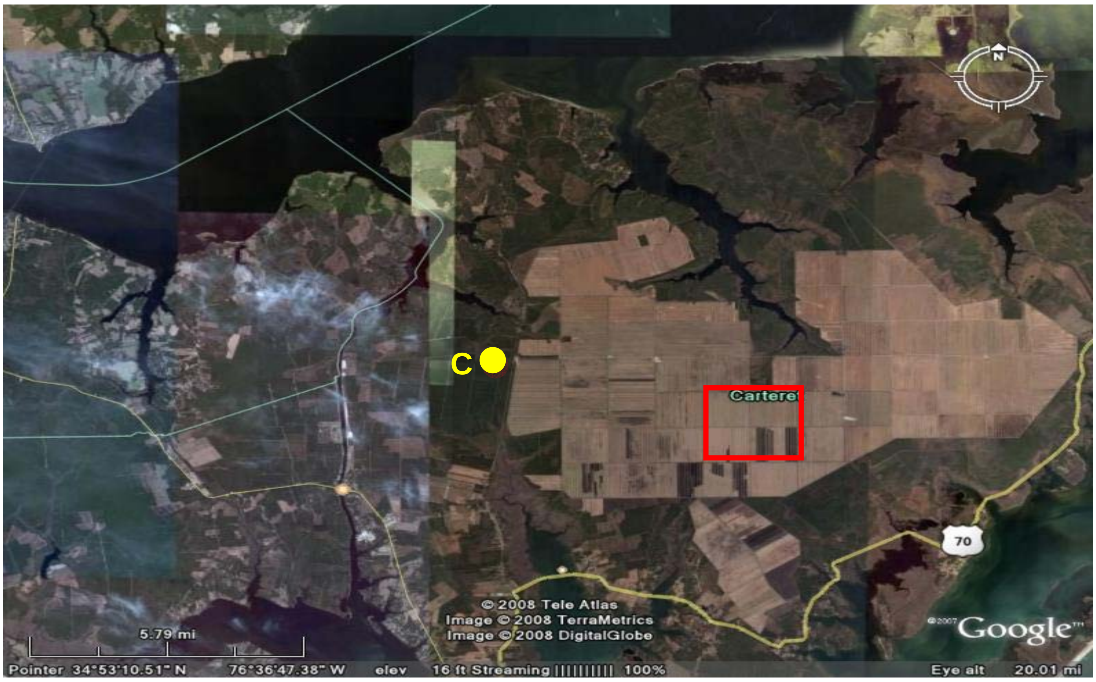
2.2.1. Site description

The DRAINWAT-based lumped parameter model described above was evaluated on a 1037 ha agricultural watershed, which is a part of the Open Grounds Farm located at 34^{o} 52’ latitude and 76^{o} 37’ longitude in eastern Carteret County in the lower coastal plain of North Carolina (Fig 2.1). This farm is adjacent west to the NC highway 70, and east to the large pine forest owned and managed by Weyerhaeuser Company. The farm is also located adjacent to the Albermarle-Pamlico Estuarine System, third largest such system in North America (Anderson and Onorato, 1994). This estuarine system is an important habitat for many birds and animals including numerous species of ducks, rails, snipe, river otters, raccoons, beavers, black bear, and deer (Fig 2.1.). Marshes next to this farm at the headwaters of the South River accommodate a food chain that maintains both shellfish and finfish. Estuarine creeks on Open Grounds Farm serve as nurseries for commercially important species including menhaden, mullet, brown shrimp, and blue crabs (Anderson and Onorato, 1994).

The farm is an 18,220 ha row-crop operation including 5,000 ha of corn, 5,000 ha of soybeans, and 1,600ha of cotton (Fig. 2.2.). Those fields were fertilized with fertilizers containing urea, ammonium, and nitrate (Poe et al., 2003). The drainage system of the watershed is made up of the field ditches and collector canals. The depth of field ditches is generally 90 cm and the ditches are spaced 100 m apart. The depth of collector canals is 180 cm, and the canal is spaced 1610 m apart (Fig 2.3). The slope of this watershed is less than 0.1%. This site is poorly drained, and has a shallow water table (Soil Survey of Carteret County, 2008).

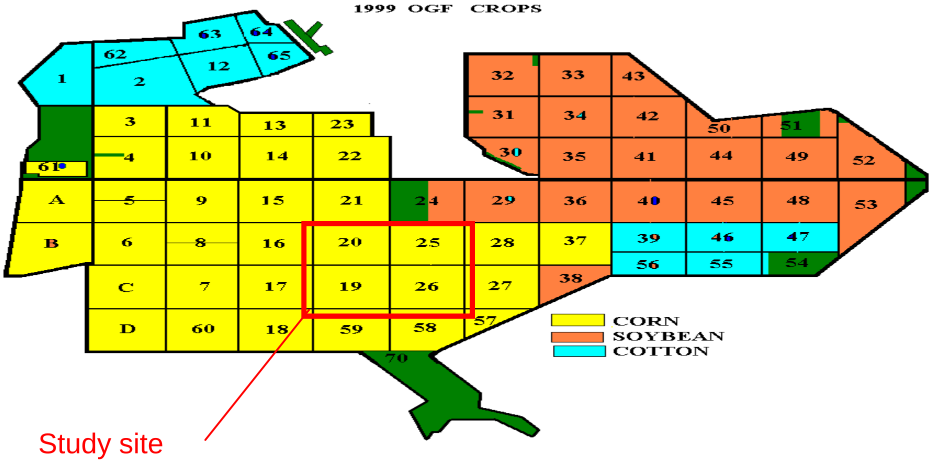


(a)

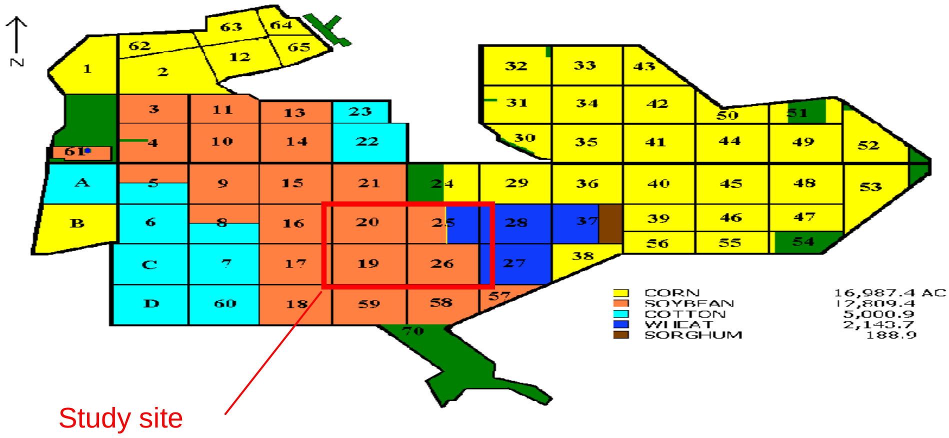


(b)

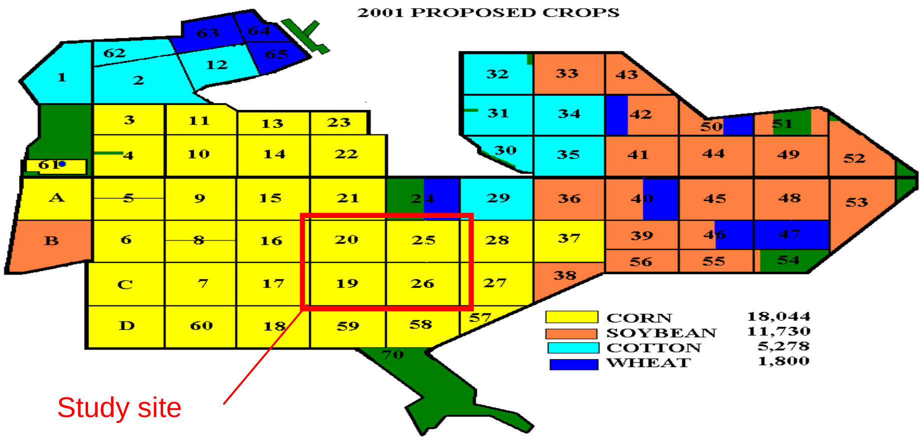
Fig. 2.1. Location map of the study watershed at Open Ground Farm in the Carteret County near the coast of North Carolina along with a satellite image.
(a) Location of Open Grounds Farm. (b) Satellite image showing the layout of the study site. C indicates the location of raingauge at the Weyerhaeuser's Carteret study site, and red box is the watershed studied. (Satellite image is from Google earth, 2008).



(a) 1999



(b) 2000



(c) 2001

Fig. 2.2. Cropping records of Open Grounds Farm from 1999 to 2001 (Gabriel Onorato, 2008. Personal communication). <u>NOT TO SCALE</u>

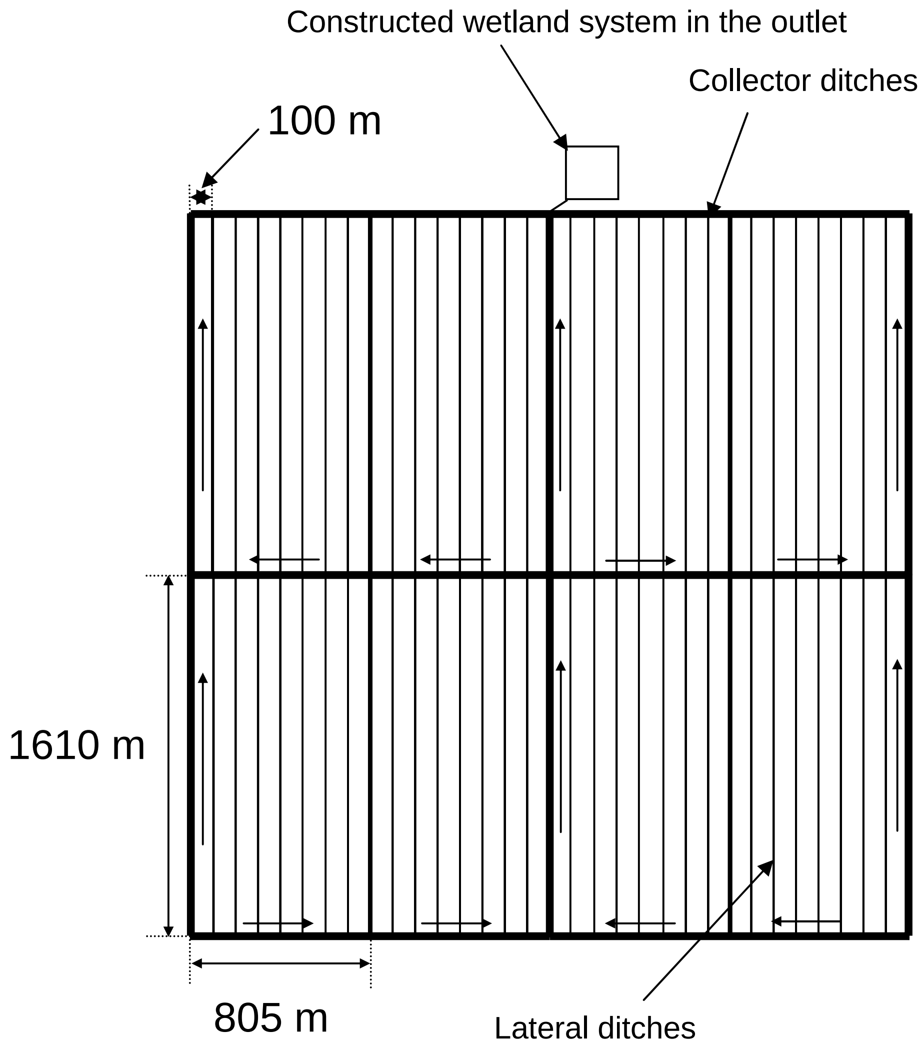


Fig. 2.3. Dimension of each field in the watersheds.

2.2.2. General climate of study site

The climate of eastern North Carolina is influenced by the Atlantic Ocean, which keeps temperatures mild in winter and moderate in the summer. The influence of the Atlantic Ocean raises the average winter temperature and decreases the average daily variation. As a result, the average daily maximum temperature on the coast is less than 89 °F (31.6 °C) during the summer, and the average daytime winter temperature in the coastal plain is usually in the mid-60's. Temperatures rarely drop below freezing even at night in the coastal plain. The average annual snowfall of coastal plain is usually less than 2.5 cm, and in some years there may be no snow or ice at all (State Climate Office of North Carolina, 2009).

Even though there are no distinct wet and dry seasons in North Carolina, average rainfall does vary during the year. Precipitation during summer is normally the greatest, and July is the wettest month. Summer rainfall is also the most variable, occurring mostly in connection with showers and thunderstorms. Daily showers of less than 30 minutes are common, but there are also periods of one to two weeks without rain in summer. Autumn is the driest season, with November as the driest month. Precipitation during winter and spring occurs mostly in connection with migratory low-pressure storms, which appear with greater regularity and in a more even distribution than summer showers. The average annual rainfall of the coastal plain ranged from 1200 to 1370 mm (Skaggs et al., 1991; State Climate Office of North Carolina, 2009).

Severe weather can have acute effects on water quality (Shelby et al., 2005), and it is not a rare event in eastern North Carolina. In some years several hurricanes or tropical storms can directly brush across the coastal areas. Shelby et al. (2005) studied the effect of severe weather on hydrology and water quality in 1999. They found that the two-month period of hurricanes (September and October in 1999) produced total nitrogen loads approximately equal to annual average loads. Sixty one percent of total annual nitrogen of 1999 exported from an agricultural subwatershed was lost in this two-month period. The effects of severe storms in North Carolina are not frequently documented, the study of Shelby et al. (2005) provides good information to understand how watersheds respond to extreme storm events. This information is relevant flow rates and loading rates in our study, especially for the second half of the first year (1999), which was affected by hurricanes.

2.2.3. Stream flow

Daily stream flow at the outlet of the 1037 ha study watershed (Fig 2.3) was calculated by Poe et al (2003) using measured values of stage and velocity of an automatic flow meter in an open channel (ISCO 750 area velocity flow module). Flow data for available periods were used to calibrate the model with the predicted outflow. However, problems with the measured flow data were observed during the field installation and data monitoring. There were a lot of physical problems getting the flow sensor to stay fixed in the channel. It would wash out during strong flow events, get covered with sediment or snagged by passing debris. Another major technical problem with the sensor itself was that it stays fixed on a velocity reading until it receives a strong enough signal to change it. This resulted in many "stuck on" values, since they were trying to measure low flow in a flat watershed where flow was at times reversed by wind tides. The sensor was finally moved from the open channel bed to the inside of the culvert pipe bottom only in 2002, after the end of this study, and better flow data were obtained after that. To address the "stuck on" velocity problem, a histogram analysis of all velocity data were performed, but repeat values (velocities that did not change during a 3 hr or greater period) were excluded. The most frequent baseflow velocity was taken and substituted for repeat velocities. During storm events we sometimes interpolated velocity data to complete hydrographs. Unfortunately, all of these adjustments may lead to some uncertainties in measured data that were used for model testing as described later in this chapter.

2.2.4. Water quality data

Water samples were collected by Poe et al. (2003) at the inlet and outlet of the constructed wetland and analyzed (2003) at the Institute of Marine Science, University of North Carolina-Chapel Hill in Morehead City, North Carolina. Measured nitrogen concentration and flow rates were used to calculate N loading rates. Nitrogen loading was monitored every day from July 1999 to November 2001, and the methods will be discussed in Chapter 3 (Table 2.2).

Table 2.2. Total amounts of atmospherically deposited N (ADN) and fertilizer present each year in the watershed. N is divided into dissolved inorganic N (DIN) and total N for each source. Units for N are thousands of kilograms (10^3 kg N). The percentage of each form of N that was ADN, and the percentage of the estimated total watershed N that entered the wetland are also included (Poe et al., 2003)**.**

Year	*Fertilizer applied (10^3 kg N)*		*ADN on Watershed (10^3 kg N)*		*% Watershed N as ADN*		*Load to Wetland (10^3 kg N)*		*% Watershed N Total to Wetland*	
	DIN	TN	DIN	TN	DIN	TN	DIN	TN	DIN	TN
1999	-	-	-	-	-	-	-	-	-	-
2000	23.8	35	3.69	5.27	13	13	2.1	4.7	8	12
2001	112	210	2.86	4.09	2	2	2.3	2.8	2	1

2.2.5. Model description

2.2.5.1. DRAINWAT model

DRAINWAT is a watershed-scale version of DRAINMOD (Skaggs, 1978) developed for drained agricultural and forest lands. It has been tested and modified to simulate the hydrological balance and nutrient loading from large watershed over 8000 ha (Amatya et al., 2003, 2004). DRAINMOD is an agricultural water management model which was originally developed to simulate the performance of drainage and related water management systems on a field scale (Skaggs, 1978, 1991). The model simulates the hydrology of poorly drained, high water table soils on an hour-by-hour, day-by-day basis for long periods of time. This model is also capable of predicting the effects of water management practices on water table depths, the soil water regime and crop yields.

DRAINMOD uses a combination of methods such as the Hooghoudt equations for modeling subsurface drainage, Kirkham equation for subsurface drainage during ponded conditions, the Green-Ampt method for infiltration, and other approximate methods for quantifying processes including runoff, evapotranspiration, and depression storage. The usage of this model has been extended to analyze the hydrology of certain types of wetlands to determine whether the wetland hydrologic criterion is satisfied for drained or partially drained sites, and to determine the hydraulic capacity of systems for land treatment of wastewater (Amatya et al., 2003).

DRAINWAT was developed by linking the forestry version of DRAINMOD (DRAINLOB) to the field and stream routing components of the watershed scale model (FLD&STRM) for agricultural lands. DRAINLOB (McCarthy et al., 1992) and FLD&STRM (Konyha and Skaggs, 1992) were other DRAINMOD based models.

The DRAINLOB, forestry version of DRAINMOD, was developed by replacing Hooghoudt equation of subsurface flow with the solutions of Boussinesq equations to increase prediction accuracy of subsurface drainage, which is one of the major contributing components of forest water balance. Evapotranspiration and subsurface flow components of DRAINMOD were modified and interception component was added in DRAINLOB, to consider these hydrologic processes (Amatya and Skaggs, 2001; Amatya et al., 2003).

In DRAINLOB, total evapotranspiration is calculated as the sum of wet canopy evaporation based on interception loss, dry transpiration, and evaporation from soil (McCarthy et al., 1992). Penman-Monteith evapotranspiration method was incorporated into DRAINLOB to estimate forest dry transpiration and wet canopy evaporation based on hourly weather data, leaf area index and stomatal conductance function. This model assumes that there is no transpiration when the canopy is completely wet. Based on the storage in the canopy, both transpiration and evaporation from the wet surfaces can occur during a given period. When there is no more wet canopy evaporation, the model calculates transpirational losses the same way as in DRAINMOD. Soil evaporation is estimated as a function of potential ET decreasing exponentially with leaf area index parameter (Amatya and Skaggs, 2001).

For predicting subsurface drainage, Kirkham's equation is used during ponded water conditions as it is in DRAINMOD. However, subsurface drainage flux is computed using average water table conditions in the entire soil profile for the rest of the periods. These water table conditions are obtained by solving non-linear Boussinesq equations (McCarthy et al., 1992). With this method, the total drainage flux including that flux from bank storage released during the transition from ponded water conditions to elliptic water table profile are addressed in DRAINLOB.

As described by Amatya et al. (1997), the FLD&STRM model (Konyha and Skaggs, 1992) was modified from DRAINMOD by adding a stream routing module and then extended to bigger size agricultural areas. Konyha and Skaggs (1992) incorporated the overland flow routing as well as in stream routing processes into DRAINMOD. The rainfall excess computed at the midpoint of the field goes first into the ditch via overland surface and then goes to the subcatchment outlet using the ditch routing component of the model. Flow in the ditch is simulated as an in-stream component within the subcatchment to account for the delay in flowing through the ditches to the outlet. In addition, the one dimensional St. Venant equations for unsteady state flows are solved using numerical procedures for the main in-stream channel flow routing in the watershed, taking care of changing ditch boundary conditions, backwater effects and tidal surges characteristics of watersheds in flat coastal plains (Konyha and Skaggs, 1992). The channel stream network is approximated as a specific number of stream elements and junctions at branching channels for conservation of mass and momentum are solved simultaneously to determine depth and flow rates. The model output of watershed outflow was found to be sensitive to time of concentration in the fields, bottom slope of the main channel, and channel roughness (Konyha and Skaggs, 1992; Amatya et al., 2003).

By combining these two models, DRAINWAT is capable of modeling agricultural, forested and mixed land use watersheds of medium scale, and also was successfully tested with large scale watersheds (Amatya et al., 2003). DRAINWAT was developed as a sequenced set of simulations to first combine outflow from each field with that of other fields that drain into the collector ditch of the subcatchment. The field can be defined as the smallest area with relatively uniform soil and land use conditions. The same model parameters can be applied in each field. An instantaneous unit hydrograph with a time of concentration concept is used to route the overland flow and ditch flow to the subcatchment outlet (Amatya et al., 2003).

The simulated outflow from all subcatchments provides lateral inflow to the main channel network. The lateral inflow is then routed through the channel system to the watershed outlet using numerical solutions of one dimensional St. Venant equations for mass conservation and momentum (Amatya et al., 2003).

2.2.5.2. Lumped parameter water quality model

2.2.5.2.1. In stream process

The lumped parameter model explored in this study assumes that the amount of nitrogen load decreases as drainage water flows through the canal network to the watershed outlet and it is exponentially dependent on residence time as water travels from the field edge to the outlet. Moreover, this model assumes that dispersion of the constituent concentration along the canals and ditches is insignificant due to minimal effects of tidal influence, and that there are no sources or sinks of N, other than that represented by natural decay (Loucks et al. (1981) as cited by Amatya et al. (2004).

The nitrogen transport can be approximated with a single attenuation coefficient. This approach is advantageous because it is easily combined with other existing hydrologic models and may be applied on time scales from individual storm events or seasonal or annual loads (Amatya et al., 2003).

Assuming a steady state condition, N concentration at any point in a canal stream network can be expressed shown in Equation (2.1) using two variables and one input parameters.

$$C_x = C_o e^{-kT} \quad \text{...}(2.1)$$

Where

C_x = N concentration at any point and time in a canal network

C_o = initial concentration at the field edge

k = N decay rate coefficient

T = travel time

(Amatya et al., 2004)

This method can be applied either as a part of the DRAINWAT model or as a separate algorithm with DRAINWAT outputs serving as inputs to determine loads at the outlet. This method for calculating experimental decay and load was accomplished in a separate algorithm by Amatya et al. (2004).

Exponential decay concepts for nutrient losses during transport in a stream have been used in other studies (Heatwole et al., 1987; Wagner et al., 1996; Amatya et al., 2004). It is recognized that this method is approximate, but it is easy to apply to other modeling studies and has been used successfully in other studies. The relationship for exponential decay of concentration can be also applied to nutrient load which is a product of the concentration and flow rate (Trepel and Palmeri, 2002; Amatya et al., 2004). Therefore, nutrient load delivered from field i to the watershed outlet can be expressed as:

$$L_i = L_{io} e^{-kTi} \quad \text{...} \quad (2.2)$$

Where

L_i = nutrient load delivered from field i to the watershed outlet (mass/area)

L_{io} = the nutrient load at the edge of field i

K = decay rate coefficient

Ti = the time required for the nutrient to be transported from the edge of field i to the watershed outlet. (Fernandez et al., 2002; Amatya et al., 2004)

After attenuation, Li is not the same as Lio, and the total cumulative annual load (L) of a nutrient at the watershed outlet can be expressed as:

$$L = \sum L_i \quad \text{...} \quad (2.3)$$

2.2.5.2.2. Delivery ratio

The delivery ratio (DRi) can be defined as the ratio of Li to Lio (Fernadez et al., 2002) and can be expressed as:

$$DR_i = L_i / L_{io} \quad \text{...} \quad (2.4)$$

Then combining equation (2.3) and equation (2.4):

$$DR_i = Le^{-kTi} \quad \text{..........} \quad (2.5)$$

Equation (2.5) implies that loss in the canal stream network increases with increasing distance or travel time from the field edge to the outlet, and as such the distribution of the sources within the basin is taken into account (De Wit, 2001). This method is simple and Amatya et al. (2001) implemented this in a spreadsheet environment.

2.2.5.2.3. Nutrient decay rate coefficient

The nutrient decay rate coefficient (k) is used to approximate the cumulative effects of several complex in-stream processes such as ammonification, nitrification, denitrification, sedimentation, and plant uptake and release. It is a fairly uncertain parameter with large variability. The values of k may vary with light, temperature, season, and location in the canal, with values increasing as flows become more infrequent or decrease in magnitude. Birgand et al. (2007) showed that most of the nitrate removal in a North Carolina canal draining agricultural lands was due to denitrification. Because denitrification rates increase with temperature, k would be smaller for wet, cold seasons than for dry, hot seasons. The rates may also vary with the form of nitrogen in the drainage water. Field effective values for k may shift with management practices and land use as the form of N at the field edge changes from labile to more recalcitrant forms of N, or vice versa.

2.2.5.2.4. Travel time

Travel time is the time required for the nutrient leaving the field edge to arrive at the watershed outlet and is calculated as

$$T_i = L / V \quad \text{..........} \quad (2.6)$$

Where

L = distance traveled by the nutrient from the field edge to the watershed outlet (m)

V = the average velocity of water as it moves through the canal stream network (m/day)

The average velocity and the travel time vary from event to event and may be estimated as a function of season and location in the watershed. The procedures for estimating nutrient load from a watershed using DRAINWAT outputs with the above equations for nutrient transport were presented elsewhere (Amatya et al., 2004).

2.2.6. Model inputs and parameterization

2.2.6.1. Model inputs

2.2.6.1.1. Precipitation

No rainfall data were measured at the study site, and the nearest rain gauge was at the Carteret site, an experimental watershed for forest water management study owned and managed by Weyerhaeuser company. Therefore, rainfall data used in this study were obtained from the raingauge C (Fig. 2.1) that collected rainfall on a continuous basis by an automatic tipping bucket rain gauge approximately 5 km from the study watershed as part of a study conducted by Amatya et al. (2006). The instantaneous rainfall data from the gauge for the 1999-2001 period were first processed to obtain data in daily, monthly, and annual total and then also for DRAINMOD format (See Appendix I).

2.2.6.1.2. Weather

There was no weather station at this study site. Daily average weather data such as air temperature, relative humidity, wind speed, and solar radiation were also borrowed from an automatic weather station located on watershed C at the Carteret site for the 1999-2001 periods (Amatya et al., 2006). Weather data is used for estimating daily potential evapotranspiration (PET).

2.2.6.1.3. Potential Evapotranspiration (PET)

In DRAINMOD and DRAINMOD-based models, daily PET can either be estimated within the model using the Thornthwaite method with daily minimum and maximum temperatures or input as daily file outside of the model. In this study, the daily PET for the model was calculated using the Penman-Monteith (PM) method for a grass reference for baseline simulation. Numerous methods for calculation of PET from weather data have been developed and tested for various environmental conditions. A large number of studies have reported that PM method is the most reliable if required weather and vegetation data are available (Amatya et al., 1995). However, inputs for PM equations are difficult and expensive to obtain for many applications. In those cases, PET methods dependent on more easily obtainable input such as net radiation (Priestley and Taylor, 1972), solar radiation (Turc, 1961), and temperature (Hargreaves and Samani, 1985; Thornthwaite, 1948) have been used as alternatives. Because temperature data are available for almost all applications, the Thornthwaite method with correction factors is widely used for estimating reference PET because Thornthwaite method generally underestimates PET during winter and overestimates during summer periods (Amatya et al., 1995).

Grass and alfalfa have been used as the reference crop. Although alfalfa has the physical characteristics closer to many agronomic crops than the grass, grass is more widely used for calculating reference-PET because: 1) the physical characteristics of the grass are better known and defined, and 2) measured evapotranspiration rates of grass are more readily available and accessible than that of alfalfa (Irmak and Haman, 2003). For this reason, grass references for calculating PET were used in this study.

2.2.6.2. Model parameterization

2.2.6.2.1. Watershed delineation and channel configuration

In order to run the DRAINWAT model, the 1037 ha watershed was first delineated into eight fields which were homogeneous in terms of soil type and crop. The size of each field was recommended to be less than 250 ha in the flat lower coastal plain to get a good prediction of the daily outflow (Konyha and Skaggs, 1992). The watershed boundary, the outlet, and the sub-boundaries for each individual field were identified. Each of the eight fields was assigned a number for identification in the model. The outlets of each field for simulating drainage outflow were identified and node numbers were assigned to each outlet along the stream (Fig. 2.4).

Designating nodes along the stream reaches is a way of identifying locations of the outlets of each individual field draining to the drainage canal, branching points, weir control structures, boundary conditions, and even any point of interest where you want to predict flow rate, velocity and canal water depth in the source codes. Water balance between nodes was estimated in the drainage-canal network by solving the non-linear 1-

D St. Venant equations for mass conservation and momentum to compute flow rate and depth (Konyha and Skaggs, 1992).

The nodes were numbered from '1' at the watershed outlet, and increase going upstream in the ditch-canal network. There is no need for a specific order in numbering left or right branch first or the main stream first with the nodes. A pre-determined flow path along the stream canal to the watershed outlet was identified for each field to calculate total distance for estimating travel time. In addition, the total number of nodes and all node numbers in the canal stream network in the watershed were identified.

2.2.6.2.2. Channel routing parameters

The main ditch canal network in the watershed was delineated along with the location of outlets draining each individual field in the network, as shown in Fig. 2.3 and Fig 2.4. Travel distance from the field outlet to the watershed outlet along the ditch canal network was measured for each of the fields using available maps (Fig. 2.4). The characteristics of the ditch and canal stream network in the watershed are given in Table 2.8. The dimensions of collector ditches and canals required for the channel routing component of the model are also shown in Table 2.8. A very flat channel bottom slope of 0.1 m/km, characteristic of the lower coastal plain and commonly used in other studies (i. e., Amatya et al., 2004, 2005), was assumed for the whole watershed for the baseline simulation. Manning's n of baseline simulation was 0.04 based on chow's suggestion for agricultural areas with field crops (Table 2.9) as discussed later in this chapter.

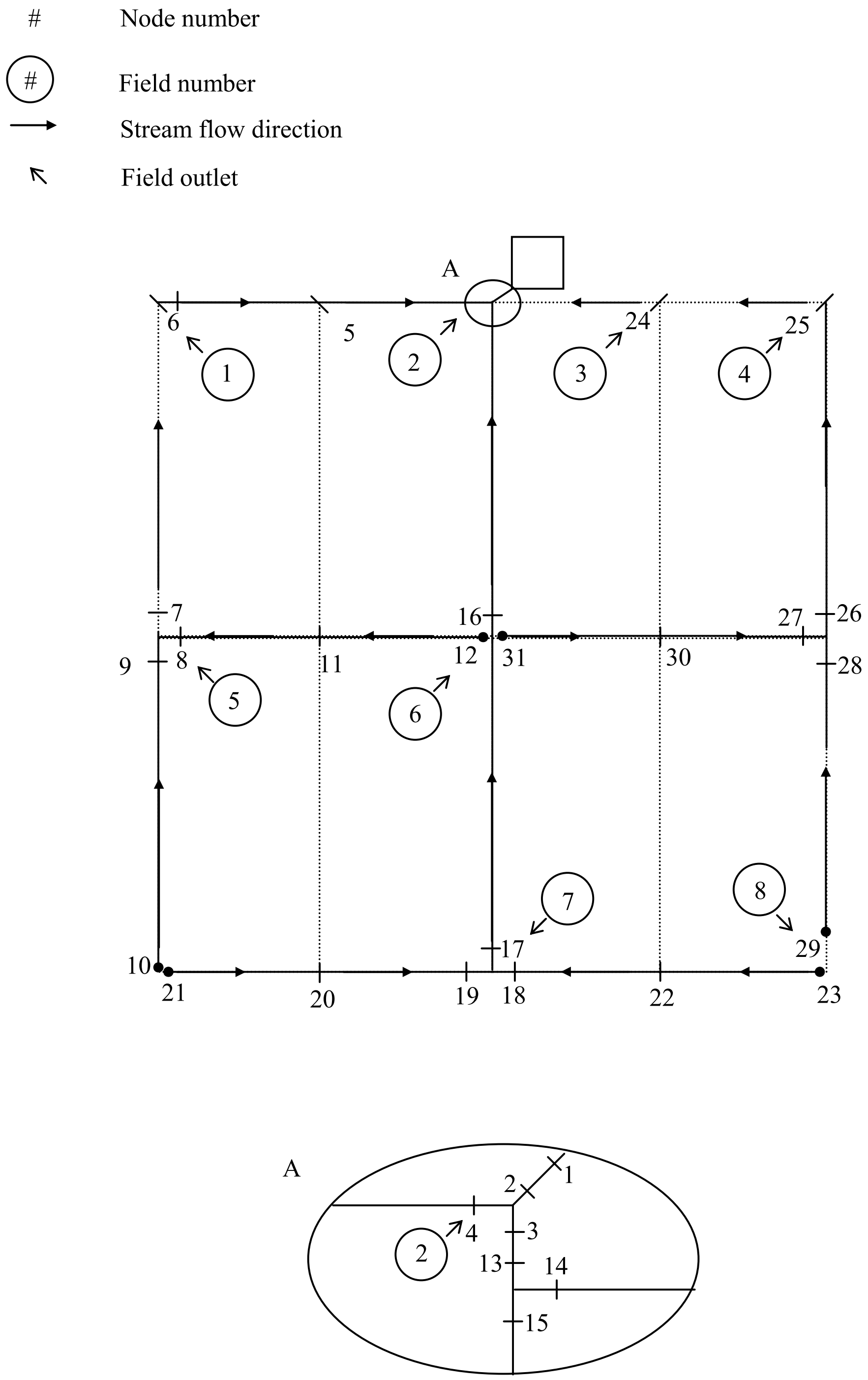


Fig. 2.4. Field layout along the channel for modeling the watershed. Each field has seven lateral ditches. The main outlet (node number: 1) of the watershed is the inlet to the constructed wetland system.

The watershed was delineated into identical eight subcatchments with the size of 129.5 ha. Each subcatchment is comprised of several fields drained by parallel lateral ditches (Fig. 2.3). Field delineation, areas of the subcatchments, and detailed dimensions of the lateral ditches are shown in Fig. 2.3, Fig. 2.4, and Table 2.3 as mentioned before in the previous sections. The sizes of ditches and drainage canals and other related input parameters are shown in Table 2.3.

Table 2.3. Characteristics of ditches and drainage canals.

Parameters	*Ditch*	*Drainage canal*
Ditch spacing (m)	100	-
Bottom width (m)	0.5	1.50 to 2.40
Ditch depth (m)	0.9	1.8
Side slope	0.38	0.58
Bottom slope	0.0001	0.0001
Manning’s #	0.04	0.04

2.2.6.2.3. Soil hydraulic properties and crops

Soils in the watershed are very poorly drained. Three organic soils, Belhaven (Loamy, mixed, dysic, thermic Terric Haplosaprists), Ponzer (Loamy, mixed, dysic, thermic Terric Haplosaprists), and Wasda (Fine-loamy, mixed, semiactive, acid, thermic Histic Humaquepts) were identified as the dominant soils in the watershed studied.

Five (# 1, 5, 6, 7, 8) among eight delineated fields contain the Belhaven series (Ba, Table 2.3). Thicknesses of organic matter vary from 40 to 130 cm. Depth to impermeable layers is greater than 150 cm (Table 2.4, 2.5) and impermeable layers were sometimes down to 270 cm from soil surface (Table 2.5). Depths to the seasonal high water table are shallow (less than 30 cm, Table 2.4). Organic horizons of Belhaven soils are very acidic unless they are limed. The acidity of mineral horizons underneath organic horizons varies from extremely acidic to moderately alkaline. The amount of fiber is very high in the lower layer. The organic material of this layer is paste-like, has a greasy feel (colloidal) and is massive under natural wet conditions. Upon aeration after drainage, structure of the organic material evolves. Excessive drying causes shrinkage and hard subangular blocky peds to form. These peds dry irreversibly. Logs, stumps, and fragments of wood occupy 0 to 5 percent of the upper organic horizons in cultivated and cleared areas, and 5 to 35 percent in the areas that are not cultivated. (Soil Survey Staff, 2009).

Two fields (# 2, 4) were occupied by the Ponzer soil (Po, Table 2.3). Ponzer soils exist in similar drainage conditions and landscapes where Belhaven soils are found. Depth to impermeable layers is greater than 150 cm (Table 2.4, 2.5) and thickness of soil profile (from soil surface to impermeable layer) was 250 cm in some places (Table 2.5), but depths to the seasonal high water table are less than 30 cm (Table 2.4). Compared to colloidal, paste-like organic layers of Belhaven, Ponzer soils have non-colloidal organic layers with the color of 7.5YR to 2.5Y in hue. Thickness of the organic materials commonly is 40 to 75 cm, but sometimes up to 130 cm. The organic materials are also very acidic unless the soil surface has been limed. The underlying mineral horizons are extremely acid through mildly alkaline. Logs, stumps, and fragments of wood occupy 0 to 20 percent of the organic layers. Flakes of mica are few to common in the mineral horizons of some pedons (Soil Survey Staff, 2009).

Wasda soil (Wa) was found in only one field (# 3, Table 2.3). The typical soil profile of Wasda is somewhat different from Belhaven and Ponzer. The soil profiles of Belhaven and Ponzer are mainly composed of O and C horizon, but Wasda has relatively well-developed mineral horizons such as A and B from 36 to 107 cm depth (Table 2.4). Organic material and clayey materials over unconforming sediments ranges from 100 to more than 150 cm. Depth to Bedrock is greater than 150 cm, but soil depth was relatively shallow compared with Belhaven and Ponzer (Table 2.5). Depth to Seasonal High Water Table is 0 to 30 cm, December to May. Reaction ranges from extremely acid to strongly acid in A and Bg horizons and very strongly acid *to mildly acid in the Cg horizon.* Mica, feldspar, and other weatherable minerals range from few to common in the B and C horizons. More information can be found at the Official Soil Series Website in USDA-NRCS (Soil Survey Staff, 2009).

Water characteristics of the three soil types were published values as reported in Amatya et al. (1997) for Ponzer, and Skaggs and Nassadzadeh-Tabrizi (1986) for Belhaven and Wasda (Table 2.5).

Table 2.4. Soil hydraulic properties from soil survey.

Soils	*Depth (cm)*	*Texture*	*Permeability (cm/hr)*	*High water table*		
				Depth (cm)	Kind	Months
Belhaven	0-66	Muck	0.5-15.0	0-30	Apparent	Dec-May
	66-81	Sandy loam, Fine sandy loam	5.1-15.0			
	81-165	Loam, Clay loam, Sandy clay loam	0.5-1. 6			
	165-183	Loamy sand, Fine sand	15.0-50.8			
Ponzer	0-107	Muck	0.15-5.1	0-30	Apparent	Dec-May
	107-140	Loam, Sandy loam, mucky loam	0.15-5.1			
	140-152	Variable	--			
Wasda	0-36	Muck	0.5-1.5	0-30	Apparent	Dec-May
	36-107	Clay loam, Sandy clay loam, Sandy loam	1.5-5.1			
	107-152	Sandy loam, loam	1.5-5.1			
	152-188	Sand	15.2-50.8			

Monthly variations of rooting depths of crops were shown in Table 2.6. The rooting depth is required by DRAINMOD and other DRAINMOD based models. This depth defines the zone from which water can be removed to supply evapotranspiration. The dry zone can extend equally to the root zone, and the sum of wet and dry zone depths supplies the water table depth on a time step (Singh et al., 2006). Because most water is extracted near the soil surface where the majority of roots grow, the effective rooting depth was assumed to be 60% of the maximum rooting depth (Skaggs 1990; Cox et al., 1994). The numbers of rooting depths of crop in Table 2.6 are the default values commonly used in DRAINMOD and DRAINMOD-based model (Skaggs, 1990).

Table 2.5. Soil hydraulic properties from literatures used as inputs in DRAINWAT

	Soil type		
Property	*Belhaven*	*Ponzer*	*Wasda*
Depth to impermeable layer (cm)	270	250	200
Hydraulic conductivity (cm/hr)	20 (0-30)	20.8 (0-112)	20 (0-30)
(Depth range (cm))*	1.0 (30-80)	2.1 (112-142)	0.4 (30-80)
	0.0 (80-270)	7.5 (142-250)	1.0 (80-200)
Water content at wilting point (cm^3/cm^3)	0.45	0.29	0.45
Saturated water content at root zone (cm^3/cm^3)	0.73	0.63	0.76

*The numbers in parenthesis represent the range in depth (cm) for that conductivity value.
(Amatya et al, 1997; Skaggs and Nassehzadeh-Tabrizi, 1986; Soil Survey of Carteret County, NC, 2008)

Table 2.6. Rooting depth of the crops used in DRAINMOD in 1999-2001.

Date	*Root depth (cm)*
January 1st	3.0
April 16th	3.0
May 4th	4.0
May 17th	15.0
June 1st	25.0
June 20th	30.0
July 18th	30.0
August 20th	20.0
September 24th	10.0
September 25th	3.0
December 31st	3.0

(Skaggs, 1990)

Corn and soybean yields based on the soil types were compared and summarized in Table 2.7. Corn yields on all soil types were greater than soybean yields in this area, but there were no substantial differences among different soil types. Kaspar et al. (2004) reported the relationship of corn and soybean yield to soil and terrain properties. In our study, however, it seems that the differences of soil properties among three soil types did not have any substantial effects on crop yield.

Table 2.7. Yields per acre of crops. (Unit: Bu)

Soil name (Map symbol)	*Corn*	*Soybean*
Belhaven (Ba)	125	40
Ponzer (Po)	130	40
Wasda (Wa)	130	40

(From Soil survey of Carteret County, NC. 2008)

Manning's Equation, the most commonly used empirical equation for open channel flow, was used in this model to calculate the flow velocity.

$$v = R^{2/3} S^{1/2} / n \quad \text{.......} \quad (2.7)$$

where,

v = outflow velocity (m/s)
R = Hydraulic Radius (m) = A/P_w
A = Cross-Sectional Area of Flow (m^2)
P_w = Wetted Perimeter (m)
S = Channel Slope
n = Manning's Roughness Coefficient

(Arcement Jr. and Schneider, 2009)

Then the flow rate (Q) at each node of the canal stream reach was calculated using the

$$Q = vA \quad \text{.......} \quad (2.8)$$

2.2.6.2.4. Field design parameters.

Field parameters for DRAINWAT baseline simulations are shown in Table 2.3. Regardless of soil types, same values of field parameters were used for all fields due to homogeneous land use and topography. For baseline simulation, the field area measured was 129.50 ha, field slope of 0.01%, maximum surface depressional storage (STMAX) of 1.0 assumed. Time of concentration, travel time for runoff to move from the most reomote point in the catchment to the outlet, was estimated to be 4.65 hours using the equation 2.9 suggested by Huggins and Burney (1982).

$$T_c = 0.02 L_c^{0.77} S_c^{-0.385} + \left| \frac{2nL_o}{\sqrt{S_o}} \right|^{0.467} \quad \text{.......} \quad (2.9)$$

(Huggins and Burney, 1982)

Where:

Tc = time of concentration, minutes

Lc = Length of channel reach, m

Sc = average slope of channel reach, m/m

n = Manning's roughness coefficient

Lo = Length of overland flow, m

So = Slope along that path, m/m

Table 2.8. Field parameters for DRAINWAT baseline simulation

Field #	*Dominant soil type*	*Soil type code*	*Area (ha)*	*STMAX (cm)*	*Field slope (%)*	*Time of concentration (hr)*
1	Belhaven	Ba	129.50	1.0	0.01	4.65
2	Ponzer	Po	129.50	1.0	0.01	4.65
3	Wasda	Wa	129.50	1.0	0.01	4.65
4	Ponzer	Po	129.50	1.0	0.01	4.65
5	Belhaven	Ba	129.50	1.0	0.01	4.65
6	Belhaven	Ba	129.50	1.0	0.01	4.65
7	Belhaven	Ba	129.50	1.0	0.01	4.65
8	Belhaven	Ba	129.50	1.0	0.01	4.65

2.2.6.2.5. Model simulation

As described earlier, weather data for the three - year period from the nearest weather station at the Carteret site were used for the rainfall and potential evapotranspiration model inputs. A uniform spatial distribution of rainfall and PET was assumed for the whole watershed. Hourly rainfall and daily PET in DRAINMOD for the watershed used in the model are presented in Appendix I for the 30 month simulation period from July 1st 1999 to December 31st 2001.

2.2.6.2.6. Model outputs

DRAINWAT simulation was then run using source.exe in standard DOS window. After running the model, several output files were produced. The WS22.out file has the information about year, day of the year or hour of the day, predicted results of 24 hourly stream depth at the node of interest (this is usually watershed outlet unless specified), daily cumulative outflow (in depth), daily cumulative inflow (in depth), flood storage on overbanks (in depth), surface runoff, subsurface drainage outflow, water table depth for an individual field, total rainfall, net rainfall, interception loss, free throughfall, and evapotranspiration (Amatya, 1993).

The daily outflows predicted by the model need to be in close agreement with measured data for the study watershed before proceeding with water quality simulation because the lumped parameter watershed water quality modeling relies more on predicted outflows both at the field edge and the watershed outlet than in the nutrient concentrations, transport and transformation parts of the water quality modeling.

Simulated outputs for three years of daily outflows at the watershed outlet were compared to measured data to test the performance of the hydrologic model. Predicted daily outflows from each field and daily average velocity at each node in the canal network were saved in output files. A separate exponential decay routine was developed to compute daily total nitrogen load arriving at the outlet from each field using equations 2.2 through 2.6 as discussed earlier (Amatya et al., 2004). Predicted daily field outflows and daily average velocities at nodes along the flow path were inputs to this routine. DRAINWAT also produced other output files such as STR21.OUT, SNAPSHOT.OUT, and DMDATAxx.OUT for each of the individual fields (Amatya, 1993).

The 'STR21.OUT' file has the information on the stream water balance, and the data that this file contains are as follows. Year, day of the year for daily simulation or hour of the day for hourly simulation, volume of outflow, volume of inflow, change in storage, error in water balance, number of iterations, indicator for repeating field hydrology, dry channel conditions. The 'SNAPSHOT.OUT' file has the snapshot information on the outputs of fields and stream for the specified date and time, and the data that this file contains are as follows.

- Stream conditions: node number, depth, channel flow rate, lateral inflow rate flood volume.
- Field conditions: field number, total runoff, surface runoff, subsurface drainage, ditch water level, and water table depth.

The 'DMDATAxx.OUT' file has the information on the DRAINMOD hydrology output for each field, and the data that this file contains are as follows (Amatya, 1993).

For each month: Day, rainfall, infiltration, ET, drainage, total air volume, dry zone depth, depth to the water table, storage, runoff, total water loss, canopy loss, canopy drip, free throughfall, total throughfall, and canopy storage.

Other output files such as DMYGEN03.out (outputs of all DRAINMOD input parameters read from '*.DAT* file for each field), FLDGEN04.OUT (outputs of all field parameters read from '*.F11' file for each field), and STRGEN03.OUT (outputs of all stream routing parameters read from '*.F16' input file) were produced. For the water quality simulation, two other output files, 'NOD_VEL.OUT' and 'FLD_FLO.OUT', were produced. These two files were made in ASCII format (Amatya, 2000).

The 'NOD_VEL.OUT' file contains simulated daily mean velocities of water movement in the canal/stream based on 24 hourly values for all the nodes in the watershed. The 'FLD_FLO.OUT' file has the simulated daily total outflows from each field. The flow data in those files are the final values calculated after the repetition of mass balance calculation at the end of the day between the field ditch and the canal in the routing component of the model.

The DRAINWAT model measured the intermediate distance between each pair of nodes from the field outlet to the watershed outlet. A reach file named "*.rch" were made based on this information. The number of intermediate distances will be n-1 (Amatya, 2000).

As a result, the model calculated the daily velocity for each segment of flow path as an average velocity of two adjacent nodes identified earlier for each field from the field edge to the watershed outlet. Travel time was computed for all segments of the flow path for each field and for each day of the year based on this average daily velocity and intermediate distance between two adjacent nodes calculated earlier. Travel times for all segments are added to get total travel time in the flow path for each field for each day (Amatya, 2000; Amatya et al., 2003, 2004).

2.2.6.2.7. Nutrient parameters

Nutrient load at a field edge can be estimated as the product of an export coefficient and the field area. The export coefficient is based on soil type, land, and water management practices, crop, and fertilization (Amatya, 2000). Tables of export coefficients would generally be prepared for a specific location, which would implicitly incorporate effects of weather and climate on the values. Export coefficient can be estimated from published data or from measured data, or process-based models such as DRAINMOD-N (Breve et al., 1997). The value can also be estimated as a product of outflow and export concentration (Amatya et al., 2004).

Total N was estimated using a spreadsheet. DRAINWAT-predicted total annual outflow from each field was multiplied by the average total N concentration to obtain total N load at the field edge in the spreadsheet. Total distance from the field edge to the watershed outlet for each field was put into the spreadsheet. Travel time for each field was estimated using a single value of average velocity throughout the network, assuming it independent of season and location. The average velocity was obtained from DRAINWAT simulations for 30 month period (July 1st, 1999-December 31st, 2001). Travel time for each field was then obtained by dividing the distance from the field edge to the outlet by this average velocity (Amatya et al., 2004). Due to the inaccuracy of outflow measured data, however, these results could not be calibrated, presented and discussed in this study.

2.2.7. Evaluation of model performance

2.2.7.1. Storm event analysis

Six storm events were analyzed using most reliable measured data to evaluate DRAINWAT performance in predicting outflow event on daily basis. All events were defined to start at the time of the initial rise of the hydrograph and extend for at least 5 days to include subsurface drainage caused by the storms (Amatya et al., 1997).

2.2.7.2. Sensitivity analysis

Model sensitivity was tested to identify the parameters that have the greatest effect on the simulation results. Several researchers of DRAINMOD and DRAINMOD based models have studied which parameters have the greatest effect on the model output (Skaggs et al., 1991; Fernandez et al., 2002; Haan and Skaggs, 2003; Wang et al., 2005). Four input parameters that have the potential to influence the outflow simulation were selected and analyzed. Those parameters are: Manning's roughness coefficient, depressional storage, channel bedslope, and PET by three methods.

Manning's roughness cannot be measured directly. It varies widely. Depressional storage is complex to measure. Channel bottom slope is generally flat, but direct measurement of field is not easy. PET is important in hydrologic cycle, but hard to measure accurately.

Soil hydraulic conductivity and water table depth are also considerable for sensitivity analysis. However, the sensitivity of soil hydraulic conductivity was already extensively tested by other studies (i. e., Konyha and Skaggs, 1992; Haan and Skaggs, 2003). Water table depth is important at the initial stage of simulation, but empirically it is not sensitive after 2 ~ 3 months.

2.2.7.2.1. Manning's roughness

Manning's roughness coefficient (n) is the roughness parameters representing the resistance to flood flows in channels and flood plains. Manning's formula, shown in the previous section, is an indirect computation of stream flow, and has been applied in flood-plain management, in flood insurance studies, and in the design of bridges and highways across flood plains (Arcement and Scheneider, 2009). This coefficient can be influenced by a lot of factors including soil types and land cover on the channel bed and banks. Arcement and Scheneider (2009) suggested five adjustment factors, which were channel irregularities, variation in channel cross section, obstructions in the channel (i.e., logs, stumps, boulders, etc.), channel vegetation, and meandering. They insisted that these factors increase channel roughness, and n values need to be adjusted accordingly. Table 2.9 shows the base n values under various conditions. In this study, the value of n for baseline simulation was 0.04 based on the "cultivated areas with mature field crops" in Table 2.9(b).

Table 2.9. Base Values of Manning's *n* for Channels.

Type of Channel and Description	*Minimum*	*Normal*	*Maximum*
1. Main Channels			
a. clean, straight, full stage, no rifts or deep pools	0.025	0.030	0.033
b. same as above, but more stones and weeds	0.030	0.035	0.040
c. clean, winding, some pools and shoals	0.033	0.040	0.045
d. same as above, but some weeds and stones	0.035	0.045	0.050
e. same as above, lower stages, more ineffective slopes and sections	0.040	0.048	0.055
f. same as "d" with more stones	0.045	0.050	0.060
g. sluggish reaches, weedy, deep pools	0.050	0.070	0.080
h. very weedy reaches, deep pools, or floodways with heavy stand of timber and underbrush	0.075	0.100	0.150
2. Floodplains			
a. Pasture, no brush			
1.short grass	0.025	0.030	0.035
2. high grass	0.030	0.035	0.050
b. Cultivated areas			
1. no crop	0.020	0.030	0.040
2. mature row crops	0.025	0.035	0.045
3. mature field crops	0.030	0.040	0.050
c. Brush			
1. scattered brush, heavy weeds	0.035	0.050	0.070
2. light brush and trees, in winter	0.035	0.050	0.060
3. light brush and trees, in summer	0.040	0.060	0.080
4. medium to dense brush, in winter	0.045	0.070	0.110
5. medium to dense brush, in summer	0.070	0.100	0.160
d. Trees			
1. dense willows, summer, straight	0.110	0.150	0.200
2. cleared land with tree stumps, no sprouts	0.030	0.040	0.050
3. same as above, but with heavy growth of sprouts	0.050	0.060	0.080
4. heavy stand of timber, a few down trees, little undergrowth, flood stage below branches	0.080	0.100	0.120
5. same as 4. with flood stage reaching branches	0.100	0.120	0.160

(Chow, 1959; as cited and summarized by Furniss et al., 2008)

2.2.7.2.2. Depressional storage

STMAX and STORRO have been used to describe the surface drainage design. STMAX, surface depressional storage, represents the maximum surface storage in cm which must be filled before surface runoff happens. STORRO means the storage in local depressions such that water is prevented from moving freely to a position over the subsurface drain, and suggested as 1/2 of STMAX generally. The value of STORRO in a field guides whether DRAINMOD and its associated models use Hooghoudt's equation or Kirkham's equation. If the water depth on the surface is less than STORRO, then Hooghoudt's equation is used in DRAINMOD, otherwise, Kirkham's equation is used (Skaggs, 1990).

Surface depressional storage influences the hydrology of an agricultural field. As the depth of depressional storage decreases, the percentage of time in which the surface is ponded decreases, and average depth of the water table increases (Skaggs et al., 1991) Sensitivity analyses of STMAX and STORRO were carried out using values from 2.5 mm to 15 mm of STMAX. The simulation results of hydrology in each value were compared in study year.

2.2.7.2.3. Channel bedslope

"It's flat! It's low!" Philips (1997) described the slope of lower coastal plain in North Carolina. As he described, eastern North Carolina is geographically stable and geomorphic risk factors such as local slope are low (Philips, 1997). For this reason, drainage systems such as ditches and channels are necessary. As an example, Lindow (2007) restored a stream reach which was previously straightened stream near Cove City in the Lower Coastal Plain in North Carolina. An expected dynamic equilibrium was defined for restored streams, and equated changes in stream form to fluvial and geotechnical parameters for predictive analysis. The value of channel bed slope in his study was 0.001. This is flat, but channel bed slope for my study site is even more flat. The bed slope for baseline simulation was assumed to be 0.0001.

2.2.7.2.4. Potential evapotranspiration

The daily outflow sensitivities of the Penman-Monteith (PM) method and the Thornthwaite method with monthly correction factors were tested in this study. The PM method is one of the most accurate among commonly used methods of PET calculation as discussed earlier in the section 2.2.6.1.3. and this method was used for baseline simulation. Two sets of monthly correction factors for the Thornthwaite method were used for sensitivity analysis (Table 2.10). The first set of monthly correction factors, named as Thorn1 in this study, was developed as an average for the NC coastal plain by Amatya et al (1995). Thorn2, the second set of monthly correction factors, was used in earlier DRAINMOD simulations for agricultural lands in North Carolina (Chescheir et al., 1990).

The PM method and the Thornthwaite method have different input data requirements. PM requires detailed information on meteorological variables such as net radiation, wind speed, relative humidity, and vegetation parameters. On the contrary, the Thornthwaite method is based on air temperature only.

Table 2.10. Correction factors for calculating potential evapotranspiration.

Month	*PM*	*Thorn1*	*Thorn2*
Jan	1	1.94	2.52
Feb	1	2.32	3.00
Mar	1	2.09	2.49
Apr	1	1.73	1.69
May	1	1.23	1.31
Jun	1	1.02	0.99
Jul	1	0.89	0.90
Aug	1	0.84	0.87
Sep	1	0.95	0.94
Oct	1	1.07	1.20
Nov	1	1.23	1.45
Dec	1	1.38	2.01

2.2.7.3. Lumped parameter water quality modeling

For water quality modeling using the lumped parameter exponential decay model, the seasonal nitrogen concentration values can be estimated. This value depends on measured or published data for each individual field based on a given soil type, land use, and water management practice (Amatya et al., 2003).

The seasonal nutrient decay rate constant (k) depends on measured or published data. Separate values of K were selected for the wet and dry seasons for simulating nitrogen loading at the outlet. Most of the values reported in the literature for k as a result of denitrification vary between 0.01 and 0.20 day^{-1} (Bowie et al., 1985). A constant decay rate value of 0.05 used by Fernandez et al.(2002) was used in this application. This value also falls within the in stream loss rate range of total N recently compiled by Alexander et al. (2000) for streams of Chesapeake Bay.

The travel time, initial field loading at the field edge, and nutrient decay rate constant were used to compute the loading at the outlet contributed by each field for that day using an exponential decay rate equation. This equation accounts for the delivery of the nutrient loading from the field edge to the watershed outlet.

Wet and dry seasons are arbitrarily defined using days of the year inside the codes. April 15 to November 15 (day 105 and day 319 for non-leap year, and day 106 and day 320 for leap year, respectively) defined as a dry summer-fall period and November 16 to April 14 (day 320 and day 104 for non-leap year, and day 321 and day 105 a leap year. respectively) is defined as a wet winter period.

The model then puts together the total flows and total nutrient load at each field edge and the portion of the load that arrives at the watershed outlet from each field using the exponential decay rate method for the wet and dry seasons of each year. The 'WQLOAD.OUT' file will contain this information in ASCII format (Amatya et al., 2003). If the predicted outflows, velocities, and field concentration are reliable, the prediction of nutrient loading in the watershed is dependent on the decay parameter 'K' in the exponential decay rate method.

Finally, the simulations of water quality were conducted using a program developed by Amatya (2000). However, these results were not presented and discussed due to the inaccuracy of outflow calibration results caused by limited measured data. Only methodology for water quality simulation in this section was presented here as a reference for next study.

2.2.7.4. Statistical analysis

The accuracy of the model to predict the monthly flows and nitrogen loading at the outlet of the watershed was determined using a number of different statistical methods. The most common method is the coefficient of determination (r^2) (Aitken, 1973; Fernandez et al., 2005). The coefficient of determination is obtained from the regression of the predicted values versus the observed values.

$$r^2 = 1 - \frac{\sum (Q_o - Q_e)^2}{\sum (Q_o - Q_m)^2} \quad \text{..........} (2.10)$$

Where Q_o are observed values, and Q_e are the estimated values obtained from the regression line of Q_o on Q_S. Q_S are simulated values by model, and Q_m is the average of observed values. This coefficient is widely used in various modeling studies to determine how well the simulated values fit the observed values (Draper and Smith, 1998).

Another criterion is the Nash-Sutcliffe coefficient of efficiency E (Nash and Sutcliffe, 1970; Aitken, 1973), which is defined as:

$$E = 1 - \frac{\sum (Q_o - Q_s)^2}{\sum (Q_o - Q_m)^2} \quad \text{..........} (2.11)$$

E is similar to r^2, but it is not identical. The values of E range from negative infinity to 1, where 1 is perfect model prediction. E expresses the fraction of the error variance relative to the variance of the measured values. If E value is over 0.75, simulation results are considered to be good. If E value is between 0.36 and 0.75, the simulation results are considered satisfactory (Fernandez et al., 2005).

After the sensitivity analyses of the four parameters, Student t-test were also used to determine whether the simulated outflow values are statistically different. MS-Excel was used for statistical analysis.

2.3. Results and Discussion

2.3.1. Measured data

2.3.1.1. Measured hydrology and meteorology data

2.3.1.1.1. Precipitation

A set of rainfall data observed from the Carteret site (Amatya et al., 2006) was compared with long term data in the Morehead City, NC (Table 2.11).

As Shelby et al. (2005) described based on the measured rainfall data at their study site in Plymouth, the precipitation in eastern North Carolina during 1999 was "one of the extremes" even though the annual total of 1377 mm is similar to that of the long term average of 1391 mm at the nearest weather station in Morehead City, NC (Table 2.11). This was primarily due to a large seasonal variation in rainfall. The winter (January-March) and spring (April-June) of 1999 was dry. The total average rainfall of this period was just 468 mm, only 77 % of long term average (609 mm). Following this dry period, three hurricanes hit this area producing tremendous amount of rain in one and a half months (Shelby et al, 2005; National Hurricane Center, 1999). Shelby et al. (2005) reported 555 mm of precipitation record for this period in a watershed nearby in Plymouth, NC, and this is the biggest rainfall event in 49-year historical weather record from 1951 to 1999 (Shelby et al., 2005). Accordingly, the measured total rainfall from the Carteret site in summer (June-August) and fall (September-December) was 909.3 mm. This precipitation amount was higher than long term average rainfall (1951~2000) of 782.1 mm for this period.

In 2000, no big tropical storms or hurricanes touched down in the Carteret area, but it was very wet. The measured annual rainfall at the Carteret site in 2000 was 1719 mm, 24 % higher than long-term average of 1391 mm (Table 2.12). It was drier than normal in winter and fall. Total rainfall in this period was 487 mm, 21% lower than the normal of 616 mm. However, the total rainfall of summer and fall was 1232 mm, 59% higher than the long term average of 775 mm.

Table 2.11. Monthly measured rainfall and PET data for the Carteret site compared with long term (1951~2000) data from Morehead city, NC.

Month	*Rain, mm*			*PET (mm)*			*50 Year average rainfall in Morehead City (mm)*
	1999	2000	2001	1999	2000	2001	
Jan	78	123	33	46	36	38	114
Feb	64	62	67	65	54	43	101
Mar	40	45	90	79	87	65	101
Apr	88	214	64	107	89	106	75
May	78	78	55	128	135	125	109
Jun	120	187	140	124	139	137	109
Jul	138	288	137	152	117	119	159
Aug	371	212	101	109	127	124	175
Sep	164	253	50	74	93	93	148
Oct	98	19	14	75	76	85	104
Nov	88	155	42	65	35	51	95
Dec	50	82	59	48	35	38	101
Total:	1377	1718	852	1072	1023	1024	1391

(Amatya et al, 2006)

On the contrary, 2001 was a very dry year. The measured annual precipitation was only 852 mm, just 61 % of long term average of 1391 mm (Table 2.11). Rainfall amounts of all months except June were lower than long term averages. The lowest monthly rainfall of 14 mm was observed in October 2001. The highest monthly rainfall of 371 mm was measured in August 1999. The values of annual and monthly Penman-Monteith PET (PM-PET) for grass reference calculated using the daily average weather data (Amatya et al., 2006) were also shown in Table 2.11.

2.3.1.1.2. Weather

All daily data were processed by Amatya et al. (1995, 1996, and 2000) to obtain daily and monthly averages as shown in Table 2.12. Daily rainfall, flows, net radiation, and PET for a grass reference were integrated to obtain the annual totals. Air temperature and water table depth were obtained as the total average (Amatya et al., 2006). Actual evapotranspiration (AET) in each year was calculated as a difference of annual rainfall and outflow assuming a negligible storage change (Sun et al, 2005; as cited by Amatya et al., 2006). Procedures for these parameters have been described in detail by Amatya et al. (1995, 1996, 2000, and 2006).

Table 2.12. Measured annual hydro-meteorologic parameters for Carteret weather station.

Year	*Rain mm*	*Avg Temp ℃*	*Net Rad. mm*	*PET mm*	*Flow mm*	*Runoff Coeff., %*	*WT depth, cm*	*AET mm*
1999	1377	16.7	1438	1072	614	0.45	86	763
2000	1718	15.6	1434	1023	857	0.50	79	861
2001	852	16.3	1121	1024	45	0.05	148	807
Mean *	1538	16.4	1282	970	541	0.33	95	997
Stdev *	310	0.6	142	94	286	0.13	22	143

(from Amatya et al., 2006)

* The values of mean and Stdev for each column are the averages and standard deviations from 1988 to 2004 (Amatya et al., 2006).

2.3.1.1.3. Measured flow in Open Grounds Farm and comparison with nearest site

Measured annual outflow in the watershed studied was 136 mm in 1999 (only from July 1 to Dec. 31), 254 mm in 2000, and 91 mm in 2001 (Table 2.13). Measured daily flow data were compared with those from two other studies at the Carteret site located approximately 8.9 km west to the outlet of the study watershed (Amatya et al., 2006), and the Plymouth site where is about 35 km north-west to this study watershed (Amatya et al., 2002). Rainfall and runoff ratio (R/O ratio in Table 2.13) are also presented in Table 2.13. Runoff ratio for each year was calculated as a ratio of annual outflow and rainfall for the study watershed (Amatya et al., 2006).

Compared with other two sites, the values of annual outflow and runoff ratio of OGF site were substantially small. This discrepancy resulted mainly from the physical problems of the flow meter during the study period. For example, the flow meter in the study watershed was damaged right after Hurricane Floyd, and no data were available from September 16 to November 3. For this reason, only reliable rainfall events were selected and compared with simulation results.

Table 2.13. Measured annual outflow at Open Grounds Farm and other nearby study sites, Carteret and Plymouth (mm).

	OGF			*Carteret*			*Plymouth*		
	Outflow	Rain	R/O Ratio	Outflow	Rain	R/O Ratio	Outflow	Rain	R/O Ratio
1999 (Jun. 1 ~ Dec. 31)	136*1	909*2	0.18	453	909	0.50	339	875	0.39
2000 (Jan.1 ~ Dec. 31)	254	1719*2	0.15	857	1719	0.50	494	1275	0.39
2001 (Jan. 1 ~Dec.31)	91	852*2	0.11	45	852	0.05	245	806	0.21

*1. No data from September 16 to November 3 in OGF in 1999.
*2. Because no rainfall data were available in OGF site, precipitation data from Carteret site were used for comparison.

2.3.2. Preliminary simulation results

Simulated water table depth and daily/cumulative outflows for 30 months period of baseline simulation of the study site are summarized and presented in Fig. 2.5. Daily rainfall at the nearest raingauge from the outlet of the Open Grounds Farm watershed is presented in (a), Fig. 2.5 for 30 months study period.

2.3.2.1. Simulated water table depth

The direct response of rainfall to drainage outflows is indicated by the rise in water table depths in this system as drainage rate is a function of water table depth. Because there were no measured water table depths at the study site, only simulated water table depths for study period were presented in (b), Fig.2.5. For 1999, only fall and winter (from July 1st to December 31st) data were presented. The annual average daily water table depths were 65 cm, 55 cm, and 128 cm in 1999, 2000, and 2001, respectively. The average water table depth for the study period for 2.5 years together was 85 cm.

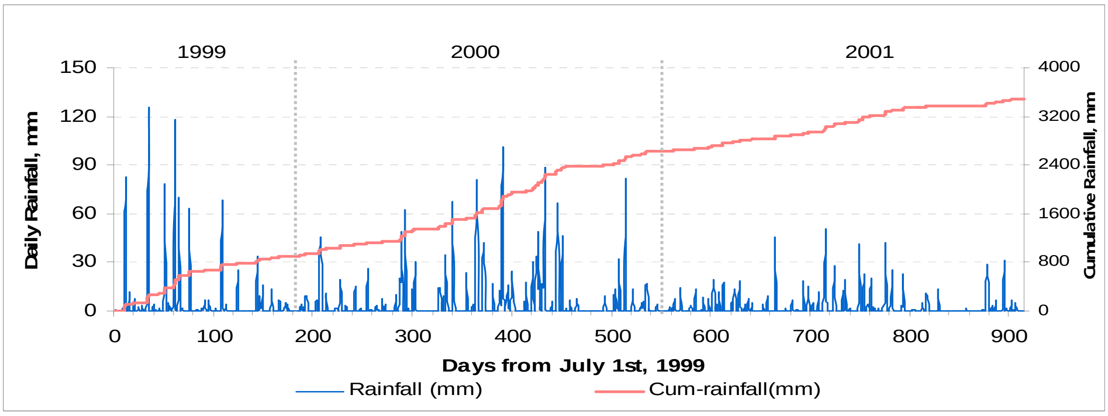
Based on the simulation results, in 1999, the water table was deep and never rose above 1.5 m before August 04, the biggest rainfall event in this area during 30 months study period. Then the simulated water table depth remained shallow and never dropped below 1.2 m. The water table rose near the ground surface when three hurricanes hit this area in September to October in 1999. In the wet year of 2000, simulated water table was so shallow that it never dropped under 1.4 m all around the year. It was dry in 2001 and water table depths were under 1.5 m for 249 days. Especially in fall and winter, the water table never came up above 1.5 m. As reported in the previous studies (Amatya et al., 2006; Amatya et al., 1996), the simulated deep water table in 2001 was most likely caused by lower rainfall and higher ET than 50 years average values observed in nearby weather station in Morehead City, NC.

2.3.2.2. Daily outflows

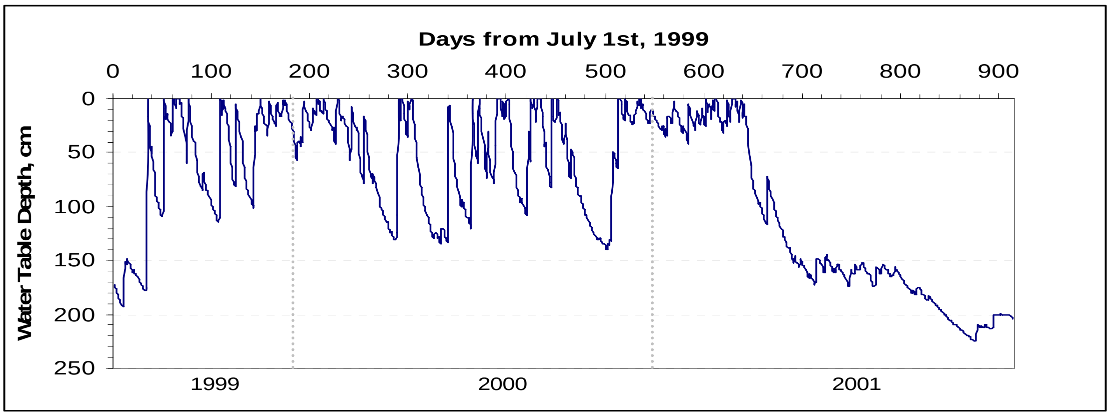
Overall, the daily outflow data simulated by the DRAINWAT model were not in good agreement with a limited available measured outflow data for the study period ((c), Fig. 2.5). Total simulated cumulative outflows were over 50% higher than observed cumulative outflows in 1999, and 60% higher in 2000. On the contrary, simulated cumulative outflow was 35% lower than the observed in 2001. This means that DRAINWAT predictions were not reliable under very dry conditions, and these results were consistent with other study results (Amatya et al., 1997, 2004, 2006). This also indicates that the model does not seem to have consistent systematic errors as there were over and under predictions. These discrepancies between measured and simulated data were larger than other studies using DRAINMOD–based watershed models.

These discrepancies occurred due to the following reasons.

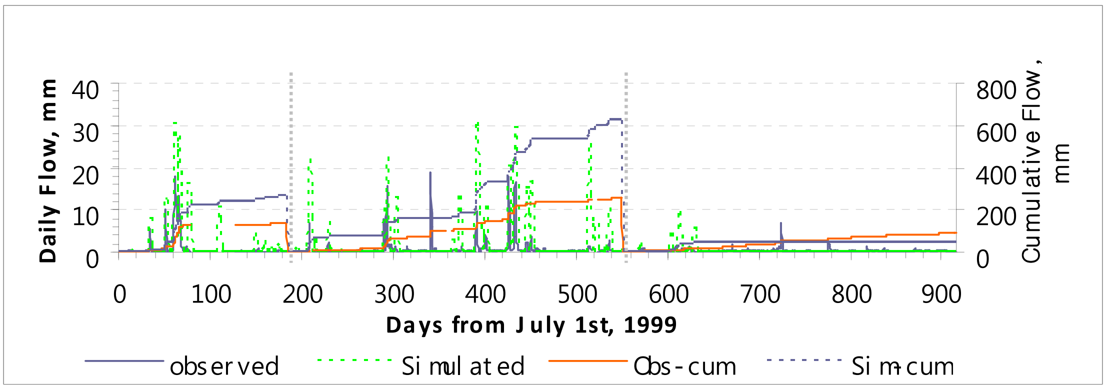
1) Potential errors in several days of measured data limited the available data for comparison. 2) Potential errors in spatial variability of precipitation as rainfall data were not available at the study site. 3) There were no actual soil hydraulic properties measured at the study site. All simulations were based on the literature data. 4) PET data were also measured in the Carteret site, 9 km from the study site. However, based on wide testing of DRAINWAT and other DRAINMOD-based watershed scale models at similar agricultural, forested and mixed land use watershed in the NC Coastal Plain as shown below in the section of 2.3.2.5, we argue that the model result may be reasonable.



(a)



(b)



(c)

Fig. 2.5. Preliminary simulation results. (a) Daily rainfall of study watershed from July 1999 to December 2001. (b) Simulated water table depth in study watershed. (c) Comparison of daily simulated outflow with observed outflow in the outlet of study watershed.

2.3.2.3. Storm event analysis

Six "so called" reliable measured outflow data were used for storm event analysis. Results shown in Fig. 2.6 present six plots of measured and simulated event hydrograph. In the events 3 and 6, underestimation of DRAINWAT was observed. Underprediction of events 3 and 6 can be explained by dry soil conditions. As mentioned above DRAINWAT simulation is not predictable in extremely dry conditions.

DRAINWAT overestimated daily outflows in events 1 and 4. Simulated outflows were delayed 1~ 2 days in events 2 and 5. We concluded that simulation results were not satisfactory and there were no unique trends between observed and simulated values.

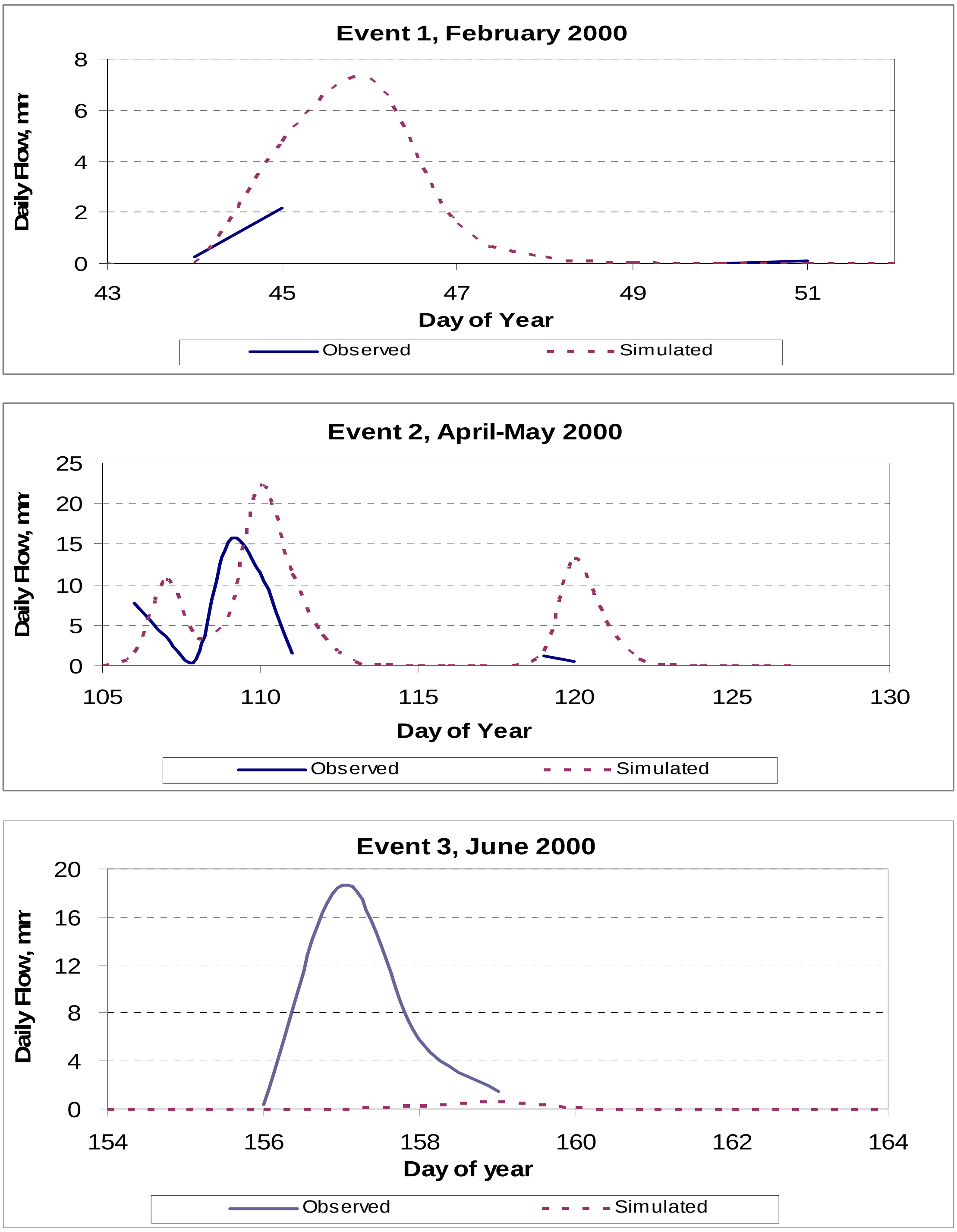
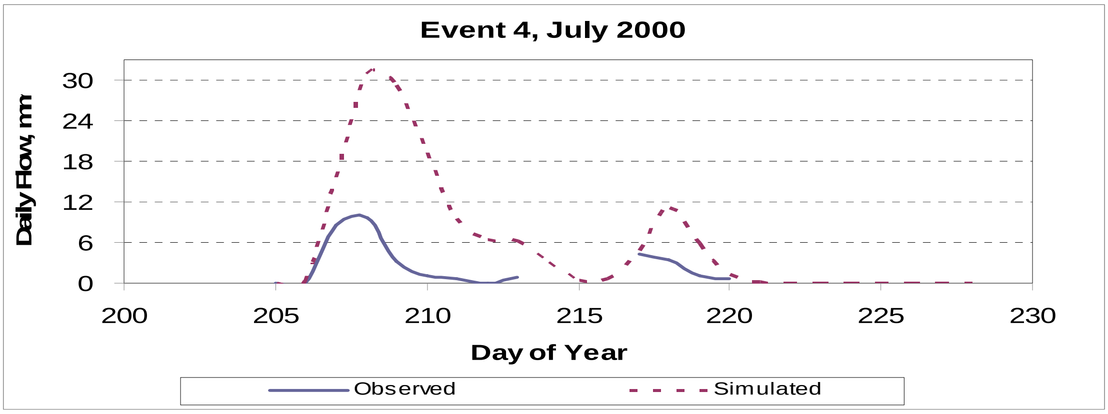
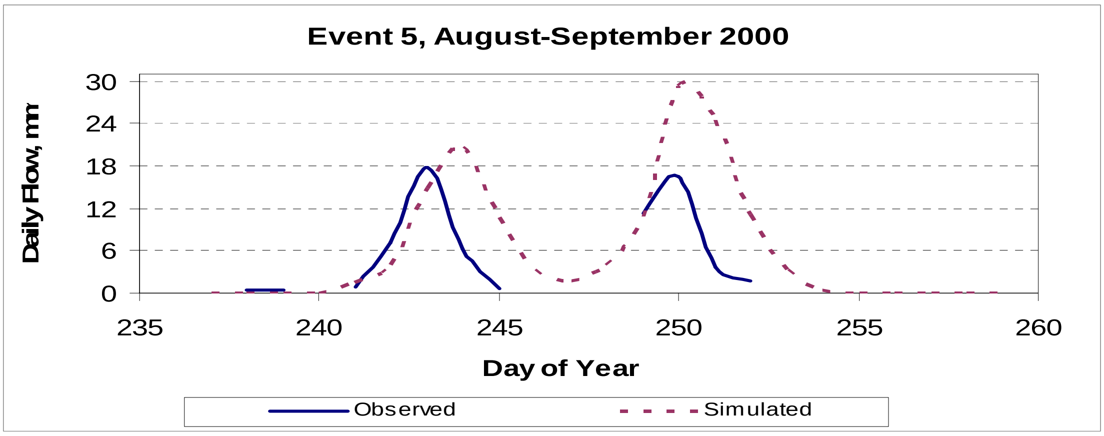


Fig. 2.6(a). Storm event analysis in events 1, 2, and 3.





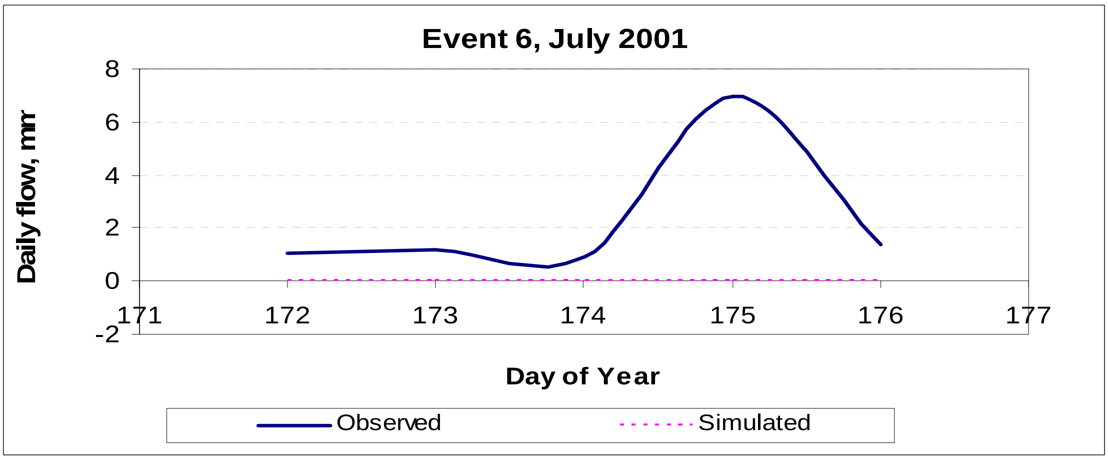


Fig. 2.6(b). Storm event analysis in events 4, 5, and 6.

These results were presented and summarized in Table 2.14 with statistical analysis results. The r^2 value varied from infinity of event 6 to 0.99 of event 1. Based on r^2 values, simulation results of event 1 was excellent, but all others were not satisfactory.

Highest r^2 value of event 1 resulted from a small sample size of 4. Because all of the simulated outflows were 0 during selected period in event 6, r^2 value was infinity. The E values range from -15.95 of event 4 to 0.20 of event 3. All of these results were considered to be unsatisfactory. Comparison indicates that simulation results did not correspond with observed data. Because the precipitation data for the DRAINWAT simulation were from Carteret sites which are located in 8.9 km from the watershed outlet, spatial variation of rainfall may be the main reason. Simulated outflows did not correspond with observed data.

Table 2.14. Summary of statistical analysis for estimating simulation results in selected period.

Events	*Average Daily Outflow, mm*		*r^2*	*E*
	Observed	*Predicted*		
1	0.64	1.19	0.99	-1.13
2	5.27	8.76	0.04	-1.56
3	6.58	0.20	0.20	0.20
4	3.31	13.37	0.33	-15.95
5	6.09	11.54	0.33	-1.00
6	2.55	0.00	Infinity	-0.31

2.3.2.5. Comparison with other studies

As presented above, the preliminary simulation results of DRAINWAT in this study were unsatisfactory. However, other studies using DRAINMOD-based watershed models in eastern North Carolina have shown great predictability.

Amatya et al. (1997) tested DRAINWAT with 5 years data from 1988 to 1992 collected on a 340 ha watershed near Beaufort in eastern North Carolina. They compared observed daily and monthly outflows with predicted data using three calculation methods for potential evapotranspiration. In their study, Penman-Monteith methods evaluated with two different stomatal conductance submodels were used for predicting ET. The one was GS_HR function developed with weather data and porometer readings (Amatya et al., 1996). The other was the Teskey method based on night time temperature and net radiation (McCarthy et al., 1992; Amatya, 1993). The average of r^2 and E value of those two submodels for monthly simulations were 0.79 and 0.82, respectively.

Fernandez et al. (2002) tested DRAINMOD-DUFLOW, another watershed scale version of DRAINMOD, and showed the temporal trend and magnitude of daily and monthly flows at a forested watershed at Parker Tract in Plymouth, eastern North Carolina. The Nash-Sutcliffe coefficient (E) value and r^2 value between observed and predicted monthly flow data were 0.84 and 0.92, respectively, which are in the range of good model performance based on Moriasi et al. (2007). The model underpredicted the monthly flows by 2.1 mm (underprediction of 16%).

Two years later, Amatya et al. (2004) also presented the measured and DRAINWAT-simulated daily and cumulative drainage outflows at the same forested watershed in eastern North Carolina in 2004. Hydrologic simulation results of their study were also in close agreement with the measured data. Total predicted cumulative outflow at the end of the five year period was only 6.5% lower than the total measured outflow. On a yearly basis, the average absolute daily deviation in daily flows varied from 0.14 to 0.41, and E values ranged from 0.71 to 0.84. Predicted annual outflows were from 15% underprediction to 5% overprediction. These results again were considered good to satisfactory based on Moriasi et al. (2007).

DRAINMOD-DUFLOW and DRAINMOD-GIS (Fernandez et al., 2003) were tested and applied to predict NO_3-N load delivered to the outlets of three coastal plain watersheds in eastern North Carolina: the 8100 ha Kendricks Creek watershed near Plymouth, the 11600 ha Chicod Creek watershed near Greenville, and the 8300 ha Upper Broad Creek watershed near New Bern (Chescheir et al., 2004). They suggested a lot of factors can affect the cumulative nitrate load at the outlet of watersheds, and classified these factors into two: 1) factors that impact the loads at the field edge such as land use, soil type, and management practice. 2) factors affecting the fate of nitrate-N as it moves through the stream network such as stream length, stream slope, channel dimensions, and decay rates.

Based on the measured data from the same watershed as Amatya et al. (2004), Fernandez et al. (2005) again evaluated the performance of two DRAINMOD-based watershed scale models, DRAINMOD-DUFLOW and DRAINMOD-W. Predictions of both of those two models were in close agreement with each other and with the observed monthly flows. The cumulative predicted outflows using the two models for the study period (1996-2000) was within ± 1% of the measured outflow. The E values for monthly comparison were over 0.8. E values for daily comparisons were less than this, but still within the acceptable range.

Fernandez et al. (2006) tested DRAINMOD-GIS, lumped parameter hydrology and water quality model that includes an uncertainty analysis component (Fernandez et al., 2003). Their prediction errors were less than 10% and ranged from under-prediction of 9% (1997) to over-prediction of 4% (1999) on an annual basis. Prediction errors for the cumulative outflows for both of their calibration (1996-1998) and validation (1999-2001) periods were within 1%. The statistics are generally acceptable for both the daily and monthly data. The modified E values for considering extreme values were larger than 0.5, within satisfactory range. The r^2 value of monthly outflow was 0.93 with increased scatter for higher flows.

Fernandez et al. (2007) reported that the hydrologic simulation s were within 5% of the measured data in their another application study using DRAINMOD-GIS, DRAINMOD-based lumped parameter watershed model, on the Chicod Creek watershed with the size of 11100 ha in eastern North Carolina that is not intensively instrumented or documented.

DRAINMOD-based watershed models were also used to evaluate potential effects of climate change on the hydrology of a 3000 ha managed pine forest in coastal North Carolina. DRAINMOD-based models were also applied to predict the hydrology and water quality at the watersheds in other areas. Bisol (2006) tested DRAINWAT in Northern Italy, and Amoah (2008) applied DRAINWAT in South Carolina with some success.

Based on these results from various study sites in the lower coastal plain region of North Carolina, the ability of DRAINMOD-based watershed models to predict drainage outflows were reasonable, excellent to satisfactory! However, in this study, due to problems with the flow meter at the outlet of the watershed as mentioned above, and possibly due to lack of rainfall and weather data in the study site, the simulation results were either unsatisfactory or there were only limited data for satisfactory evaluation of the model. For these reasons, some of the few most reliable measured flow data were selected based on field records and compared for storm event analysis. As we discussed in the previous section of 2.3.2.4, selected reliable data were also not consistent with simulated outflows for the same reasons stated above. These results of hydrological simulation were not adequate and satisfactory to be applied to nitrogen loading estimation model described above in section 2.2.5.2. Therefore, remaining part of this study focused on sensitivity analysis of some key parameters as shown below.

2.3.3. Sensitivity analyses

Four parameters including Manning's coefficient, depressional storage, and channel bed slope were used for sensitivity analysis of the simulated drainage outflow. Furthermore, the model sensitivity of three different methods of potential evapotranspiration on drainage outflows was analyzed.

2.3.3.1. Manning's roughness coefficient

The Manning's roughness coefficient (n) for the base simulation in this study was 0.04. Cumulative daily outflows were compared among three n values including 0.03, 0.05, and 0.06 in Figure 2.7, and the simulation results were summarized in Table 2.15.

Table 2.15. Comparison of cumulative outflows in different n values.

	Manning's roughness coefficient			
Year	*n=0.03 outflow, mm*	*n=0.04 outflow, mm*	*n=0.05 outflow, mm*	*n=0.06 outflow, mm*
1999	265.4	264.2	263.9	263.6
2000	624.0	624.8	625.6	625.5
2001	313.9	314.1	313.0	314.5

As n value increases, simulated cumulative outflow values were slightly changed (<1%) on the annual basis. No substantial differences were observed between those values and it was concluded that cumulative outflow was very insensitive to Manning's roughness coefficient. Konyha and Skaggs (1992) observed similar results. They found that total outflow volume was changed only from 226.0 mm to 224.3 mm when they simulated FLDNSTRM with +150% and -150% change of Manning's n.

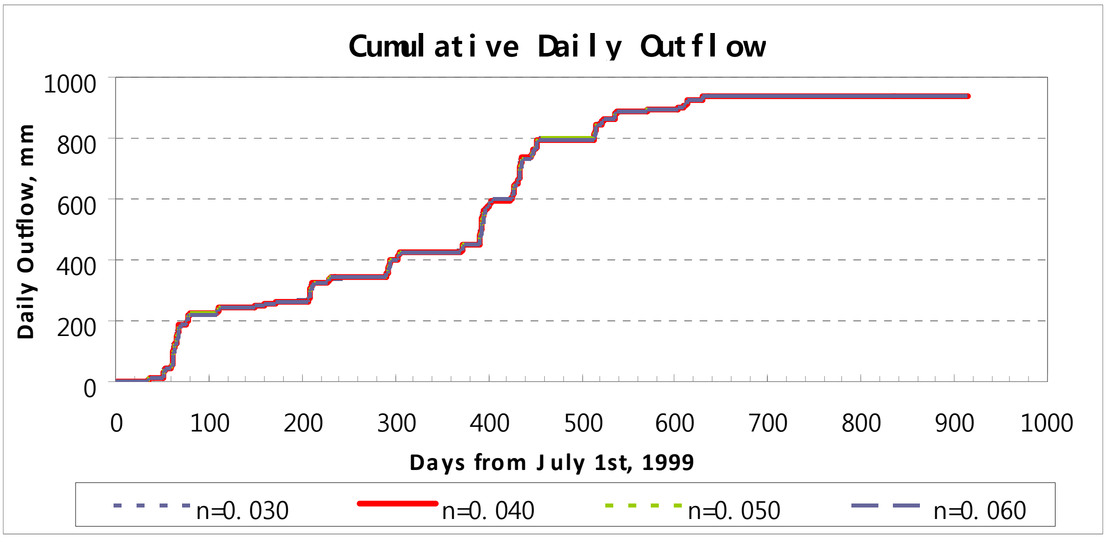
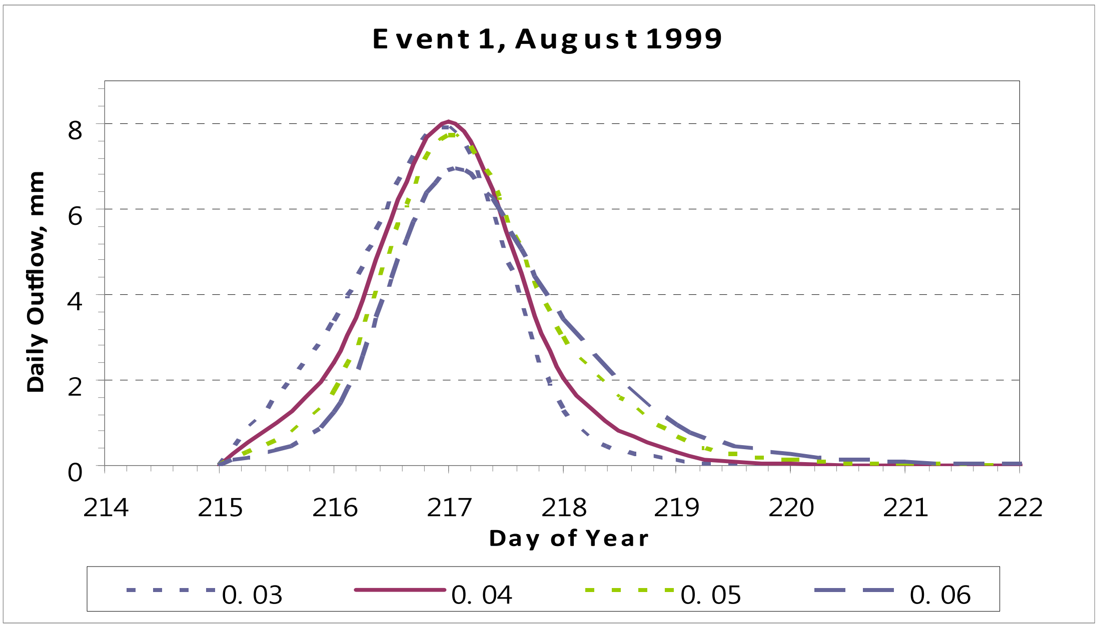


Fig.2.7. Comparison of cumulative outflow using different Manning's n values.
It was very insensitive! We cannot see any differences on cumulative basis! Similar results were found other two parameters except for the method of PET calculations.

On the daily basis, variations of simulated outflows for specific rainfall events were visible as n value increases. Simulated daily outflow decreased gradually as n increased in moderate (event 1 of day 215 in 1999 and event 4 of day 52 in 2001) and big (event 3 of day 206 in 2000) storm event (Fig 2.8). Interestingly, in small event (event 2), abrupt decrease of flow was found when n increase from 0.04 to 0.05, indicating even a small increase may dramatically reduce (5-fold, approximately) the flow rates for low flows even though the outflow change was still relatively minor.



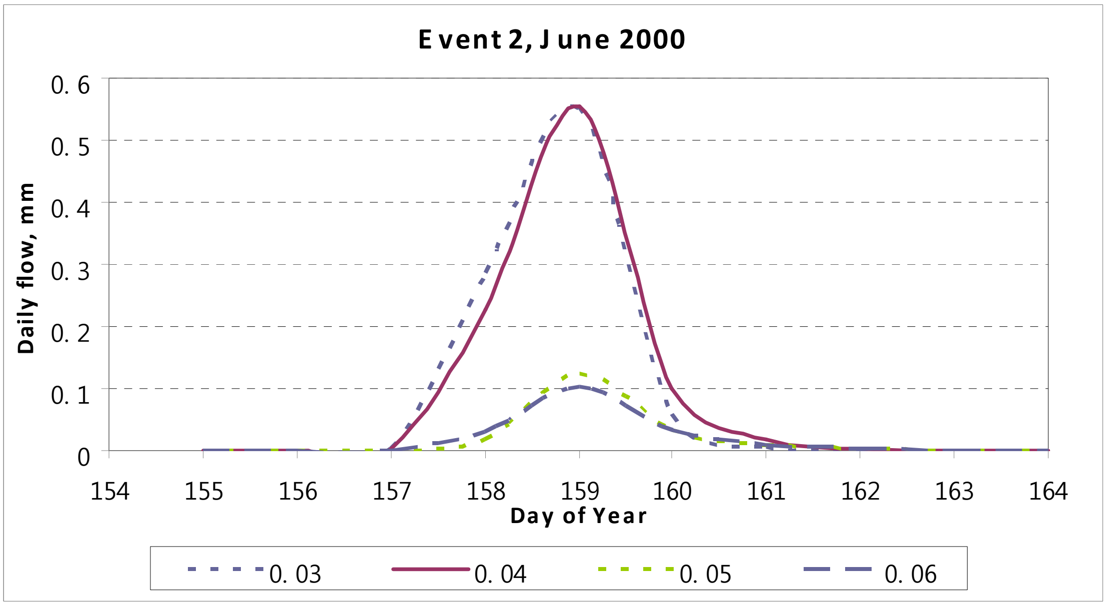
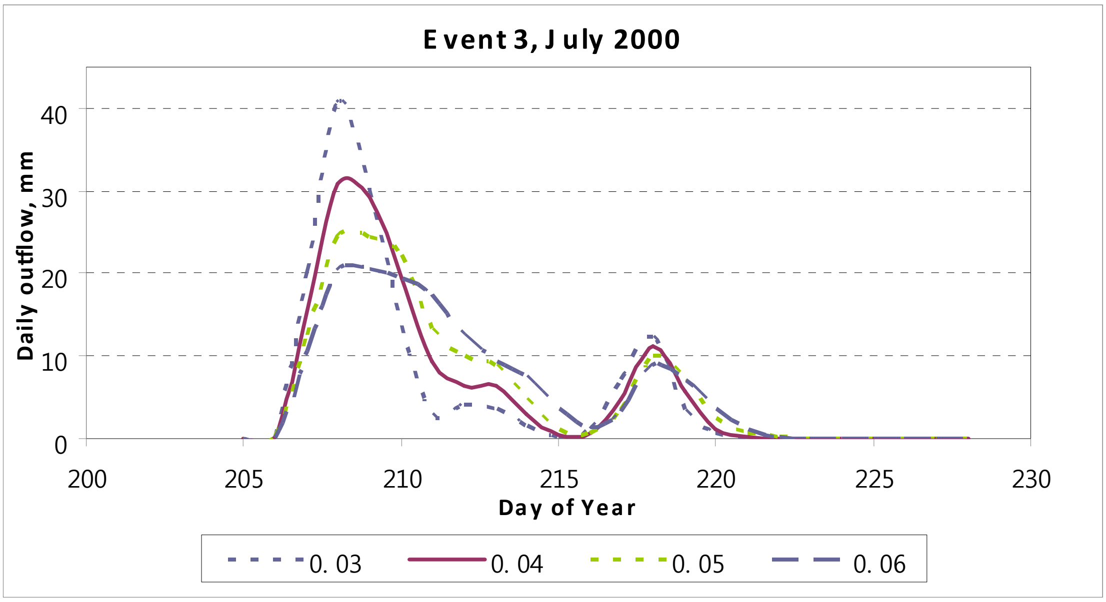


Fig.2.8(a). Sensitivity analyses for Manning's roughness (n) in events 1, and 2.



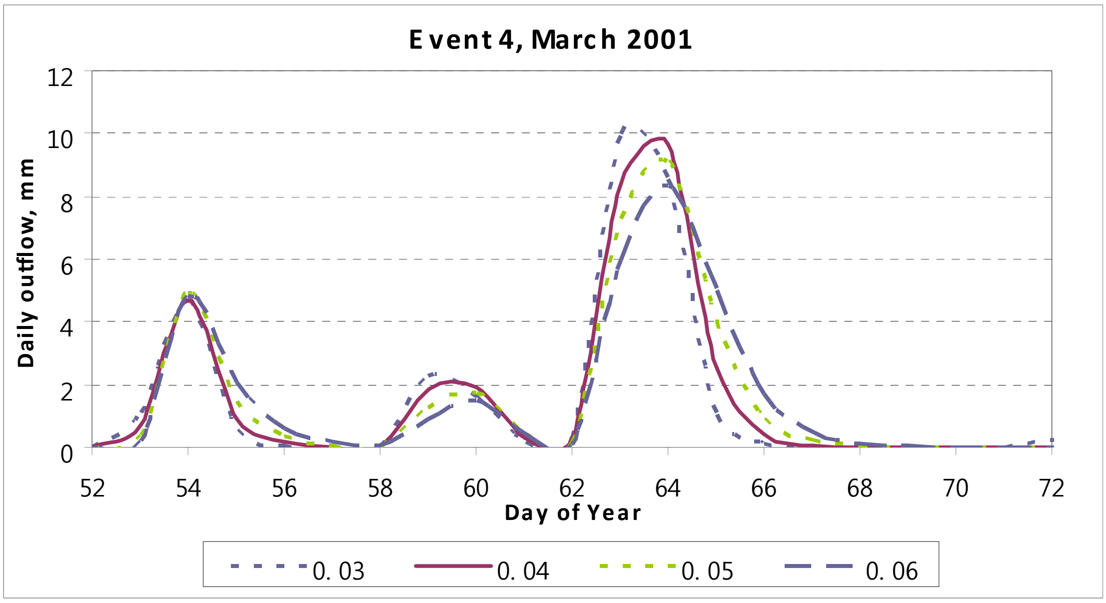


Fig.2.8(b). Sensitivity analyses for Manning's roughness (n) in events 3, and 4.

The change in peak flow rate, time to peak, and the volume of event outflows due to the increase of Manning's roughness (n) are given in Table 2.16.

Peak flow rate decreased for the events 3 and 4 as expected with increase in n, but there were no distinctive trends for the events 1 and 2. For the event 2, abrupt decrease in peak outflow rate was observed when n increased from 0.04 to 0.05 as shown in Table

2.16. The same results were already found for the change in daily outflows (Fig. 2.8), and also found for the change in volume of event outflow. We proposed that there must be some threshold where infiltration is enough to prevent runoff.

As the Manning's roughness increases, outflow rate generally decreases due to decrease in velocity (equation 2.7 and 2.8). However, the volume of event outflow increased as n increased only from 0.03 to 0.05 for event 1, and then decreased after that. The volume of event outflows even increased in the events 3 and 4.

The size of peak flow rate was similar for the events 1 and 4, but outflow volume of event 4 was substantially greater than that for event 1 in every n value. In the case of event 1, it was a single rainfall event, and there were no previous, large rainfall events to saturate the soils for 20 days. Simulated water table depth on August 3 was 177 cm, and initial outflow rate was 0 mm. large amount of rainfall was needed to fill up the unsaturated soil before the flow started. In the case of event 4, however, two small rainfall events occurred right before this event (Fig. 2.8(b)), and consequently the amount of outflow on the first day was not zero (0.22 mm, not shown). The simulated water table depth on the first day of the event 4 was already at the surface.

Time to peak remained the same without change i.e. two days for events 1 and 3, three days for event 2, except for event 4 in which case it increased from 2 to 3 with increase in n, as expected. Because these simulations were based on a daily time step, no big changes of time to peak were observed as the Manning's roughness increased. In general, time to peak is expected to increase with the increase in channel roughness due to decrease in velocity. Differences possibly would have been noticed for simulation results with an hourly time steps.

Even though small variations of peak flow rate, time to peak, and the volume of event outflows due to the increase of Manning's roughness were found, the change of those parameters were generally not substantial. Similar results were found from other studies. For example, Konyha and Skaggs (1992) reported that channel roughness was the least sensitive parameters among the five that includes time of concentration, channel bottom slope, hydraulic conductivity, drainage spacing, and channel roughness.

Table 2.16. Effects of Manning's roughness in predicted specific storm event hydrographs.

Storm event date	*Manning's roughness*	*Peak flow rate (mm/day)*	*Vol. of event outflow, mm*	*Time to peak (days)*
1. Aug. 03~14, 1999	0.030	7.9	12.7	2
	0.040	8.1	12.9	2
	0.050	7.7	13.3	2
	0.060	6.9	12.9	2
2. Jun. 01~11, 2000	0.030	0.6	0.9	3
	0.040	0.6	0.9	3
	0.050	0.1	0.2	3
	0.060	0.1	0.2	3
3. Jul. 23~ Aug. 02, 2000	0.030	40.7	117.9	2
	0.040	30.8	121.1	2
	0.050	24.5	121.8	2
	0.060	20.6	121.4	2
4. Mar. 03~14, 2001	0.030	10.0	20.1	2
	0.040	9.7	21.3	3
	0.050	9.1	21.7	3
	0.060	8.4	21.7	3

2.3.3.2. Depressional storage

The value of maximum depressional storage used in the baseline simulation was 1.0 cm. Predicted cumulative daily outflows were compared for six different values of depressional storage of 0.25, 0.50, 0.75, 1.00, 1.25, and 1.50 and the results are presented in Fig. 2.9. There were no significant differences in cumulative outflows obtained by using those values (α = 0.05) indicating that cumulative outflow was also not sensitive to depressional storage.

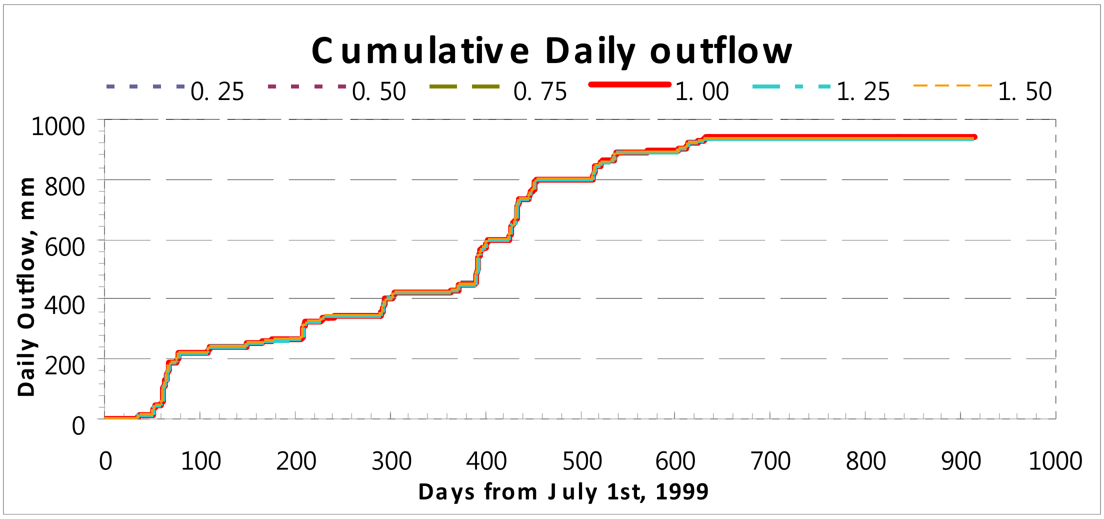


Fig. 2.9. Cumulative outflow using different maximum depressional storage (STMAX) values.

No substantial variations in cumulative outflows were found (<1%) due to change in maximum surface depressional storage (STMAX) values in each year of study period (Fig. 2.9, and Table 2.17). Based on other studies conducted nearby (Shelby et al., 2005; Amatya et al., 2005), July-December in 1999 included an extremely wet period followed by a very dry period of July; mid-August, 2000 was very wet; and 2001 was very dry compared with the 50-year average in Table 2.12. Therefore, unexpected small effects of STMAX on annual outflow can be explained two ways as Skaggs et al. (1991) suggested. Surface depressions remained nearly full during most of the year and did not substantially effect outflow during a very wet period such as July, 1999 – December, 2000. On the contrary, in a dry period like 2001, there was little or no runoff and almost nothing to be affected. Skaggs et al. (1991) conducted a simulation study to determine the effects of natural factors including STMAX on pocosin hydrology in a natural pocosin near Wilmington, North Carolina. They reported that annual average of simulated outflow (expressed with "average annual runoff") increased from 365 mm to 384 mm as the STMAX storage decreased from 2.5 cm to 0.5 cm. They also mentioned that year-to-year variation in annual outflow due to STMAX change was much greater than the effects of all other factors used in their study.

Table 2.17. Effects of maximum surface depressional storage on simulated cumulative daily outflows in each year of the study period. In 1999, only simulation results of six months (July – December) were presented.

Year	*Predicted total outflows (mm) for maximum surface depressional storage (STMAX), cm*					
	0.25	0.5	0.75	1	1.25	1.5
1999	262.6	262.7	263.6	264.2	262.0	263.1
2000	625.5	625.2	623.9	624.8	625.5	623.9
2001	312.9	312.9	313.8	314.1	311.4	313.0

Results in Fig. 2.10 present the patterns of daily outflows based on different STMAX values for four storm events. Rainfall and the fluctuations in predicted water table depths of the corresponding events are also presented. In event 1, daily outflow decreased as the depressional storage increased as expected. This event was dependent on one big rainfall event of 131.0 mm at August 3 ~ 4, 1999. The soil was very dry and the simulated water table depth was 175 cm at the first day of event. On the contrary, the water table was at the ground surface due to previous rainfall events when event 4 began at the day of 62 (March 3, 2001) with the rainfall of 28.2 mm at March 3~5, 2001. Therefore, even though there was substantial difference between the related rainfall amounts of events 1 and 4, the peak flow rates of events 1 and 4 were similar to each other due to initial soil moisture condition.

The volume of event outflow in event 2 was very small, but related rainfall of 73.9 mm at June 4~7, 2000 was more than two times greater than that of event 4. This result can be explained with dry soil condition at the initial day of event. The simulated water table depth of June 4, 2000 was 133 cm. Initial water table depth of event 3 was 43 cm, and related rainfall amount was 160.3 mm at July 23~26, 2000. Water table depth was not substantially sensitive to the change in STMAX.

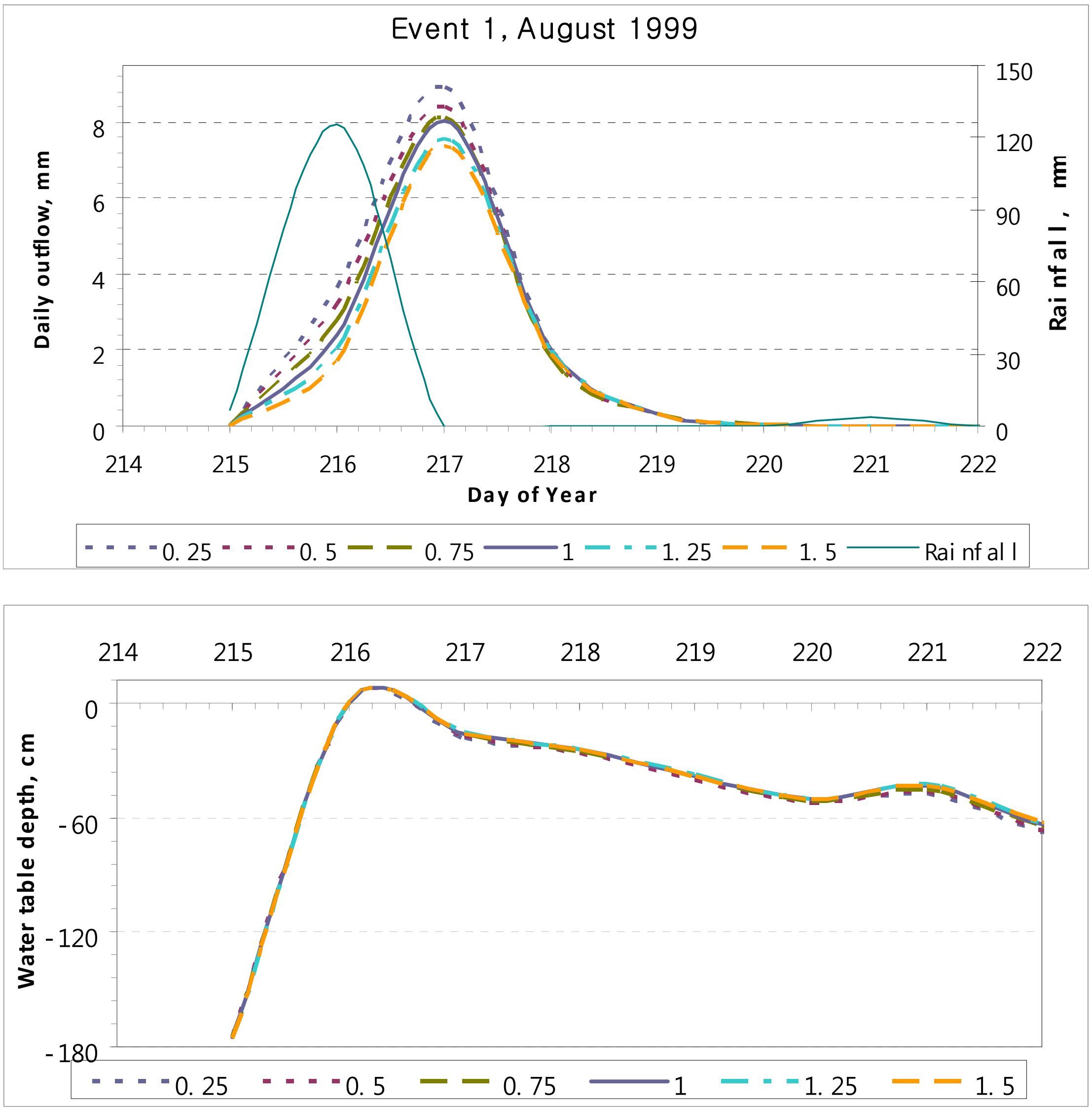
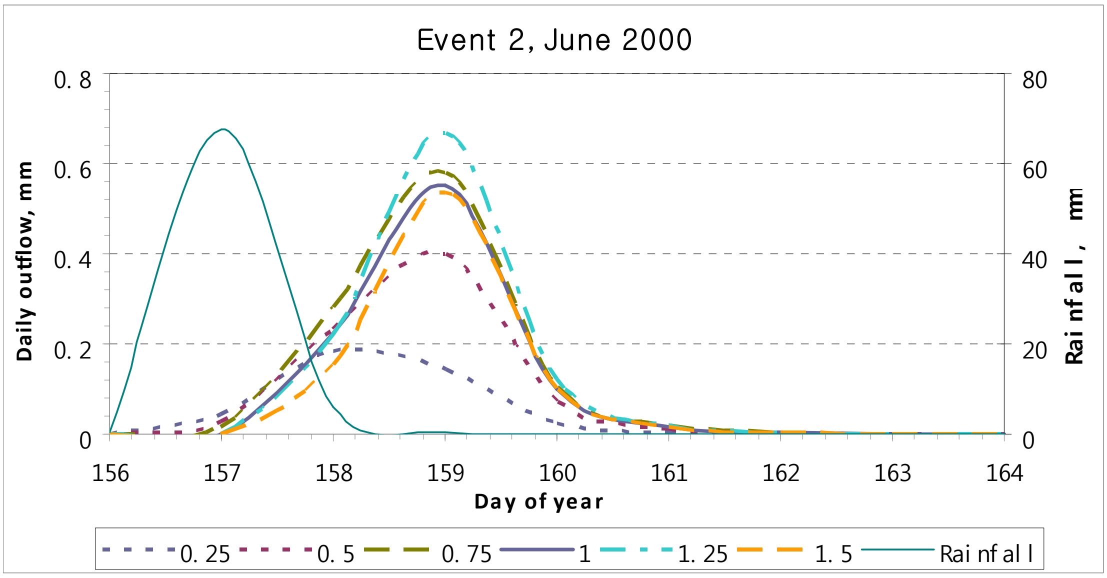


Fig. 2.10(a). Sensitivity of daily flows and simulated water table depth to the maximum depressional storage in event 1.



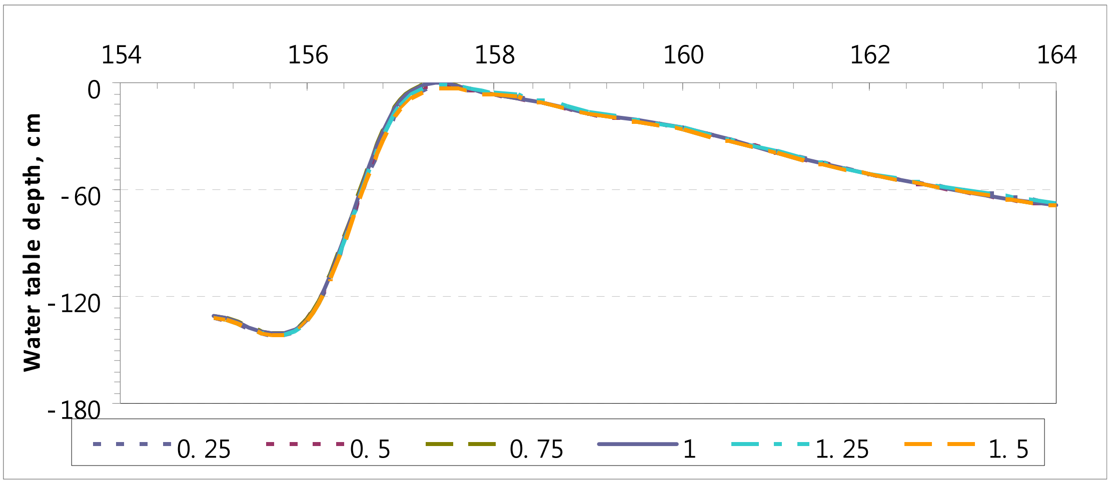
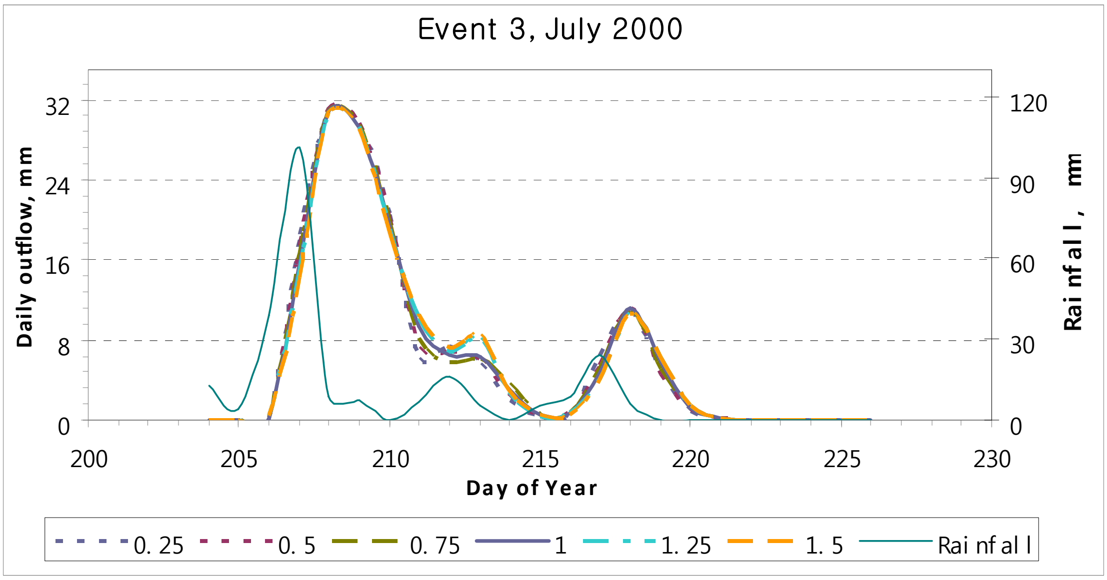


Fig. 2.10(b). Sensitivity of daily flows and simulated water table depth to the maximum depressional storage in event 2.



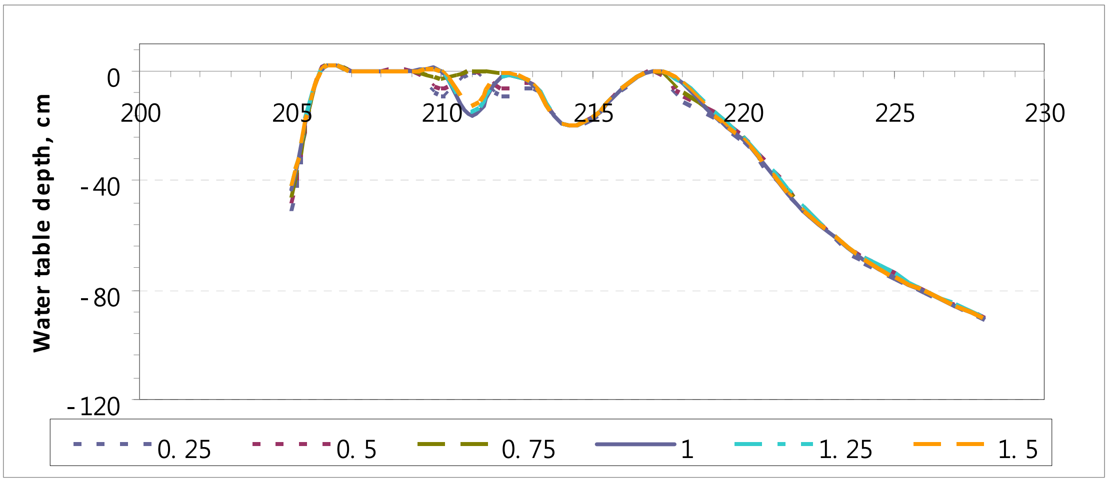
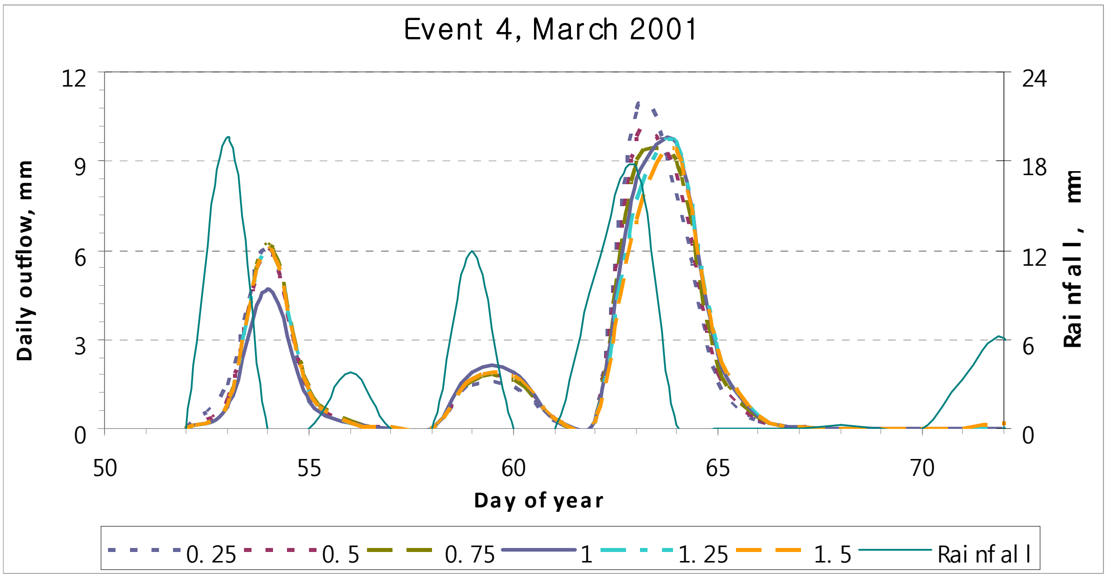


Fig. 2.10(c). Sensitivity of daily flows and simulated water table depth to the maximum depressional storage in event 3.



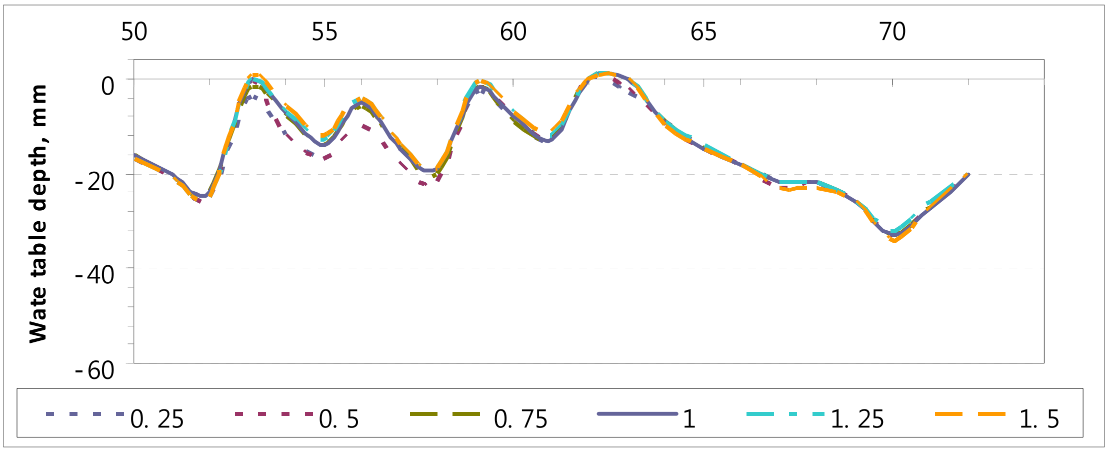


Fig. 2.10(d). Sensitivity of daily flows and simulated water table depth to the maximum depressional storage in event 4.

The sensitivity analysis results of event outflow characteristics to the STMAX were summarized in Table 2.18. In event 1 peak flow rate decreased from 8.9 mm/day to 7.4 mm/day as STMAX increase from 0.25 cm to 1.5 cm as expected. There was a slight decrease from 31.1 mm/day to 30.7 mm/day, but it was not substantial. However, peak flow rate unexpectedly increased from 0.19 mm/day to 0.67 mm/day as STMAX increase from 0.25 cm to 1.25 cm, and then slightly decreased from 0.67 mm/day to 0.54 mm/day as STMAX increase from 1.25 cm to 1.50 cm. We found that in small rainfall events, the flow rate can unexpectedly increase as the increase of STMAX even though the amount of flow change is small. No unique trends were found in the event 4. Time to peak was predicted to be two days for events 1 and 3, two - three days for event 2 and 4. In event 1, the volume of event slightly decreased as STMAX increased, but no unique patterns were found in event 2, 3, and 4.

We concluded that watershed outflow was not sensitive to STMAX over the range from 0.25 cm to 1.5 cm. Jurek (2001) also found similar results. He used 3 values of STMAX including 3 cm, 7.5 cm, and 10 cm based on field conditions for baseline simulation for his study. Then he tested the sensitivity of peak flow rates and outflow with changing these values from -50% to +25%. His baseline simulation results of peak flow rate, and total annual outflow were 11.1 mm/day and 296.4 mm, respectively. Peak flow rate did not change at all with different STMAX values. Slight reduction (3%) in annual outflow was observed with a 50% reduction in STMAX, but he concluded that this was not substantial (Jurek, 2001).

Table 2.18. Effects of maximum surface depressional storage (STMAX) on predicted storm event hydrograph characteristics.

Dates	STMAX (cm)	Peak flow rate (mm/day)	Time to peak (days)	Volume of event outflow (mm)
	0.25	8.9	2	14.9
	0.5	8.4	2	13.9
Event 1	0.75	8.1	2	13.1
Aug. 04-14, 1999	1	8.1	2	12.9
	1.25	7.6	2	11.9
	1.5	7.4	2	11.4
	0.25	0.19	2	0.41
	0.5	0.40	3	0.751
Event 2	0.75	0.58	3	1.007
Jun 01-11, 2000	1	0.55	3	0.902
	1.25	0.67	3	1.037
	1.5	0.54	3	0.811
	0.25	31.1	2	121.2
	0.5	31.0	2	121.4
Event 3	0.75	31.0	2	121.6
Jul.23~Aug.02, 2000	1	31.0	2	121.1
	1.25	30.8	2	122.5
	1.5	30.7	2	122.8
	0.25	10.7	2	20.7
	0.5	9.8	2	20.7
Event 4	0.75	9.0	2	20.7
March 03~14, 2001	1	9.7	3	21.3
	1.25	9.6	3	20.6
	1.5	9.4	3	20.0

2.3.4.3. Channel bed slope

The changes in daily cumulative flow due to the variation in channel bed slope are summarized and presented in Table 2.19 and Fig 2.11. There were no unique trends or substantial differences among different channel slope. Konyha and Skaggs (1992) reported that channel bedslope was more sensitive than Manning's roughness, but the simulated outflow during the 60-day period used in their simulation was not significantly affected by increasing channel bed slope.

Table 2.19. Effects of changing channel bed slope on simulated cumulative outflows for the 1999 – 2001 study period. Only simulation results of six months (July – December) were presented in 1999.

	Channel bed slope			
Year	*s=0.00005 outflow, mm*	*s=0.0001 outflow, mm*	*s=0.0002 outflow, mm*	*s=0.0003 outflow, mm*
1999	266.7	264.6	265.5	265.9
2000	624.5	624.3	622.9	622.1
2001	317.3	314.8	315.7	316.8

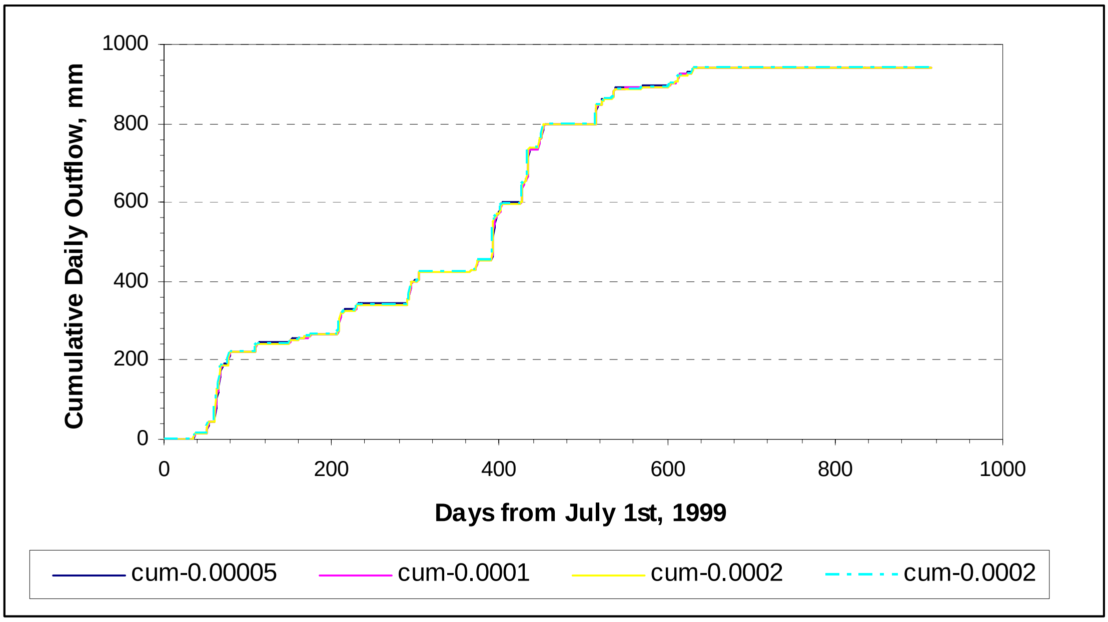
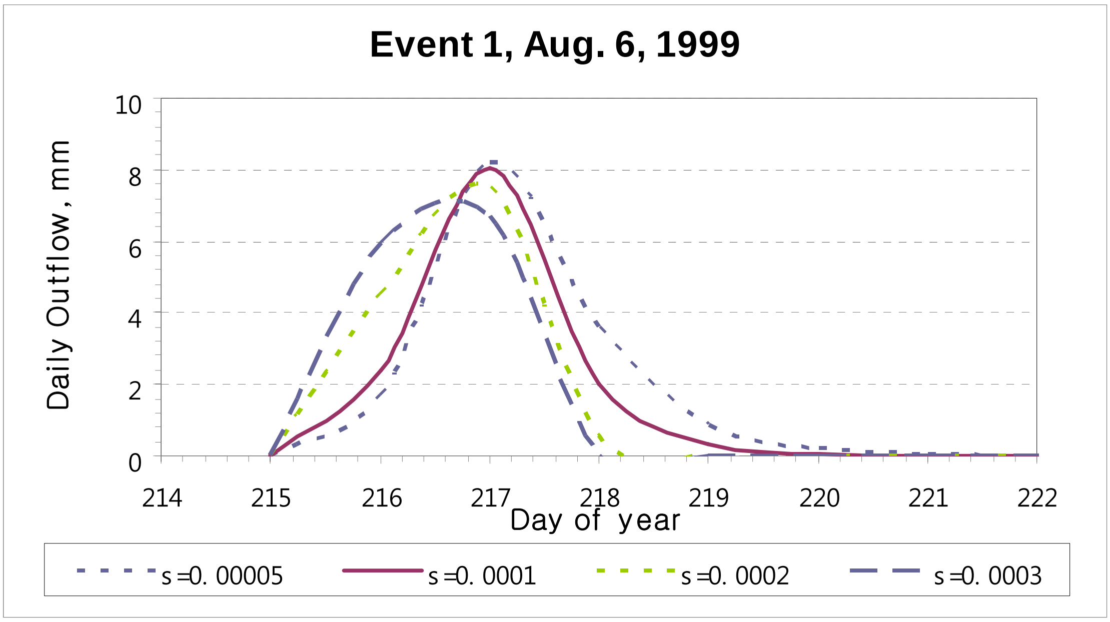


Fig. 2.11. Sensitivity of cumulative outflow to different channel bed slope values.

In the case of event outflows (Fig. 2.12), as the channel bed slope increased, peak flow rate increased, but time to peak decreased in all events as expected except event 1. Relatively small peak flow rates with steep slopes in events can be caused by the dry soil conditions right before the rainfall events.



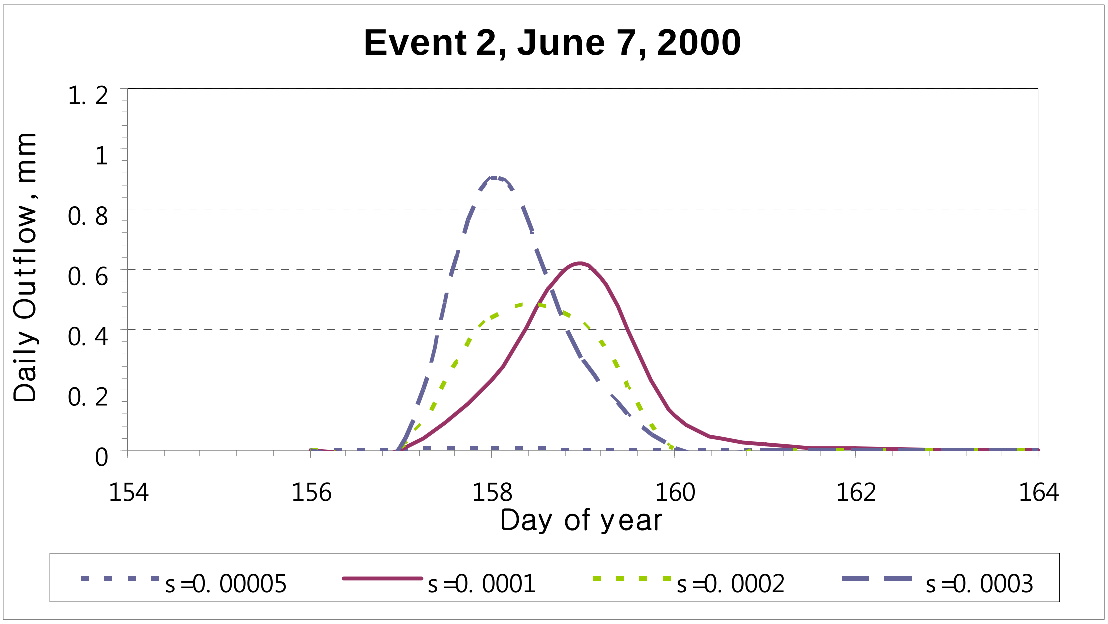
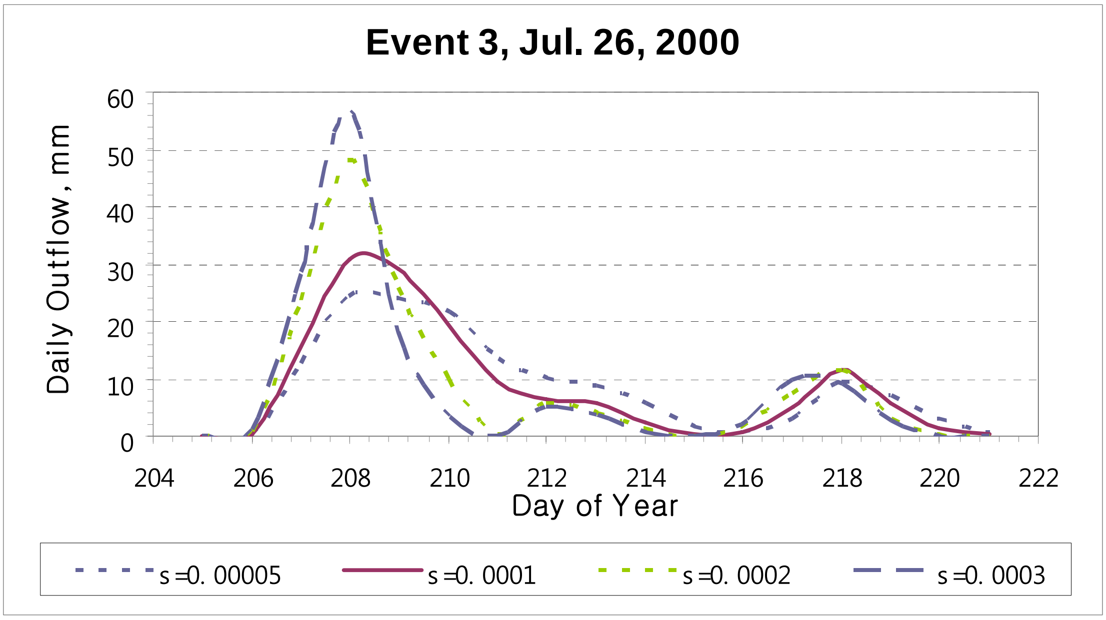


Fig. 2.12(a). Sensitivity of outflow rates to different values of channel bed slope in events 1, and 2.



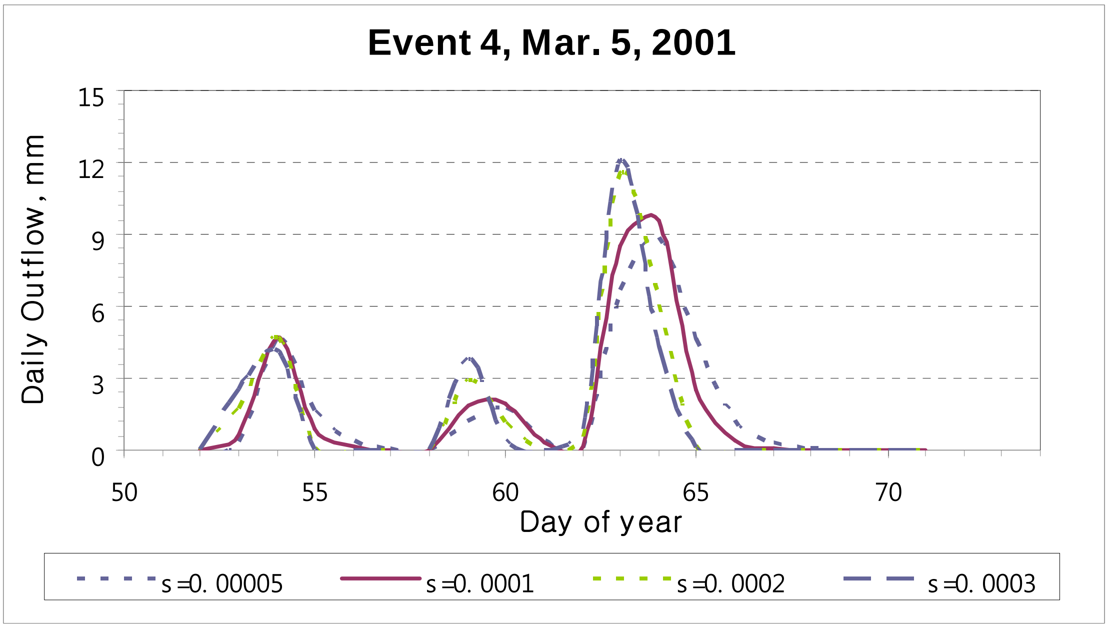


Fig. 2.12(b). Sensitivity of outflow rates to different values of channel bed slope in events 3, and 4.

The peak flow rate, time to peak, volume of event of 4 rainfall events are summarized in Table 2.20. Comparing the peak flow rate of event 1 and event 4, the rate was respectively 8.2 mm/day and 8.9 mm/day when the slope was 0.00005. As the slope increased, however, peak flow rate decreased in event 1, and increased in event 4. This result may be due to initial soil moisture condition. In the case of event 1, the water table was deep at the first day of event, and a large amount of rainfall was used to fill up the dry soil. In the event 4, the soil was already saturated when rainfall began at the first day of event. Only in big storm events (event 3), we could see clearly that peak flow rates significantly increased from 24.3 mm/day to 56.7 mm/day as channel slope increased from 0.00005 to 0.0003.

There were no changes of daily time to peak in every event as expected. No unique trends were observed in the volume of events as slope increased. However, in event 2, substantially low peak flow rates and volume of event were observed when slope was very low, (slope = 0.00005). Similar results were observed by Konyha and Skaggs (1992). This low outflow rate caused by the very low channel slopes can result in delaying peak flows, and finally increased flooding at each field of watershed. As slope increases, the channel did not hamper flow and field hydrology dominates watershed behavior (Konyha and Skaggs, 1992).

Konyha and Skaggs (1992) reported that channel bedslope was the second most sensitive parameters in their sensitivity test. They also mentioned that channel bottom slope was one of the important factors influencing the channel carrying capacity, and then outflow. As channel slope increases, generally carrying capacity of channel increases, and then outflow rates increases. However, the hydrology of their study was not significantly affected by the increased channel carrying capacity resulted from increased bottom slope and changes of other parameters.

Table 2.20. Effects of channel bedslope on predicted storm event hydrograph characteristics.

Dates of events	Channel slope	Peak flow rate (mm/day)	Time to peak (days)	Volume of event outflow (mm)
	s=0.00005	8.2	2	14.7
1. Aug. 04-14, 1999	s=0.0001	8.1	2	12.9
	s=0.0002	7.5	2	12.5
	s=0.0003	6.7	2	12.6
	s=0.00005	0.0	3	0.01
2. Jun 01-11, 2000	s=0.0001	0.6	3	0.99
	s=0.0002	0.4	3	0.86
	s=0.0003	0.9	3	1.21
	s=0.00005	24.3	2	122.8
3. Jul.23~	s=0.0001	30.8	2	120.0
Aug.02, 2000	s=0.0002	48.1	2	120.0
	s=0.0003	56.7	2	117.5
	s=0.00005	8.9	3	21.9
4. Mar. 03~14, 2001	s=0.0001	9.5	3	21.3
	s=0.0002	11.5	3	19.2
	s=0.0003	12.1	3	18.8

2.3.3.4. Methods of calculating Potential Evapotranspiration (PET)

Cumulative daily outflows obtained by using three different PET calculation methods in the model were compared (Fig. 2.13 and Table 2.21). Those three methods were Penman-Montheith (PM), Thornthwaite 1 (Thorn1), and Thornthwaite 2 (Thorn2). The PET method for the base simulation was PM. Based on student's t-test, significant differences in outflows were found between Thorn1 and other two methods ($p<0.05$), but the difference between PM and Thorn2 was not significant ($p>0.05$). These discrepancies may be mainly caused by the different amount of ET estimated by each method (Amatya et al., 1997), and may be considered as reasonable. The amount of ET in forested watershed is generally greater than that of agricultural watershed. The monthly correction factors of Thorn1 were average values with one set of correction factors from forested watershed in Carteret County and with two sets from agricultural watersheds in

Washington and Edgecomb County (Amatya et al., 1995). On the contrary, the set of monthly correction factors of Thorn2 was solely from agricultural research site.

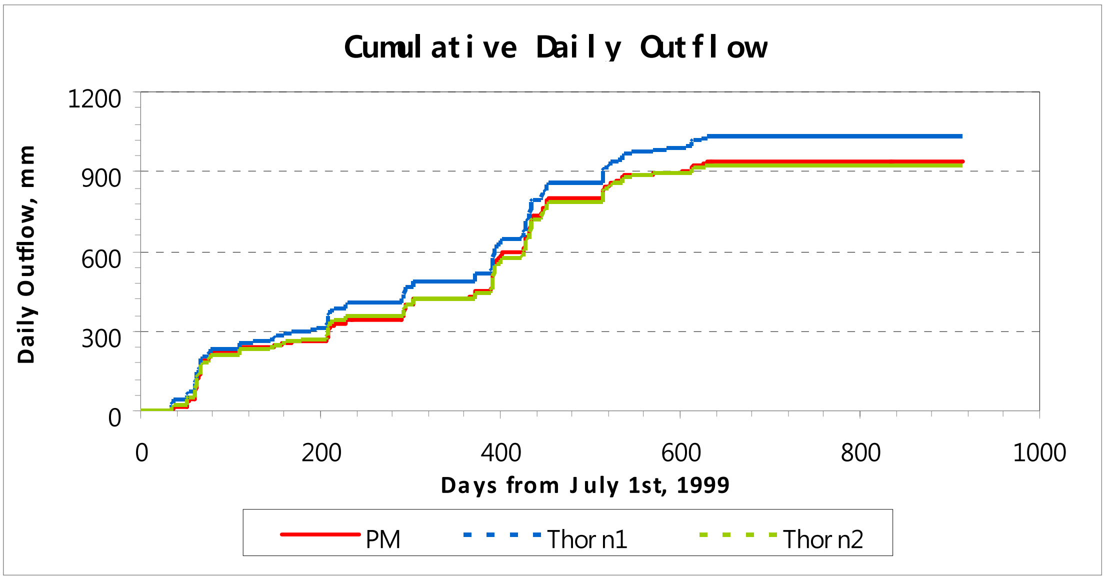


Fig. 2.13. Sensitivity of cumulative outflow to PET using different calculation methods.

Table 2.21. Effects of different PET methods on simulated cumulative outflows for the 1999-2001 study period. In 1999, only simulation results of six months (July – December) were presented.

Year	Cumulative outflow based on different PET calculation methods, mm		
	PM	Thorn1	Thorn2
1999	264.2	299.4	265.8
2000	624.8	672.6	619.2
2001	314.1	362.1	307.5

The effects of PET methods simulated storm events were presented and summarized in Fig. 2.14 and Table 2.22. With using different PET methods, the simulated outflows of selected storm events were very sensitive based on soil moisture conditions and the amount of peak flow. In the case of small peak flow in event 2, significant differences were observed with different PET methods based on student t-test

($p < 0.05$). On the contrary, there were no significant differences in high peak flow in event 3 ($p > 0.05$).

In the intermediate peak flows in events 1 and 4, simulated hydrology was substantially different according to the initial soil moisture condition. When soil was dry at the first day of flow in event 1, the peak flow rate and volume of outflow events were significantly different ($p < 0.05$). However, no significant differences were observed with different PET methods when the soil was already saturated due to rainfall events of previous days at the first day of event ($p > 0.05$).

The Thornthwaite method generally overpredicts outflows. The monthly correction factors of Thornthwaite methods were originally developed to mitigate the PET calculation errors resulted from substantially underpredicted evaporation losses of intercepted water from pine stands in North Carolina (Helvey, 1967; cited by Amatya et al., 1997). Even though these errors are reduced by correction factors in DRAINMOD (T2), the Thornthwaite method still underpredicts ET and then finally overpredicts outflows, especially for the winter period, T1 correction factors were developed by Amatya et al. (1995), but our sensitivity analysis indicates that simulated outflows in agricultural watersheds are overestimated if PET is calculated using T1 correction factors.

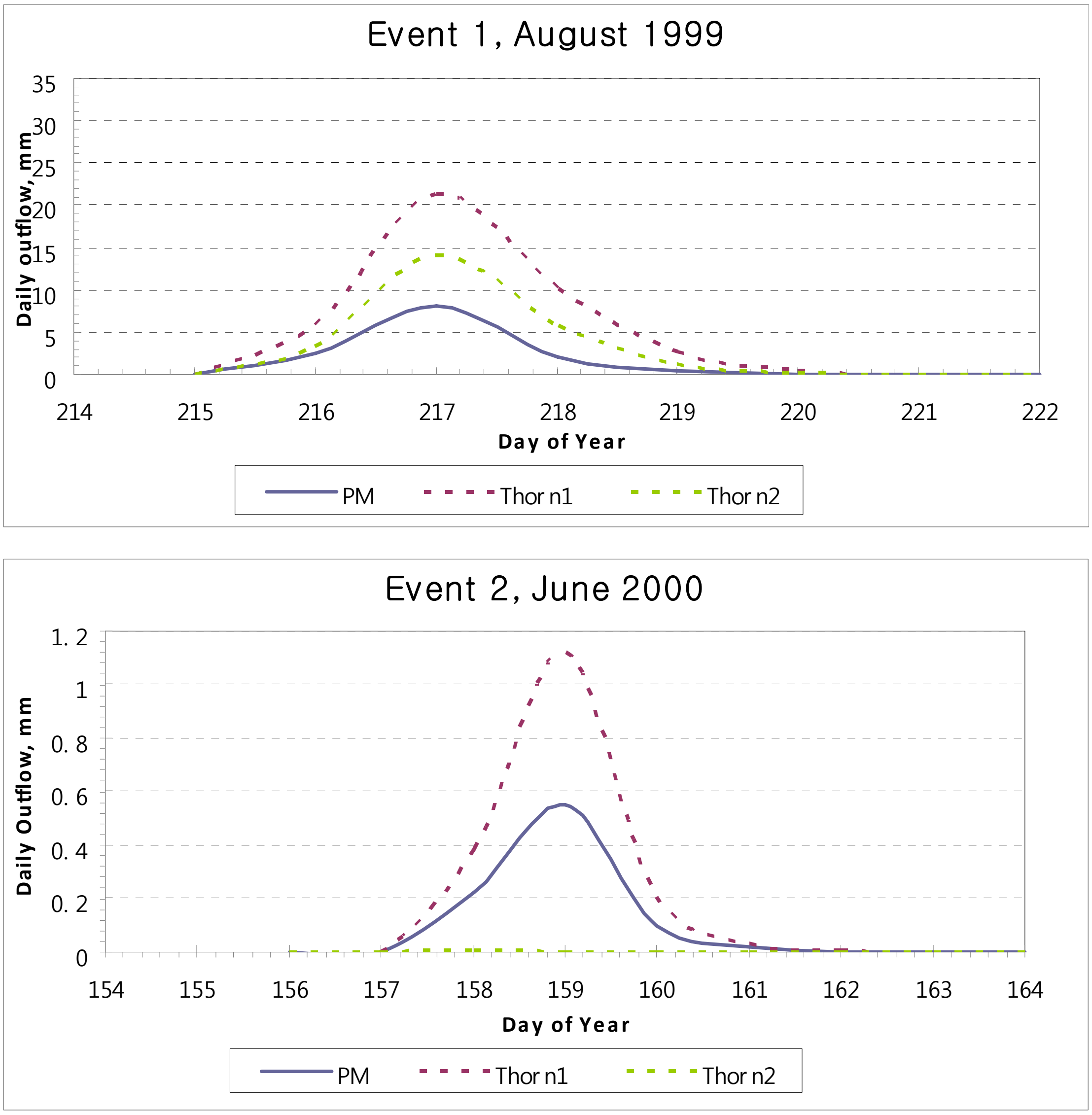
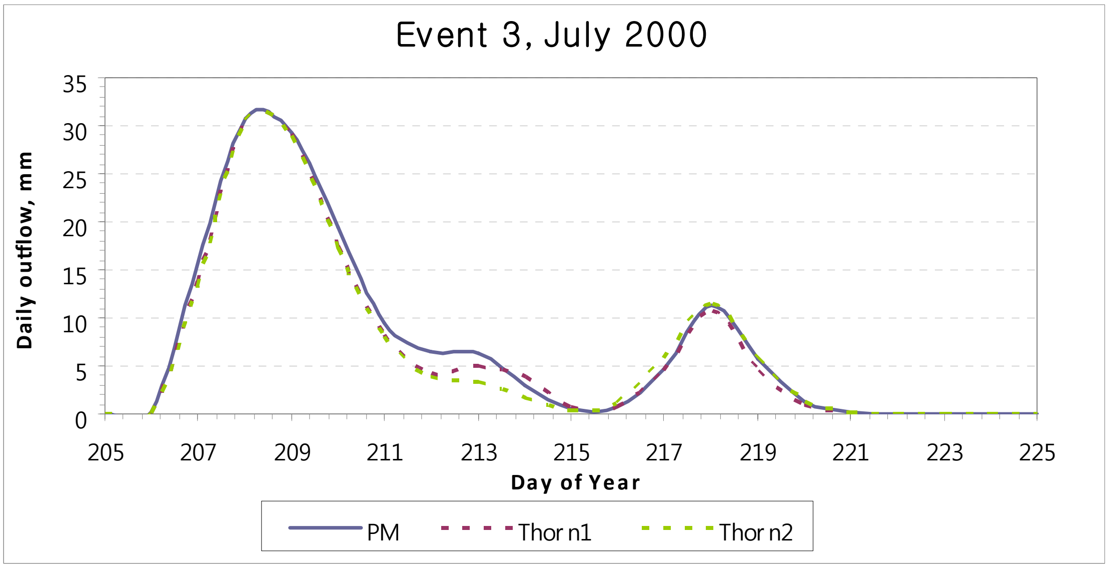


Fig. 2.14(a). Sensitivity of outflow rates to PET by different calculation methods in events 1, and 2.



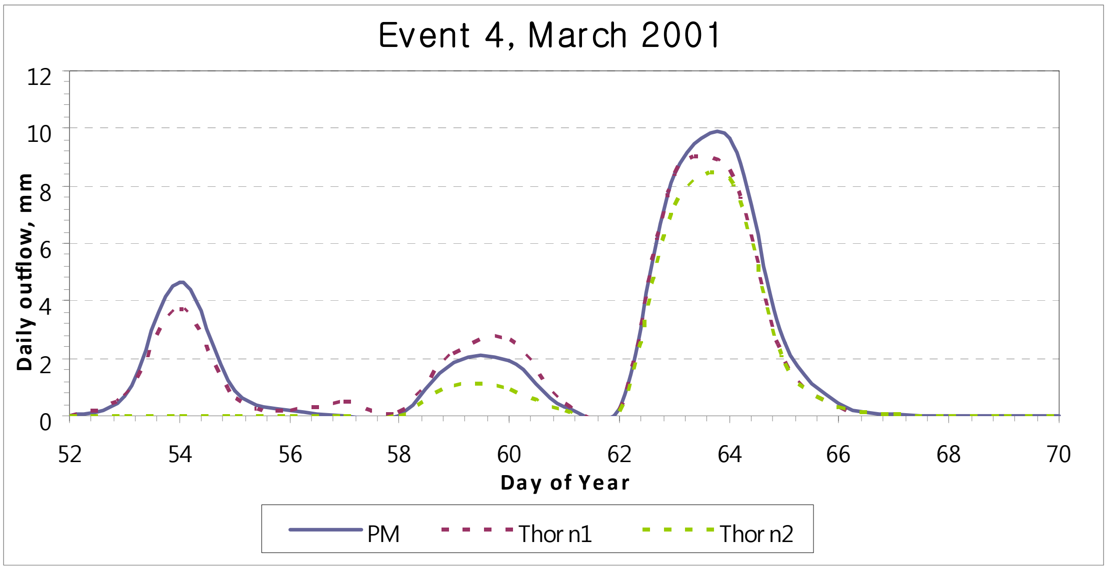


Fig. 2.14(b). Sensitivity of outflow rates to PET by different calculation methods in events 3, and 4.

Table 2.22. Effects of PET methods on predicted storm event hydrograph characteristics.

Dates	*PET methods*	*Peak flow rate* (*mm/day*)	*Time to peak* (*days*)	*Volume of event outflow* (*mm*)
	PM	8.1	2	12.9
Aug. 04-	Thorn1	21.3	2	40.4
Aug. 14, 1999	Thorn2	14.0	2	24.7
	PM	0.6	2	0.9
Jun. 01-	Thorn1	1.1	3	1.8
Jun. 11, 2000	Thorn2	0.0	3	0.0
	PM	30.8	2	121.1
Jul. 23-	Thorn1	30.6	2	112.7
Aug. 02, 2000	Thorn2	30.5	2	107.2
	PM	9.7	3	21.3
Mar. 03 -	Thorn1	8.6	3	20.6
Mar. 14, 2001	Thorn2	8.2	3	18.2

2.4. Conclusion

DRAINWAT model was tested with 30 months of data (July 1st, 1999 - December 31st, 2001) on a 1037 ha agricultural subwatershed in Open Grounds Farm. The main objectives of this study were 1) to evaluate the hydrology, especially drainage outflows, 2) to analyze the sensitivity of the model input parameters, and 3) to explore the potential of estimating the nitrogen loading from the study watershed. Reliable measured data were limited for several reasons. No soil hydraulic property data from the actual study site were available, and therefore, all literature data were used. Rainfall and other weather data were also not available for the study site at Open Grounds Farm, but from the Carteret site located 8.9 km from study site, which may cause error due to spatial variability of rainfall. Compared with other two studies nearby in Carteret and Plymouth, substantial differences of runoff ratio among three sites were observed. It is likely because of these reasons, simulation results of annual and daily outflow from the study watershed were found not to be in agreement with measured data.

Measured daily outflows from only the reliable periods of short storm events were selected and compared with the simulated outflows for storm event analysis. Comparing the values of r^2 and Nash-Sutcliffe coefficient (E) with other studies in the literature, the simulation results were still found to be poor and inconsistent with measured data. For these reasons, this study was focused on analyzing the sensitivity of simulated outflows using Manning's roughness coefficient, maximum depressional storage (STMAX), channel bed slope, and PET by different calculation methods. PET was the most sensitive input parameters to the simulated model outflows. Sensitivity analysis of input parameters indicated that daily cumulative outflow did not change significantly as the values of parameters changed except for the method of PET calculation. However, some changes in daily outflows were visible in selected short term storm events as parameter values changed. This means that PET was the most sensitive among parameters analyzed in this study. Simulation results using Thornthwaite method with Thorn 2 correction factors is found to be more consistent with baseline simulation than those of Thorn 1. The main reason is that Thorn 2 correction factors were collected and estimated only from agricultural watersheds, and Thorn 1 correction factors were from agricultural and forested watersheds.

It was not possible to evaluate the water quality without accurate hydrology data, and water quality simulation was not completed. Only the simulation methods were re-introduced from Amatya et al. (2003, 2004) for future application studies. DRAINWAT can be effective for evaluating hydrology of various scenarios for improving agricultural and silvicultural water management, once the model can be calibrated properly with measured data as shown by its application in several other studies.

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3. Nitrogen Transformations in Constructed Wetlands at the Outlet of an Agricultural Watershed

ABSTRACT

Excessive amounts of nitrogen transported in surface and subsurface drainage from agricultural fields may deteriorate surface and subsurface water quality. Treatment wetlands are effective filters that remove nitrogen by retaining sediments, plant uptake, and denitrification. Wetlands are dynamic ecosystems and it is difficult to predict their efficiency. However, models provide a useful tool to estimate nitrogen removal. This study was conducted to monitor the vegetation succession, to evaluate the efficiency of nitrogen removal, and to test the sequential nitrogen model of the treatment wetland in the outlet of an agricultural watershed at Open Grounds Farm in Carteret County, North Carolina. The treatment wetland at this site was a constructed brackish water tidal marsh receiving water from ditches draining a 1037 ha watershed that is used for crop production. Three brackish water plant species, *Juncus roemerianus*, *Cladium jamaicense*, and *Spartina alterniflora*, were planted in a randomized complete block design. The sequential nitrogen model was applied to predict the concentration change of each nitrogen species (dissolved organic nitrogen, ammonium, and nitrate) in the inlet with 6 rainfall events. Our results of statistical analysis showed that this model did not yet properly predict the concentration of nitrogen species from drainage water by the wetland system in the inlet. If this model can be adequately calibrated with more measured data, it may be useful for decision makers to estimate the amount of nitrogen removed by treatment wetlands.

3.1. Introduction

Constructed wetlands have been designed and operated to simulate the water quality improvement function of natural wetlands, which frequently are sinks for large amounts of nutrients (Mitsch and Gosselink, 1993). This technology for wastewater treatment has been gradually emerging since the 1950's (Watson, 1992). Constructed wetlands are economical and environmentally-safe alternatives for agricultural wastewater treatment, especially nitrogen removal (Bhamidimarri et al., 1991).

An elevated level of nitrogen drained from agricultural watersheds can result in ground and surface water contamination. In addition, it is relatively simple to operate wetlands for the purpose of wastewater treatment, and wetlands are beneficial for esthetics, and wildlife utilization and production (Watson, 1992). However, constructed wetlands are dynamic ecosystems and it is difficult to predict their efficiency. Models possibly provide a useful tool for understanding wetland function and structure, testing hypotheses, and making predictions. Many models have been defined and used in various research fields such as mathematics and physics.

Even though many studies have focused on wetland modeling, it is still relatively new compared to modeling of aquatic or terrestrial ecosystems. Wetland models have been developed recently to determine the removal and retention of nitrogen in wetlands as water from various sources flows through the wetlands and into adjacent waterbodies. The actual nitrogen removal by a wetland is dependent on the hydrologic, geochemical, and redox condition, and vegetation. In order to explain these intricate nitrogen removal processes from the water flowing through constructed wetlands, a black box approach was used in the beginning. A boundary is set up around the wetland, and only inflow and outflow are considered. Retention in wetlands is the difference between those two (Howard-Williams, 1985). This approach evaluates nitrogen loss from an overall N removal term (k_N) and total nitrogen (TN) in the influent. However, it is not accurate because k_N values in the literatures vary greatly (Gerke et al., 2001; Kadlec and Knight 1996). This approach ignores the complex nitrogen cycle in wetlands (Gerke et al, 2001). Therefore, models have been developed to evaluate internal transformation in wetlands more accurately.

Kadlec and Hammer (1988) developed a mathematical model to simulate wetland hydrology and nutrient driven interactions between wastewater and the wetland ecosystem. Spatial variations due to surface water flow were described, and mass balance was calculated for P, N, and Cl. The biogeochemical processes in these ecosystems were represented using a one-dimensional, spatially distributed compartmental model. Compartments include soil, subsurface water, interstitial soil water, and various types of live biomass, standing dead and litter. This model was developed based on REBUS (Routine for Executing Biological Unit Simulations) computer simulation routine.

Dørge (1994) developed MIKE11 WET, a general simulation model consisting of a hydrological submodel and a biological submodel containing heterotrophic nitrogen dynamics and plant uptake. He modeled the biogeochemical pathways of nitrogen from organic matter to ammonia and further to nitrate. His model was relatively simple, but it could take the interactions between important biological factors into account. It could also quantify the site-specific nitrogen removal processes in wetlands influenced by the local hydrologic and nitrogen loading.

Van der Peijl and Verhoeven (1999) developed a model to investigate the dynamics and the interactions of C, N, and P in riverine wetlands. This model was composed of submodels of C, N, and P dynamics that have the same basic structure, but are not identical. This model showed that those intricate processes and their interactions in wetlands resulted in temporal dynamics of state variables, which are in general agreement with dynamics explained for wetland ecosystems. In addition, they found that the most important factor was the maximum relative growth rate of the plants through the sensitive analysis. Their model also pointed out that decomposition is the most important factor in nutrient dynamics in riverine wetlands.

Lee et al. (1999) developed the SET-WET model. This is a dynamic, compartmental simulation model for design and evaluation of constructed wetlands to optimize non-point source pollution control measures. Their model simulates the hydrologic, nitrogen, carbon, dissolved oxygen, bacteria, vegetative, phosphorus and sediment cycles of a wetland system. This model can be used by an existing hydrologic simulation model, such as ANSWERS or BASINS, but may also be used in situations where measured input data to the wetland are available.

The same authors modified and extended this SET-WET model and developed the WETLAND model (Lee et al., 2002). This model was found to be effective in predicting the five day biological oxygen demand (BOD_5), suspended sediment, nitrogen, and phosphorus, but not so effective for predicting the effluent dissolved oxygen (DO). The most significant input parameters in this model were those parameters directly affecting bacterial growth and DO uptake and movement.

Wynn and Liehr (2001) presented a mechanistic, compartmental simulation model of subsurface flow constructed wetlands. This model consisted of six submodels including N and C cycles, autotrophic and heterotrophic bacterial growth and metabolism, and water and oxygen balances. This model was generally insensitive to changes in individual parameters because of the complexity of the ecosystem and the model as well as the numerous feedback mechanisms. The model was most sensitive to changes in parameters directly affecting microbial growth. The approaches and modeling objects of these models were summarized in Table 3.1.

More recently, Mayo and Bigambo (2005) developed a mathematical model for prediction of nitrogen transformation in horizontal subsurface flow constructed wetlands. Six major N transformation processes including mineralization, nitrification, denitrification, plant uptake, plant decay, and sedimentation were considered in their study. These models are briefly described in Table 3.1 with their references.

Almost all of these models are complex and have many parameters. Although the complex models are capable of simulating the impacts of the dynamics of various nutrient transformation processes, they are usually difficult for planners and managers to use owing to high input data requirements, problems in calibration, and parameterization (Fernandez et al., 1999). Model users and modelers need to find the proper balance between the model's accuracy, accessibility, and its input requirement (Lee et al., 1999). The intermediate model to evaluate nitrogen removal by constructed wetlands may possibly be an alternative.

Gerke et al (2001) calibrated and validated the sequential model of nitrogen transformation that was first suggested by Kadlec and Knight (1996) for a wetland constructed for treating lagoon effluent in Kingman, Arizona. Even though this model can be useful for nitrogen management, it has not been verified widely yet.

Table 3.1. Modeling approaches on nitrogen and other nutrients in wetlands.

General model descriptions	Nutrients	References
Spatially distributed hydrology and ecosystem model	N, P, Cl	Kadlec and Hammer, 1988.
MIKE 11 WET. General model for freshwater wetlands in temperate zone	N	Dørge, 1994.
Dynamic simulation model consisting of C, N, and P submodels. The state variables are connected through flows of C, N and P within each submodel.	N, C, P	Peijil and Verhoeven, 1999
SET-WET. Dynamic compartmental simulation model designed for evaluating constructed wetlands to optimize non-point source control measures.	N, C, P	Lee et al., 1999
WETLAND. Extended from SET-WET.	N, C, P	Lee et al., 2002
A mechanistic, compartmental simulation model for subsurface flow constructed wetlands.	N, C	Wynn and Liehr, 2001
A sequential nitrogen model capable of recognizing the reactivity of nitrogen species in the inflow and predict concentrations of individual N species in the effluent.	N	Gerke et al., 2001
A mathematical model for prediction of nitrogen transformation in horizontal subsurface flow constructed wetland.	N	Mayo and Bigambo, 2005

The objectives of this study were 1) to evaluate the nitrogen transformations in constructed wetlands using a sequential nitrogen model, and 2) to develop approaches to enhance nitrogen removal by constructed wetlands based on this model.

3.2. Materials and Methods

3.2.1. Site description

A set of treatment wetlands was constructed at the outlet of a watershed at Open Grounds Farm in 1999. The farm is about 14,000 ha (35000 acres) in size and is located in eastern Carteret County, North Carolina. The farm is located adjacent to the Neuse River, Pamlico and Core Sound Estuarine System, a breeding and nursery ground for a wide variety of finfish and shellfish and an important habitat for many birds and animals. See Chapter 2 for details.

This set of treatment wetlands (Fig. 3.1) was constructed to process water draining from a watershed of 1037 ha (2700 acres), which is all used for crop production. The total size of constructed wetlands is 5 ha, and less than 0.5% of the watershed. Mitsch (1992) suggested that 3-5% of contributing areas was reasonable size of natural wetlands for effective nonpoint source control. However, our sophisticated design, including a serpentine flow pathway and intermittent wetland pond, enables this wetland system to maximize nutrient removal and retention (especially via denitrification) as well as pathogen and sediment removal (Poe et al., 2003).

This wetland system is composed of four blocks (30 m × 240 m) and each block has four cells 30 m × 20 m in size. Three hydrophytic plant species, *Juncus roemerianus, Cladium jamaicense,* and *Spartina alterniflora*, were planted. One cell in each block (control) was left unplanted to study natural plant succession and to examine differences in N removal vial denitrification between planted and unplanted cells (Poe et al, 2003). These four cells were positioned in a randomized complete block design to compare the treatment effects of the vegetation. The dominant soil was Deloss fine sandy loam (fine-loamy, mixed, semiactive, thermic Typic Umbraquult). More detail on wetland system design can be found in Poe et al. (2003).

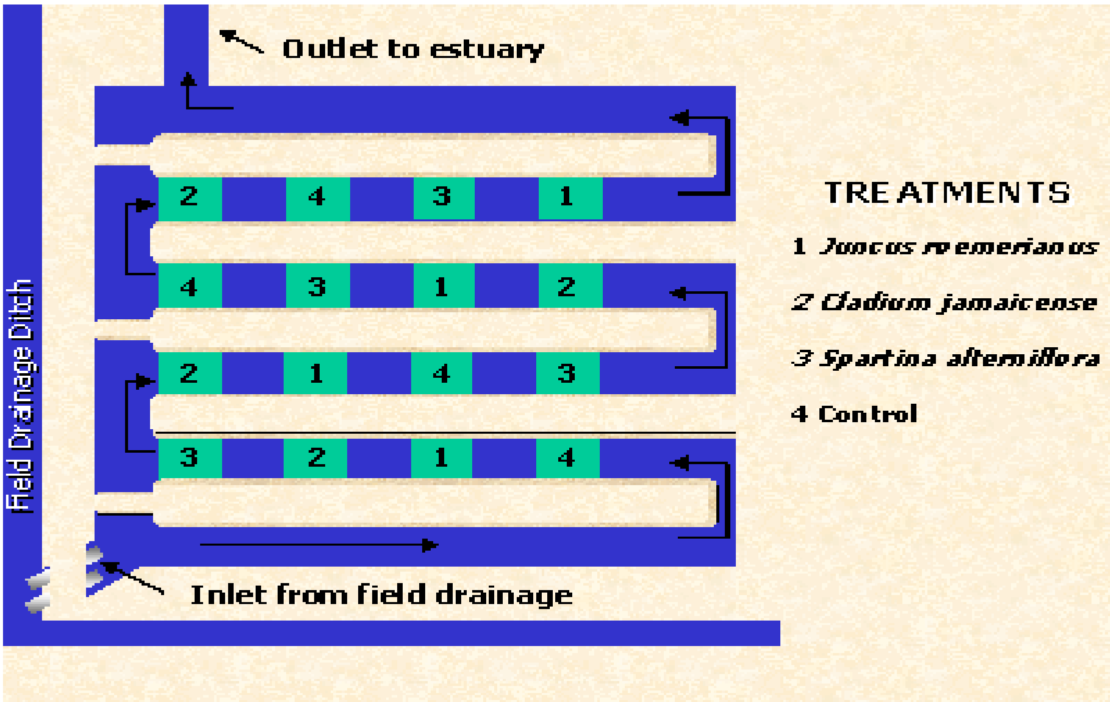


Fig 3.1. Constructed wetlands at the outlet of an agricultural watershed at Opengrounds Farm, Carteret County.

3.2.2. Sample collection and chemical analysis

Water Samplers (ISCO model 6700, Lincoln, NE) were installed at the inlet and outlet of the constructed wetland by personnel of the UNC Institute of Marine Sciences who collected data using the following methods (Poe et al., 2003). Samplers monitored and recorded the amount and velocity of flow, water level, temperature, dissolved oxygen (DO), salinity, and pH. Data were recorded every 30 minutes and downloaded every two weeks. Bi-weekly nutrient samples were collected from a transect which included: a site on the farm, the 16 wetland ponds, inlet to and outlet from the wetland, and a downstream site on the South River. Concurrent with nutrient sampling, a hand held Yellow Scientific Instruments Probe was used to record salinity, pH, temperature, and DO at each site. Water samples were filtered through Whatman GF/F glass fiber filters (25mm) and the filtrate was analyzed with a Lachat QuickChem automated ion analyzer for NOx, NH_4^+ and PO_4^{3-} concentrations (Lachat Instruments, Milwaukee Wisconsin, USA: NH_4^+, Method 31-1-7-06-1-A; NO_2^-/NO_3^-, 31-107-04-1-C; PO_4^{3-}, 31-115-01-3-A). Among these data, the data of nitrogen loading were used for our modeling study. The total N load into and out of the wetland was calculated from the sum of daily water volume based on 0.5-hour records of velocity, level, and biweekly nutrient concentration interpolated to a daily time scale. ISCO samplers were set to collect 500 mL water samples at flow-paced intervals following rain in excess of 5 cm. Sampling was set to facilitate collection throughout the entire event. Water samples and monitoring data were collected from the field within 12 hours of a rain event. More details on water sampling and methods of analysis can be found in Poe et al. (2003). Aboveground plant biomass was determined by harvesting plant material in 0.25 m^2 quadrats from twelve wetland cells once in each of three years in November at the end of the growing season. Samples from quadrats were transferred into the drying room and dried at 70°C for three days.

3.2.3. Modeling procedure

Modeling is an individual activity, and all modelers approach a problem differently. The phases described and identified here are based on Jørgensen (1989).

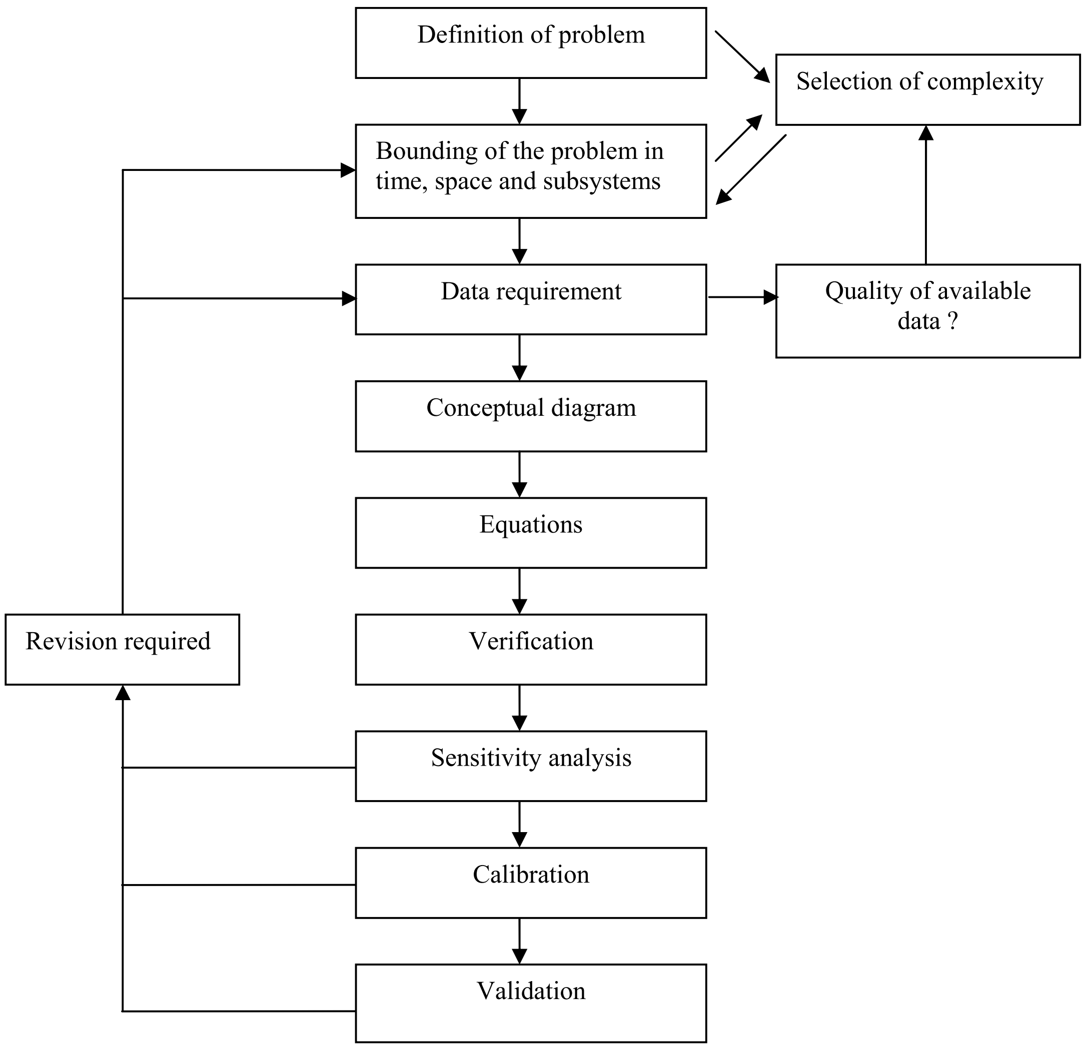


Fig. 3.2. A tentative modeling procedure (Jørgensen, 1989. Revised and used under the permission of the author**).**

Based on his suggestions above in Fig. 3.2., we lumped several steps together and followed this process. Other processes can be added or removed according to a modeler's preference.

3.2.3.1. Determining the goal and scope

A number of physical processes that transport nitrogen compounds in wetlands without molecular transformation are as follows. 1) particulate settling and resuspension, 2) diffusion of dissolved forms, 3) plant uptake and translocation 4) litterfall, 5) ammonia volatilization, 6) sorption of soluble nitrogen on substrates, 7) seed release, and 8) organism migrations (Kadlec and Knights, 1996), and 9) dissolved nitrogen transport by water flow. However, these processes are just considered as 'relocations' of nitrogen in a wetland ecosystem. In order to evaluate the nitrogen removal from the water in constructed wetlands, the processes related to nitrogen transformation need to be identified at first.

There are five principal processes that transform nitrogen from one form to another: 1) mineralization, 2) nitrification, 3) denitrification, 4) nitrogen fixation, and 5) nitrogen assimilation. Among these various processes, nitrogen fixation is not generally observed in a treatment wetland where receiving wastewater is high in nitrogen. The assimilation process (plant uptake) is also excluded in this study because assimilated nitrogen in hydrophytes is usually returned when they die back in winter like other physical transport processes introduced above in some extent. Three major processes including mineralization, nitrification, and denitrification were selected for the sequential model of nitrogen transformation (Kadlec and Knight, 1996).

3.2.3.2. Building a conceptual model

The sequential model was selected and modified to recognize the reactivity of nitrogen species in the influent and to predict concentrations of individual nitrogen species in the effluent. This model was also used to identify several approaches to improve the efficiency of nitrogen removal.

This mathematical approach was suggested by Thomann and Mueller (1987) to predict and analyze the performance of wastewater treatment wetlands, and its basic assumption is that the systems are steady-state and plug-flow reactors. No dispersion or mixing was assumed when nitrogen is transported (Kadlec and Knight, 1996). Field data from the wetland were utilized in order to calibrate the sequential model of nitrogen transformations suggested by Gerke et al. (2001), Kadlec and Knight (1996), Thomann and Mueller (1987).

The reaction network of a conventional nitrogen cycle in treatment wetlands is presented in Fig. 3.3. This simplified nitrogen reaction network is assumed to be composed of interconversions of nitrogen in the water accompanied by exchanges with the sediments and biomass and the atmosphere (Kadlec and Knight, 1996). This network also allows the exchange of ammonium nitrogen (AN) between the water and the live and dead biomass (B). Uptake into living plants and sorption onto sediments are accounted by an uptake flux (J_{UAN}). Uptake of nitrate nitrogen (NN) by macrophytes is also an allowed transfer route (J_{UNN}) even though this route is considered not so important as AN uptake. Organic nitrogen release by biomass decomposition is another transfer route (J_{RON}).

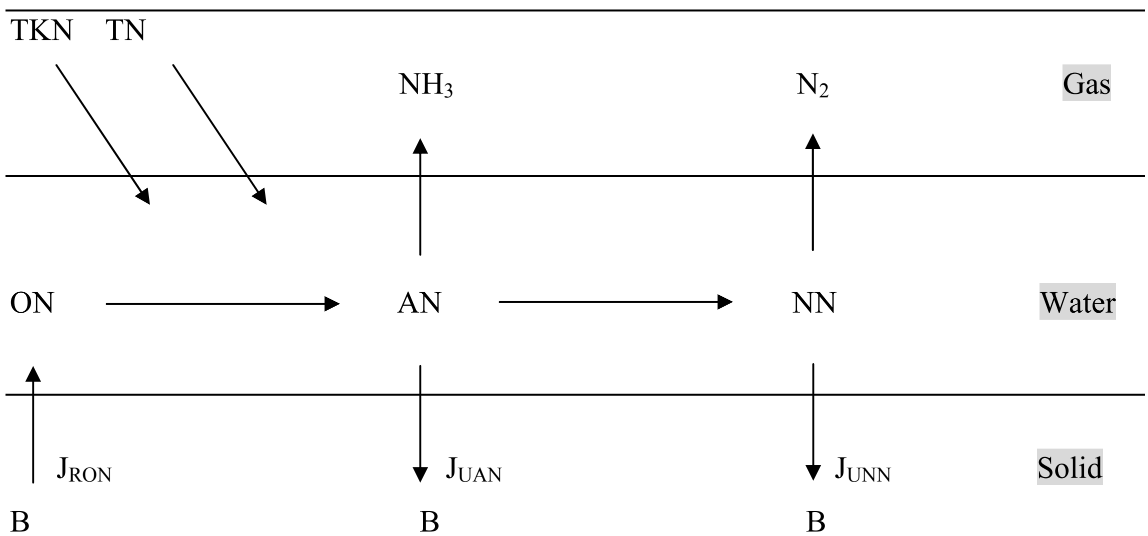


Fig. 3.3. A simplified sequential reaction processes of nitrogen transformation in the wetland environment. Organic nitrogen (ON) may be ammonified to ammonium nitrogen (AN). The wetland contributes organic nitrogen from decomposition of biomass (B). Ammonium may be lost via ammonia volatilization, nitrification, sorption, and plant uptake. Nitrate is formed by nitrification and lost by denitrification and uptake. Transfers

to and from the wetland sediments and biomass are denoted by the fluxes. The first order areal rate constants are denoted by k. (Kadlec and Knights, 1996. used under permission)

Ammonium can be irreversibly converted to ammonia gas, and this conversion process is first order. Nitrate may be converted to nitrogen gas via denitrification, and this process is also first order. More detailed explanation for this reaction network can be found in Kadlec and Knight (1996). Anaerobic ammonia oxidation (ANAMMOX) is not allowed in this conventional nitrogen reaction network. Many scientists have become interested in ANAMMOX process recently (i.e, Mulder et al., 1995; Dong and Sun, 2007). This process may be useful for studying nitrogen removal by constructed wetlands in near future, but still more documented evidence is needed for it to be included in the nitrogen cycle.

3.2.3.3. Specifying mathematical functions

The differential equations explaining the transformation of each nitrogen species in wetlands were suggested by Thomann and Mueller (1987). These equations were modified and tested by several researchers including Kadlec and Knights (1996) and Gerke et al. (2001) in different environmental conditions.

Under the first-order assumption, three nitrogen forms they mentioned were as follows.

ON = organic nitrogen

NH_4 = ammonium nitrogen

NO_3 = (nitrite and nitrate) nitrogen

Nitrite and nitrate were considered as one because nitrite is in general not chemically stable in most wetlands due to its intermediate energetic state between ammonia and nitrate. For this reason, nitrite is rapidly transformed into nitrate.

$$\frac{d[ON]}{dt} = -K_{11}[ON] \quad \text{..........} \quad (3.1a)$$

$$\frac{d[NH_4]}{dt} = K_{12}[ON] - K_{22}[NH_4] \quad \text{(3.1b)}$$

$$\frac{d[NO_3]}{dt} = K_{23}[NH_4] - K_{33}[NO_3] \quad \text{(3.1c)}$$

K_{11} = Overall loss coefficient of ON due to particulate settlement, hydrolysis, and ammonification.

K_{12} = rate of formation of NH_4 due to ammonification

K_{22} = overall loss of NH_4 due to plant uptake and oxidation to NO_3 (nitrification)

K_{23} = rate of formation of NO_3 due to nitrification

K_{33} = overall loss of NO_3 due to plant uptake or denitrification

These sequential reaction processes were assumed to consist of zero and first order reactions which represent the simplest possible rate equations that may be selected. The resulting model for concentration change as water flows through the wetland is tractable, but necessarily more complicated than for a single species such as BOD_5.

All of ON is not converted to NH_4 because of the sedimentation of particulate ON, and generally $K_{11} > K_{12}$. Under anaerobic conditions in the sediments, denitrification, the bacterial reduction of nitrate, can occur. Nitrate is reduced to nitrogen gas and then released to atmosphere. The low level of nitrate in the sediment sets up a concentration gradient with the higher nitrate concentrations in the water column. After that, diffusion of the nitrate from the water column into the interstitial water of the sediment may occur resulting in nitrate loss from the water column.

The solution of Eq. 1a is

$$[ON] = [ON]_0 e^{-K_{11}t} \quad \text{(3 2a)}$$

Where $[ON]_0$ is the initial value of the organic nitrogen at time t = 0.

Not all of the organic nitrogen may hydrolyze to ammonia in the time scale of a given problem due to the refractory nature of some organic nitrogen forms. The refractory fraction can be subtracted from the more labile organic nitrogen. Sequential substitution of this solution into equation (3.1b) and equation (3.1c) and integration gives the solution for ammonia and nitrate.

$$[NH_4] = \frac{K_{12}[ON]_0}{K_{22} - K_{11}}(e^{-K_{11}t} - e^{-K_{22}t}) + [NH4]_0 e^{-K_{22}t} \quad \text{.. (3.2b)}$$

$$[NO_3] = \frac{K_{12}K_{23}}{K_{22} - K_{11}}[ON]_0 (\frac{e^{-K_{11}t} - e^{-K_{33}t}}{K_{33} - K_{11}} - \frac{e^{-K_{22}t} - e^{-K_{33}t}}{K_{33} - K_{22}}) + \frac{K_{23}}{\mathrm{K}_{33} - \mathrm{K}_{22}}(e^{-K_{22}t} - e^{-K_{33}t})[NH_4]_0 + [NH_4]_0 e^{-K_{33}t} \quad \text{....................................(3.2c)}$$

Gerke et al. (2001) assumed K_{12} equals K_{11}, and K_{23} equals K_{22}. They also used the rate constants which are normalized to depth, and the unit of those rate constants is m/day. In our study, the average values for surface flow wetlands presented by Kadlec and Knight (1996) were used for simulation. K_{11} is 0.049 m/day, is 0.028 m/day, and K33 is 0.041 m/day. The water depth of all events was assumed to be 1 m.

This approach ignores the mechanistic complexities involved in various removal processes and lumps them together into overall first-order rate constants. Wastewater treatment wetlands are designed for fairly constant hydraulic and pollutant loading rates, and the steady state assumption is reasonable for the flow conditions of this site (Carleton et al., 2001).

ON degrades to produce ammonium. The amount of ammonium at the time of interest is reflected by the balance between mineralization and nitrification. Nitrate is obtained from nitrification and then returned to the atmosphere through denitrification.

3.2.3.4. Making mathematical model

Determination of calibration coefficients sequentially:

$$ON \xrightarrow{K_{11}} AN \xrightarrow{K_{12},\ K_{22}} NN \xrightarrow{K_{23},\ K_{33}} N_2$$

Calibration coefficients were determined for each month using field measurements and they were calibrated in sequence. DON data were used to determine K_{11} first. These values of each month were used to determine K_{12} and K_{22}. Finally, values for K_{12} and K_{22} were used to obtain K_{23} and K_{33}.

3.2.3.5. Validation and preliminary statistical analysis

Predicted values of the amount of each nitrogen species were calculated with average calibration coefficients and constants suggested by Kadlec and Knight (1996). The predicted values were compared with measured values. Because the reliable measured flow data is limited, only 5 rainfall events in 2000 and 1 event in 2001 were selected and used to validate the model. Those events are listed in Table 3.2. As we discussed on the methodology in chapter 2, coefficient of determination (r^2) and Nash-Sutcliffe coefficient of efficiency (E) were used to determine the accuracy of simulated outflow concentrations of nitrogen species predicted by sequential nitrogen model in the inlet of constructed treatment wetland system.

Table 3.2. Selected rainfall events used for validating sequential nitrogen model.

Event No.	*Year*	*Day of Year*	*Dates*	*Rainfall, mm*
1	2000	45 – 51	Feb. 14 – Feb. 20	14.5
2	2000	109 – 115	Apr. 18 – Apr. 24	65.0
3	2000	157 – 163	June 5 – June 11	73.7
4	2000	207 – 213	July 25 – July 31	145.3
5	2000	243 – 249	Aug. 30 – Sept. 5	172.0
6	2001	175 – 181	June 24 – June 30	18.0

3.3. Results and Discussion

3.3.1. Biomass

Annual growth measurements for the three species planted in the wetland are found in Table 3.3. Biomass for *S. alterniflora* and *C. jamaicense* was almost equal, but that of *J. roemerianus* was less than other two species in 2000. In 2001, however, *C. jamaicense* and *J. roemerianus* produced greater biomass than *S. alterniflora* ($p<0.05$), but there was no significant difference between *C. jamaicense* and *J. roemerianus.*

Table 3.3. Growth measurements in treatment wetlands in Open Grounds Farm.

	Year	1999 (per plant)	2000 (per m^2)	2001 (per m^2)
Spartina alterniflora	Height	68 ± 5 cm	149 ± 5 cm	167 ± 4 cm
	Stems	26 ± 4	345 ± 33	326 ± 32
	Basal area	-	112 ± 12 cm^2	-
	Biomass	11.7 ± 2.2 g	955 ± 116 g	1364 ± 84 g
	Year	1999 (per plant)	2000 (per m^2)	2001 (per m^2)
Cladium jamaicense	Height	71 ± 4 cm	133 ± 5 cm	145 ± 6 cm
	Stems	4.0 ± 0.5	62 ± 6	117 ± 9
	Basal area	-	239 ± 24 cm^2	-
	Biomass	8.2 ± 1.4 g	1012 ± 116 g	3442 ± 372 g
	Year	1999 (per plant)	2000 (per m^2)	2001 (per m^2)
Juncus roemerianus	Height	59 ± 2 cm	88 ± 2 cm	136 ± 3 cm
	Stems	30 ± 2	1181 ± 107	-
	Basal area	-	115 ± 9 cm^2	-
	Biomass	7.0 ± 0.7 g	598 ± 54 g	2644 ± 141 g

We analyzed the difference of biomass in each species according to block and year. There was very little growth of either of the plant species during the first growing season (Table 3.4) because of late planting (June 10-14, 1999) caused by delays in site preparation. In the second year (2000), our results showed that growth was less for *S. alterniflora* in block 1 than in blocks 2 and 3, and 4 due to weed competition and higher elevation. By 2001, *S. alterniflora* was eliminated from the block 1 plot. The growth measurement indicated that *J. roemerianus* and *C. jamaicense* were better adapted than *S. alterniflora* to the salinity and tidal conditions at the site.

Table 3.4. Biomass (g/m^2) of marsh plant species in the treatment wetland. Values in parentheses are standard errors.

Species		*1999*	*2000*	*2001*
	Block			
Cladium jamaicense	1	14 (4.4)	926 (171.6)	5249 (394.0)
	2	5 (1.7)	1100 (254.2)	4393 (446.4)
	3	5 (1.2)	602 (176.8)	1975 (242.3)
	4	8 (1.3)	1418 (203.3)	2152 (471.7)
	Average	8 (1.4)	1012 (116)	3442 (372)
Juncus roemerianus	1	7 (0.8)	438 (50.9)	3255 (152.3)
	2	7 (1.6)	652 (32.2)	2506 (313.9)
	3	8 (2.1)	587 (96.5)	2666 (279.6)
	4	6 (1.1)	714 (179.1)	2146 (122.0)
	Average	7 (0.7)	598 (54)	2644 (141)
Spartina alterniflora	1	6 (4.2)	346 (39.0)	
	2	17 (4.4)	886 (171.0)	1134 (134.9)
	3	9 (1.2)	1317 (220.9)	1518 (159.9)
	4	16 (5.4)	1274 (148.4)	1441 (94.6)
	Average	12 (2.2)	955 (116)	1364 (84)

Wetlands often experience natural water level fluctuations that results in cyclic vegetation changes (Odland and Moral, 2002), especially in constructed tidal wetlands. Wetland plants are distributed according to their tolerance to flooding and saturation. Odland and Moral (2002) mentioned interspecific competition in establishing wetland zonation of vegetation. However, only one species was planted in each cell (except controlled cells), and there was no interspecific competition between wetland vegetations. The main interspecific competition of the plants in these constructed wetlands might be the battle against weeds!

As many researchers have pointed out, weed invasion into a constructed wetland ecosystem could be a major problem due to competition for resources such as water, light, nutrients, and space. Weed distribution is closely related to increased levels of disturbance in wetlands from activities which include clearing, grazing, altered fire regimes and the spread of dieback (Water notes 33, 2005).

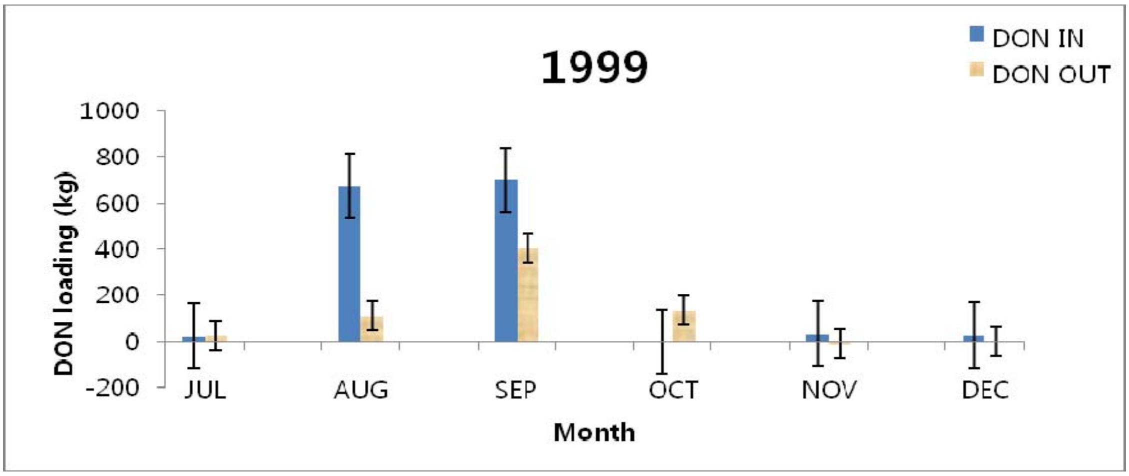
3.3.2. The efficiency of nitrogen removal

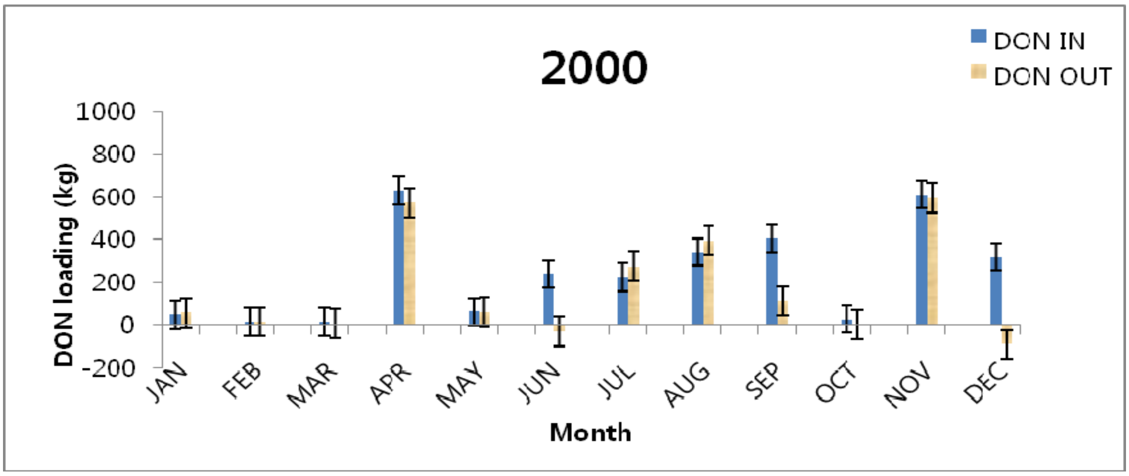
The total amounts of each nitrogen species in the water that flowed into the treatment wetlands from watersheds, and flowed out of the wetlands. Based on these values, nitrogen removal efficiencies for all N species were calculated (Table 3.5).

Table 3.5. The total amounts of each nitrogen species flowing into and out of the treatment wetlands in 1999~2001 (kg). Values in the parentheses are standard errors. (*RE : removal efficiency)

	1999			*2000*			*2001*		
	IN	OUT	*RE (%)	IN	OUT	RE (%)	IN	OUT	RE (%)
NH_4	213.4(46.4)	60.2(12.3)	71.8	347.1(34.6)	337.0(34.0)	2.9	192.5(31.1)	94.7(13.4)	50.8
NO_3	234.9(61.7)	128.1(25.3)	45.4	2010.6(185.3)	807.0(105.7)	59.9	2160.3(50.3)	1152.2(258.1)	46.7
DON	1459.8(23.1)	658.3 (142.4)	54.9	2943.4(213.2)	1980.7(231.6)	32.7	603.5(38.7)	734.0(69.9)	-21.6
TN	1907.7(20.1)	846.7(168.2)	55.6	5301.1(383.4)	3124.7(352.7)	41.1	2956.3(505.1)	1980.2(317.2)	33.0

The nitrogen removal efficiencies ranged from –21.6 to 71.8 %. The negative value of efficiency of DON in 2001 indicates a net gain into the wetland from the estuary. Most of the total nitrogen in 1999 and 2000 was DON, whereas the greatest part of the total nitrogen in 2001 was nitrate-N. The direct comparison with the amount of nitrogen among three years may not be valid because sampling periods were different among years. In addition, there were no hydrology and nutrient data from September 17th through November 3rd because extreme weather (Hurricane Floyd) damaged the water sampler and flow meter. The annual total nitrogen removal efficiencies of each year were 55.6% in 1999, 41.1% in 2000, and 33.0% in 2001. Monthly removal efficiency of each nitrogen species showed differences among sampling years. The monthly distributions of loading of each nitrogen species are shown in Fig. 3.4.





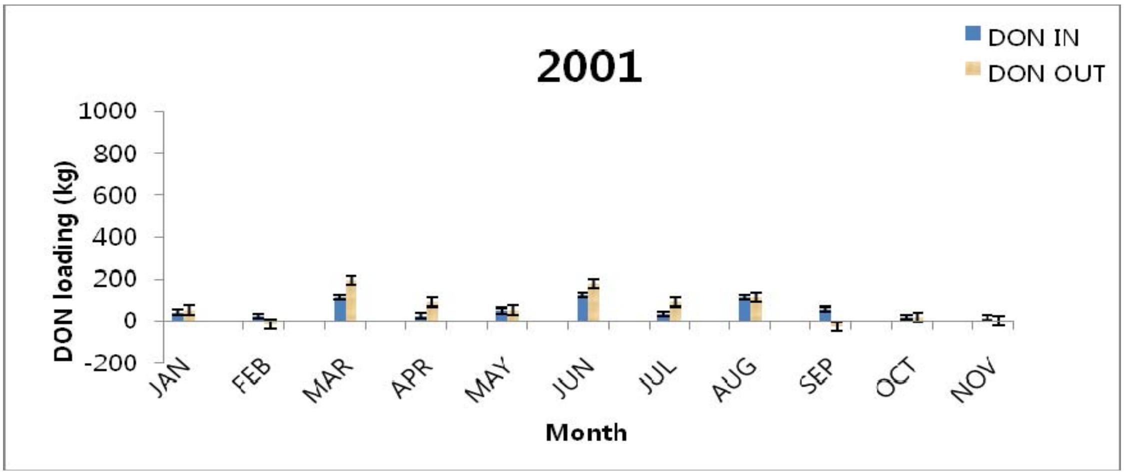


Fig. 3.4(a) The total amount of dissolved organic nitrogen flowing into and out of the constructed treatment wetland each month from july 1999 until November 2001.

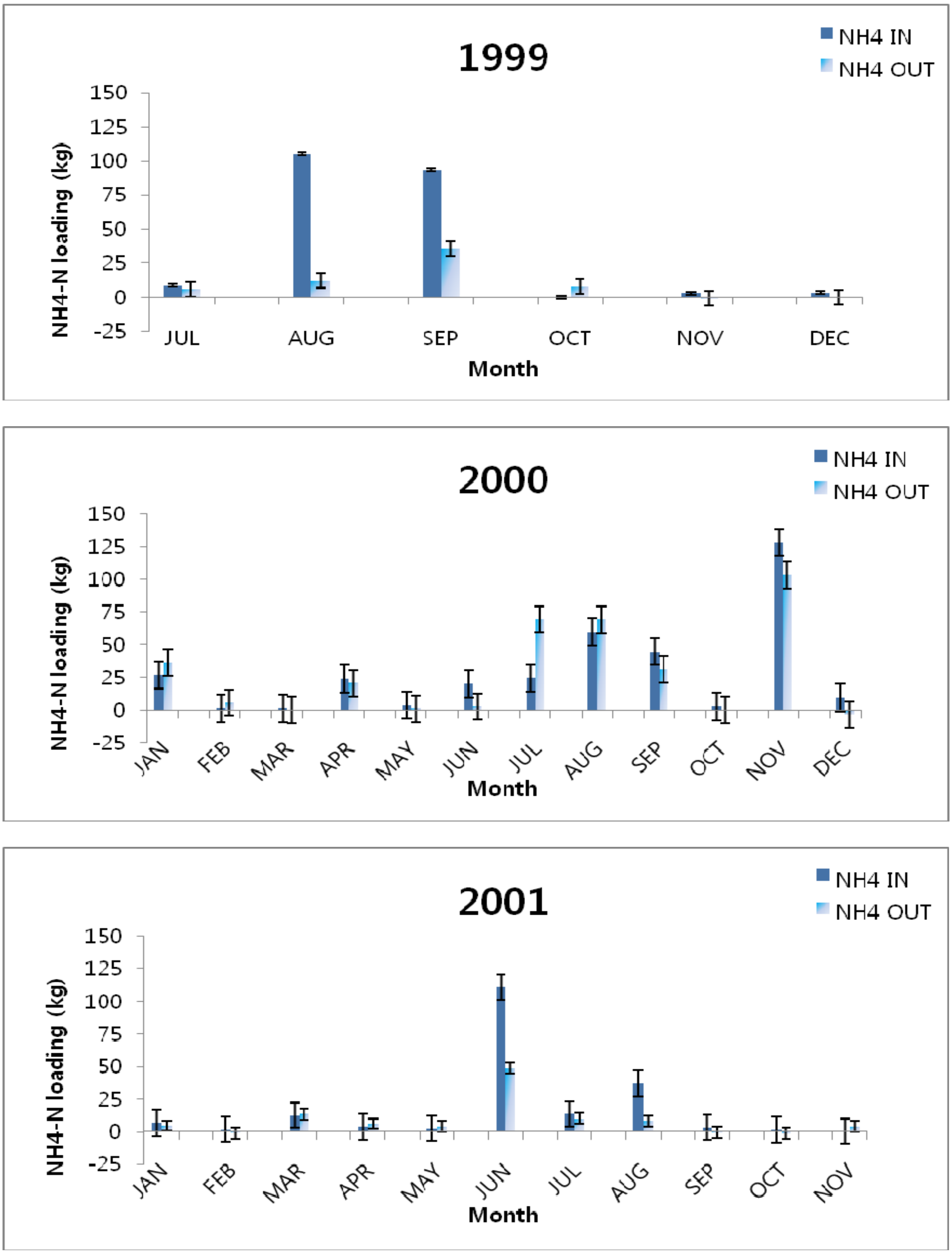


Fig. 3.4(b) The total amount of ammonium nitrogen flowing into and out of the constructed treatment wetland each month from July 1999 until November 2001.

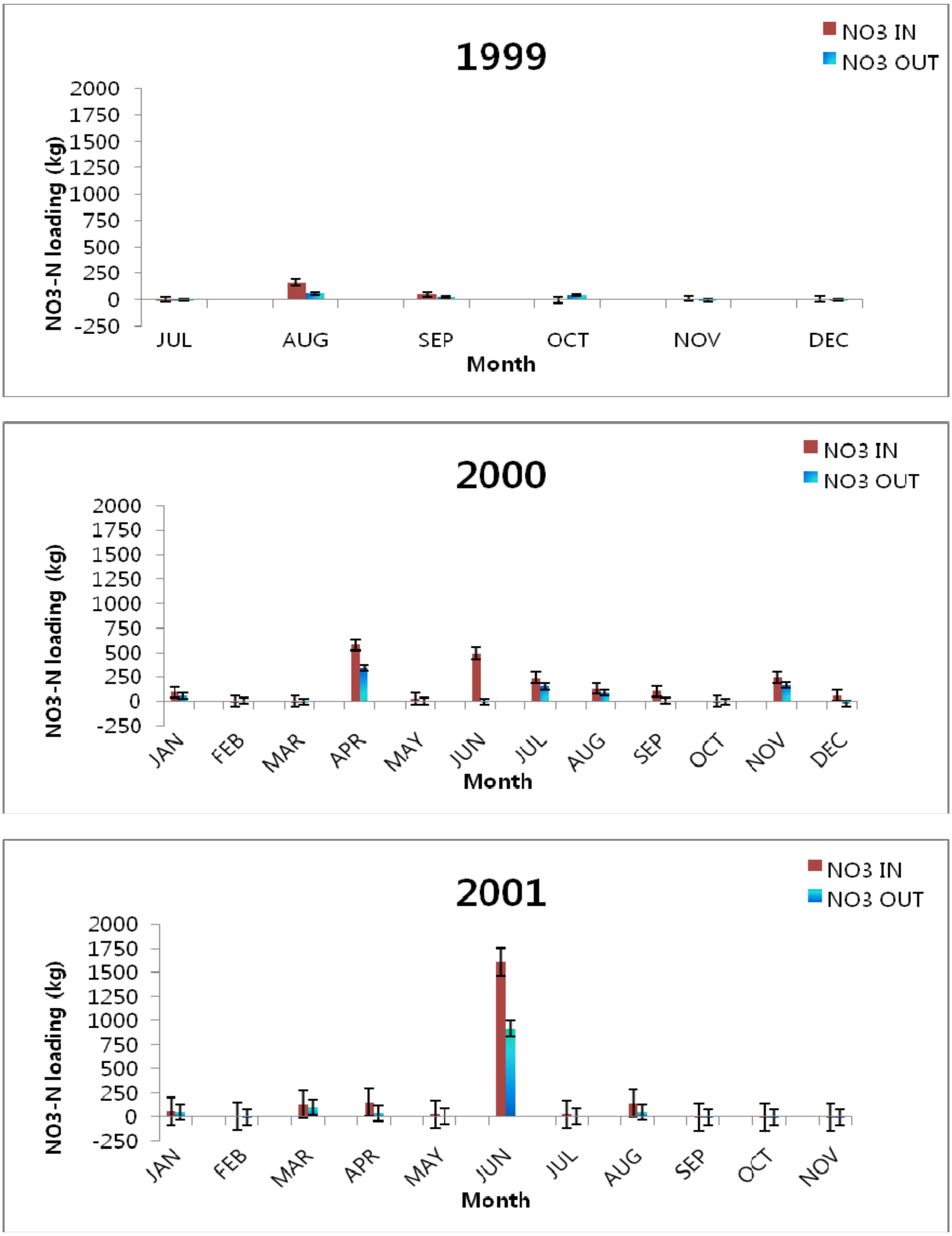


Fig. 3.4(c) The total amount of nitrate nitrogen flowing into and out of the constructed treatment wetland each month from July 1999 until November 2001.

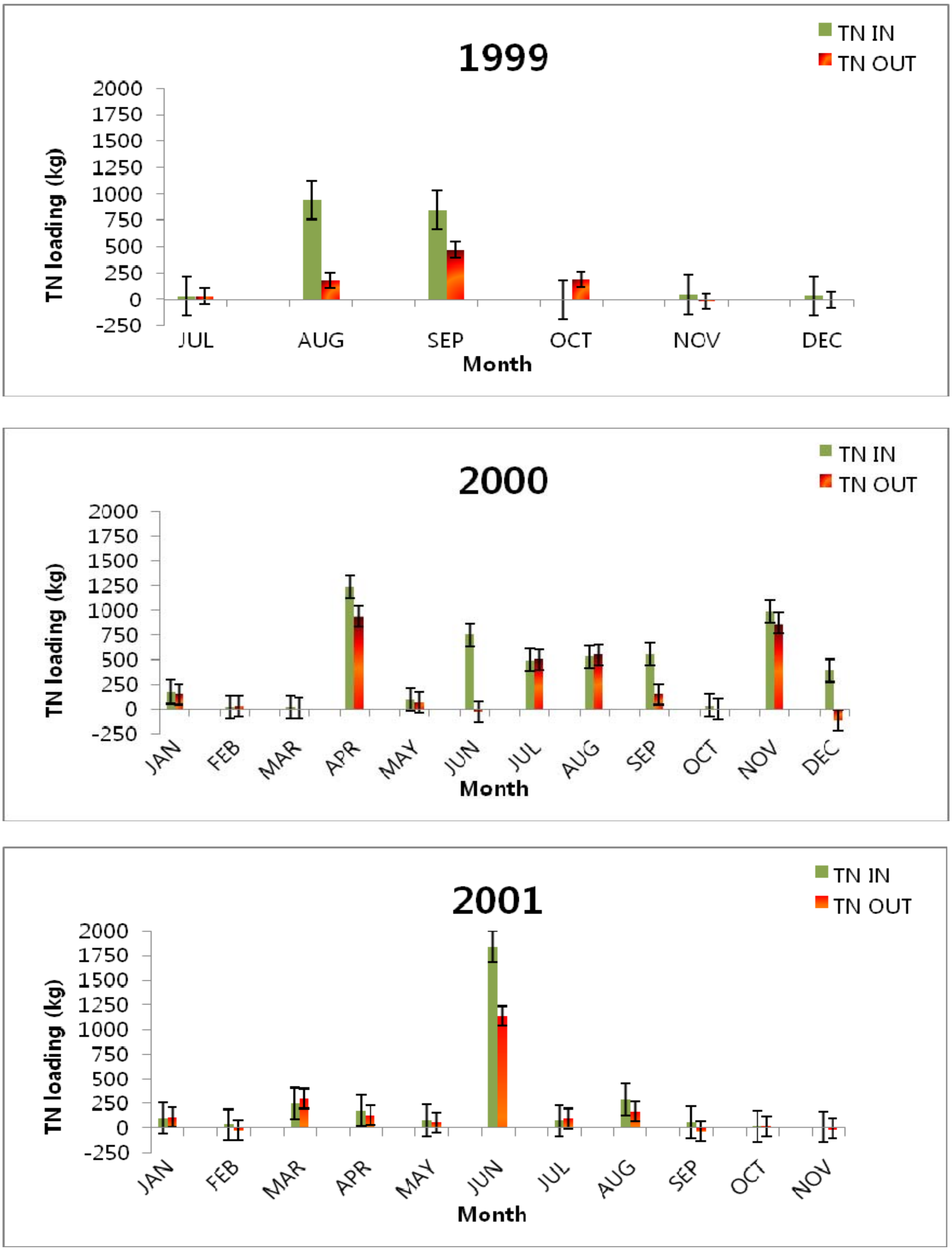


Fig. 3.4(d) Total amount of total nitrogen flowing into and out of the constructed treatment wetland each month from July 1999 until November 2001.

3.3.3. Concentration change of each nitrogen species

Measured concentration changes of ammonium, nitrate, and DON in the inlet during first six days right after the selected storm events are illustrated in Fig. 3.5. Nitrate was the dominant among those nitrogen species in all cases except event 1 and 5. Nitrate concentrations declined in all events except event 1. High concentration of nitrate in event 6 was mainly due to fertilizer spill. Studies of other wetland treatment systems (i.e., Gerke et al., 2001) reported that concentration changes of every nitrogen species were the most rapid in the first cell. However, in our study, there is a long waterway right after the inlet where the water samples were collected, and we could not observe the concentration changes right after water passed through the wetland cell. Concentration change only indicated how each nitrogen species flow into the wetland treatment system consisted of a long waterway in the inlet and another in the outlet, and 16 wetland cells connected with them, and how the concentrations varied during 6 days after storm events.

Gerke et al. (2001) reported that nitrate concentration increased slightly during the first two days of travel time accompanied by decreases in DON, and then declined. This phenomenon was not observed in our study. This difference may be caused by the hydraulic loading rate. Average hydraulic loading rate of Gerke et al.'s study in Kingman, Arizona was 4.1 cm/day, but hydraulic loading rate of our study was dependent on precipitation. In addition, there was a long water way between the inlet where water samples were collected and the first cell in the first block in our study.

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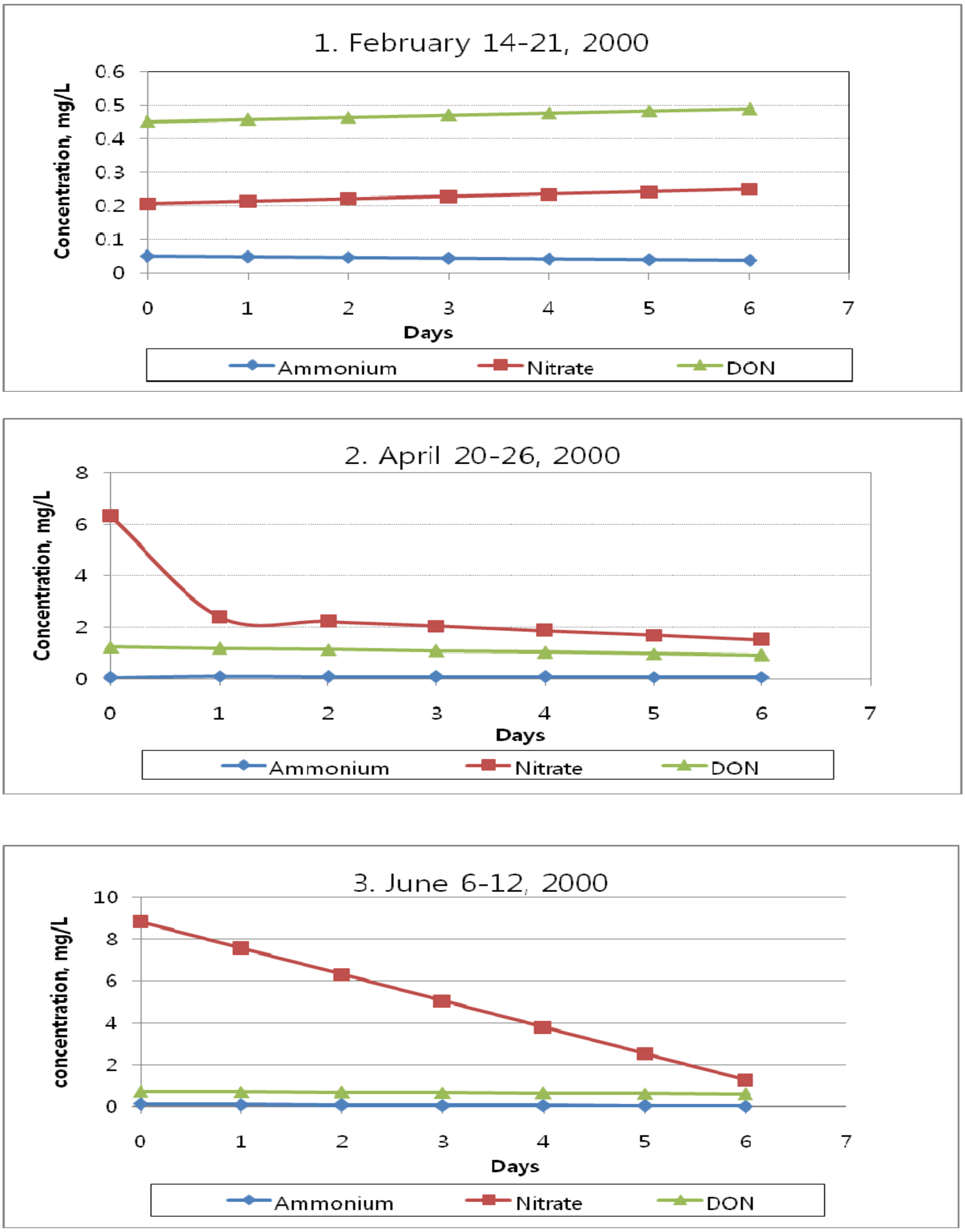
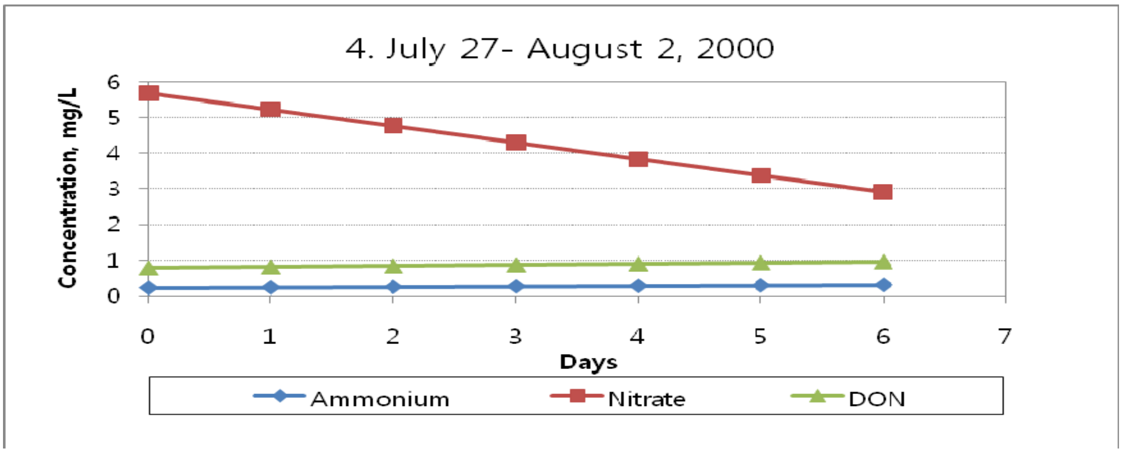
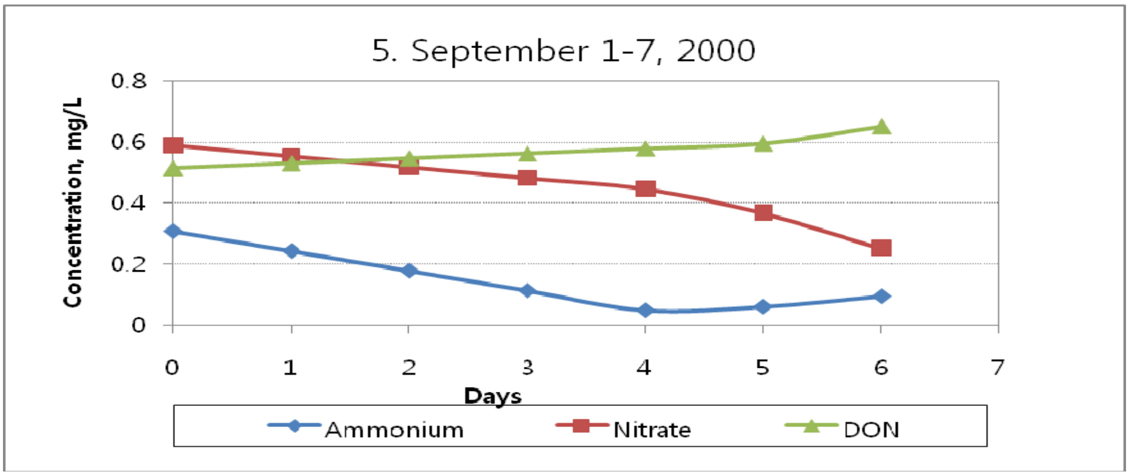


Fig. 3.5(a). Concentrations of ammonium, nitrate, and DON in inlet in events 1, 2, and 3.





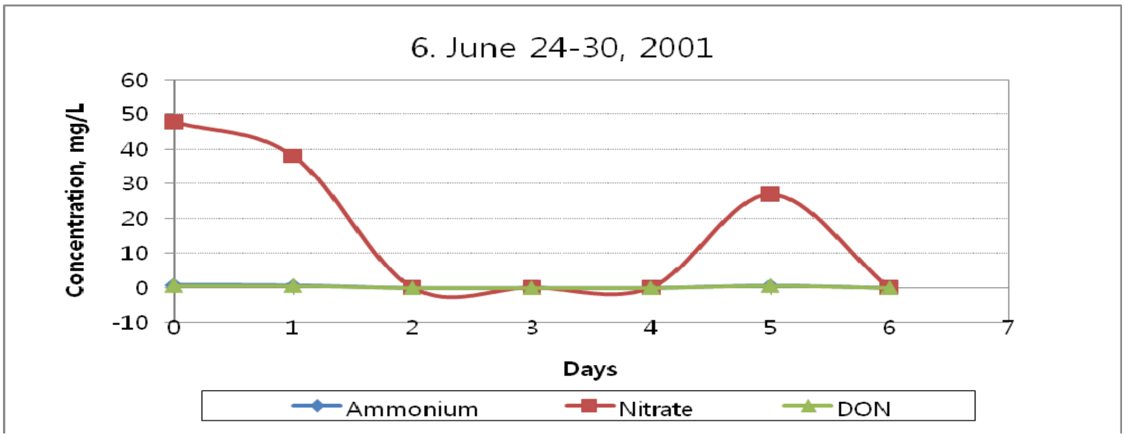


Fig. 3.5(b). Concentrations of ammonium, nitrate, and DON in inlet in events 4, 5, and 6.

3.3.4. Application of sequential nitrogen model

Predicted values of concentration change of ammonium, nitrate, and total nitrogen using sequential nitrogen model were compared with measured values in the inlet of the constructed wetlands, and the results of preliminary statistical analysis were shown in Table 3.6.

Table 3.6. Preliminary statistical analysis between predicted and measured concentrations of each nitrogen species in the inlet of constructed wetlands during selected rainfall events.

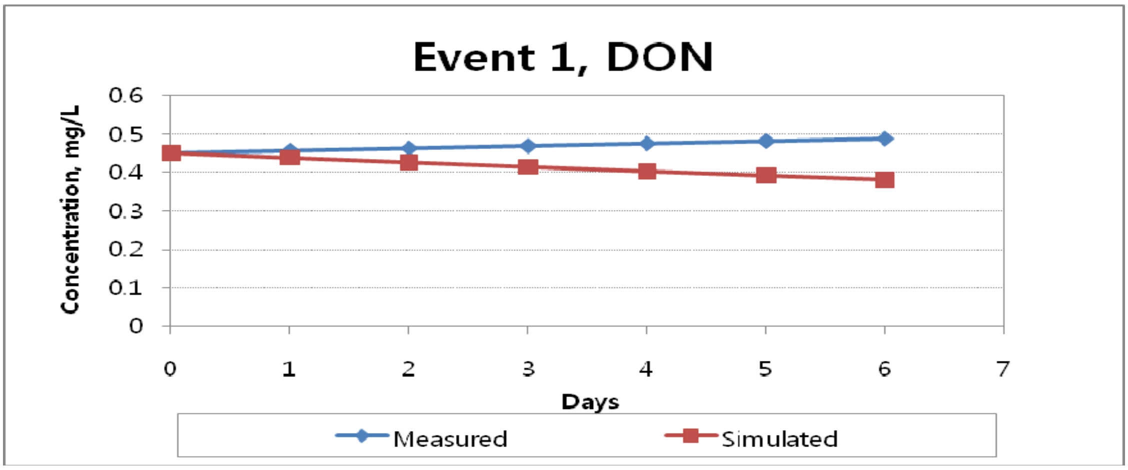
Events	*Dates*	*N-species*	r^2	*E*
		DON	-0.99	-26.71
1	Feb. 14 – Feb. 20, 2000	NH_4	0.99	0.94
		NO_3	0.99	-381.09
		DON	0.99	0.59
2	Apr. 18 – Apr. 24, 2000	NH_4	0.15	-4.64
		NO_3	-0.01	-5.65
		DON	0.99	0.98
3	June 5 – June 11, 2000	NH_4	0.99	-0.26
		NO_3	0.97	-1.37
		DON	-0.99	-8.41
4	July 25 – July 31, 2000	NH_4	-0.99	-8.26
		NO_3	-0.61	-1.19
		DON	-0.96	-6.72
5	Aug. 30 – Sept. 5, 2000	NH_4	0.91	-1.23
		NO_3	-0.98	-16.03
		DON	0.50	-1.04
6	June 24 – June 30, 2000	NH_4	0.48	-1.09
		NO_3	0.30	-1.34

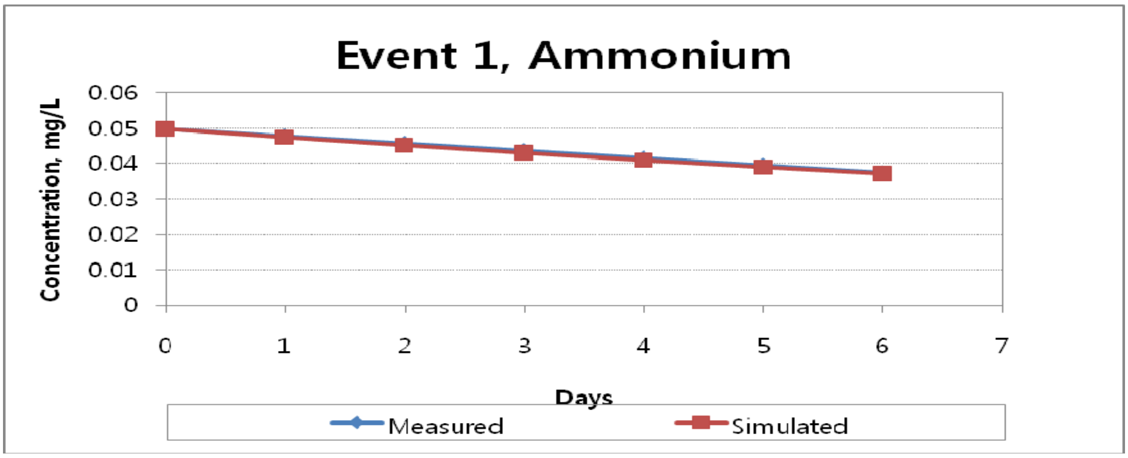
The minus values of r^2 indicated the negative correlation. The absolute value of r^2 ranged from 0.01 to 0.99. E values ranged from -381.09 to 0.98. These results of preliminary statistical analyses showed us that sequential model with average rate constants suggested by Kadlec and Knight (1996) did not adequately predict the concentrations of three nitrogen species in the inlet of the constructed wetland. These results may be resulted from 1) the small number of samples, 2) long waterways right after the inlet, and 3) not-calibrated average rate constants.

These statistical analyses were based on the concentration change of three nitrogen species during the first 6 days right after the storm events on daily basis. Total number of samples was just 7 including the concentrations of first day, and this is too small to get a reliable statistical results. Compared with results from other studies (i.e., Gerke et al., 2001), there is a long waterway in our study site as mentioned previously. Therefore, these concentration changes could not directly reflect the effect of constructed wetlands itself as we intended. Finally, only the average rate constants presented by Kadlec and Knight (1996) were used for simulation in the inlet. Calibration may be required in near future because it is usually site specific.

The model used here was developed to apply to surface flow wetlands with dense emergent vegetation in moderate climates, and was based on the sequential models of nitrogen transformation with various sites (Kadlec and Knights, 1996; Gerke et al., 2001). Jones and Stokes (1993) also developed a model for removal of nitrogen and several other constituents in surface flow wetlands, but it has not been calibrated and tested (Gerke et al., 2001). Wynn and Liehr (2001) developed and calibrated this type of sequential model for a subsurface flow wetland in Maryland. Their simulated concentrations of NH_4, NO_3, and TKN in the effluent were close to actual effluent concentrations (Wynn and Liehr, 2001).

This model is intermediate in both complexity and realism between simple and complex models. This model possibly represents a variety of waste streams ranging from fully nitrified to non-nitrified, or from agricultural runoff to storm water, but not yet calibrated and validated properly in this site. Further testing is needed to validate the model using measured values in the outlet or somewhere in the wetland treatment system.





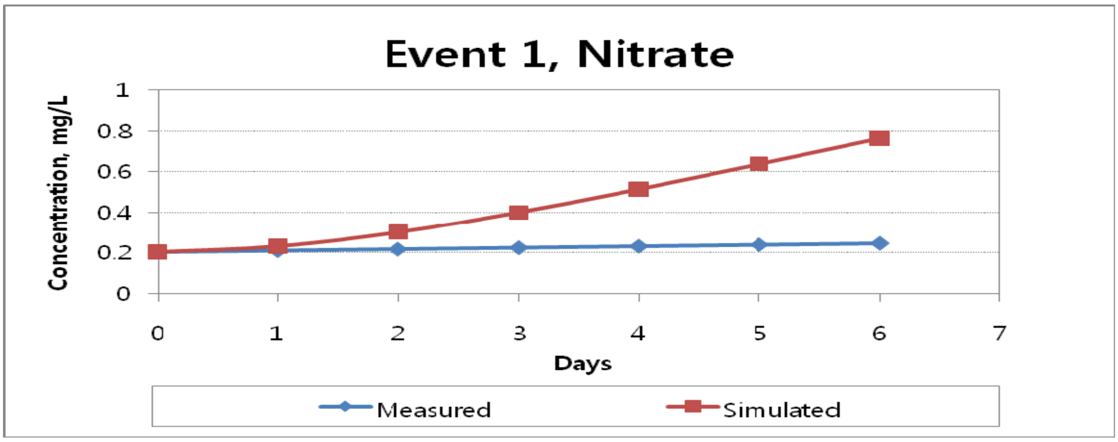
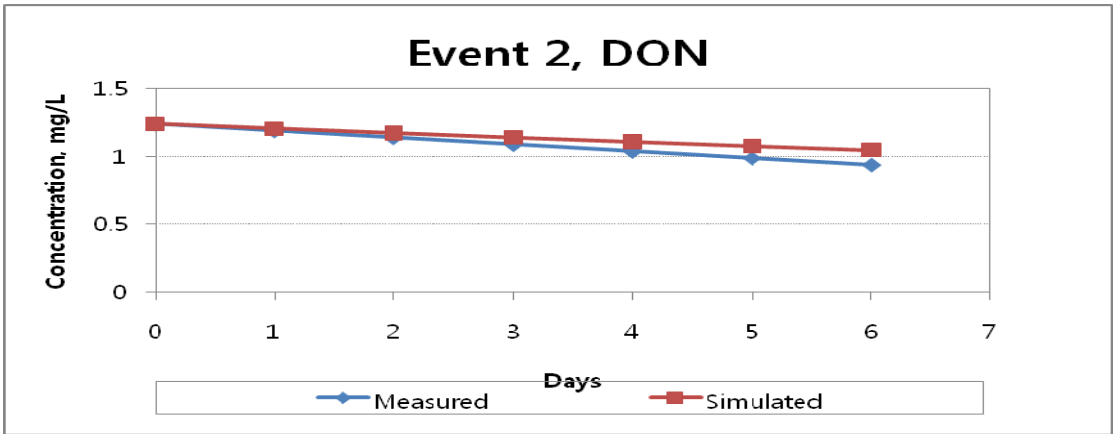
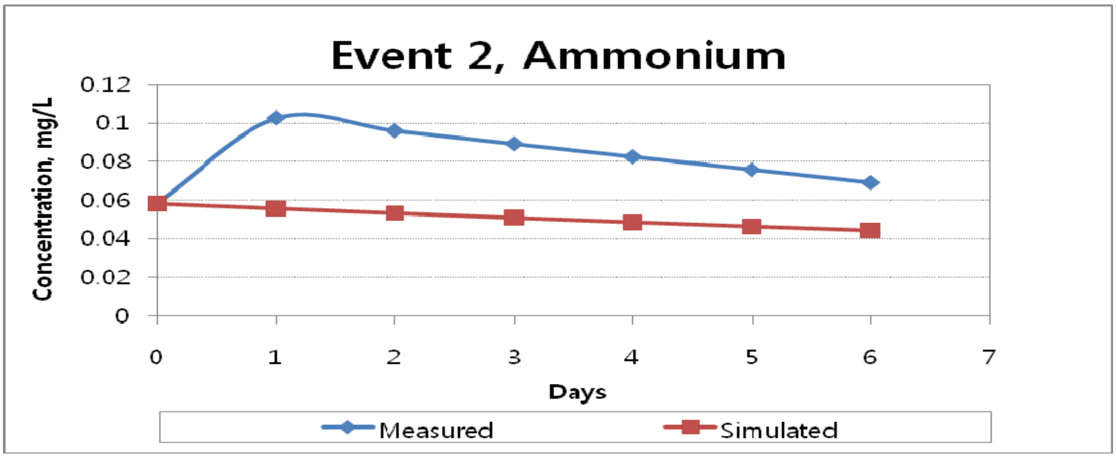


Fig. 3.6(a). Daily changes in nitrogen concentrations (measured and simulated) in water draining from the agricultural watershed for event 1 from February 14 to February 20, 2000.





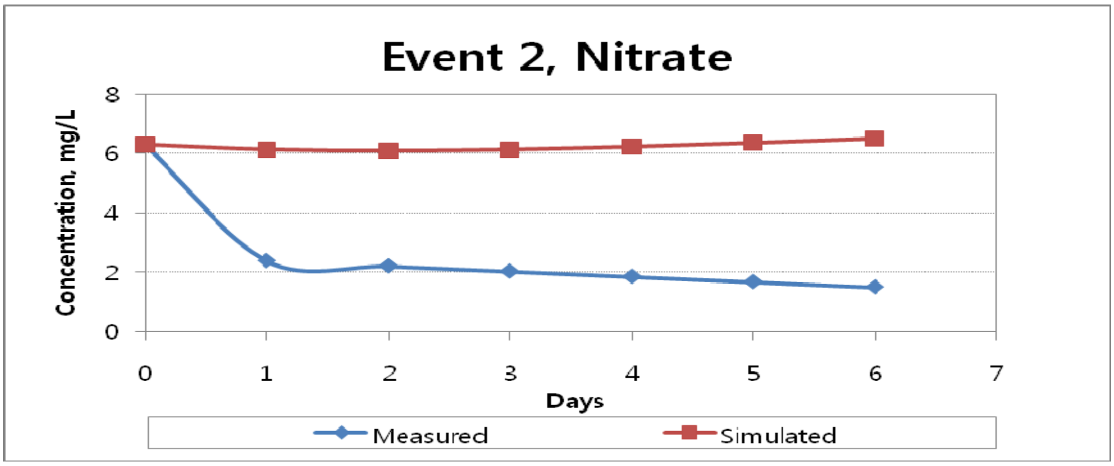


Fig. 3.6(b). Daily changes in nitrogen concentrations (measured and simulated) in water draining from the agricultural watershed for event 2 from April 18 to April 24, 2000.

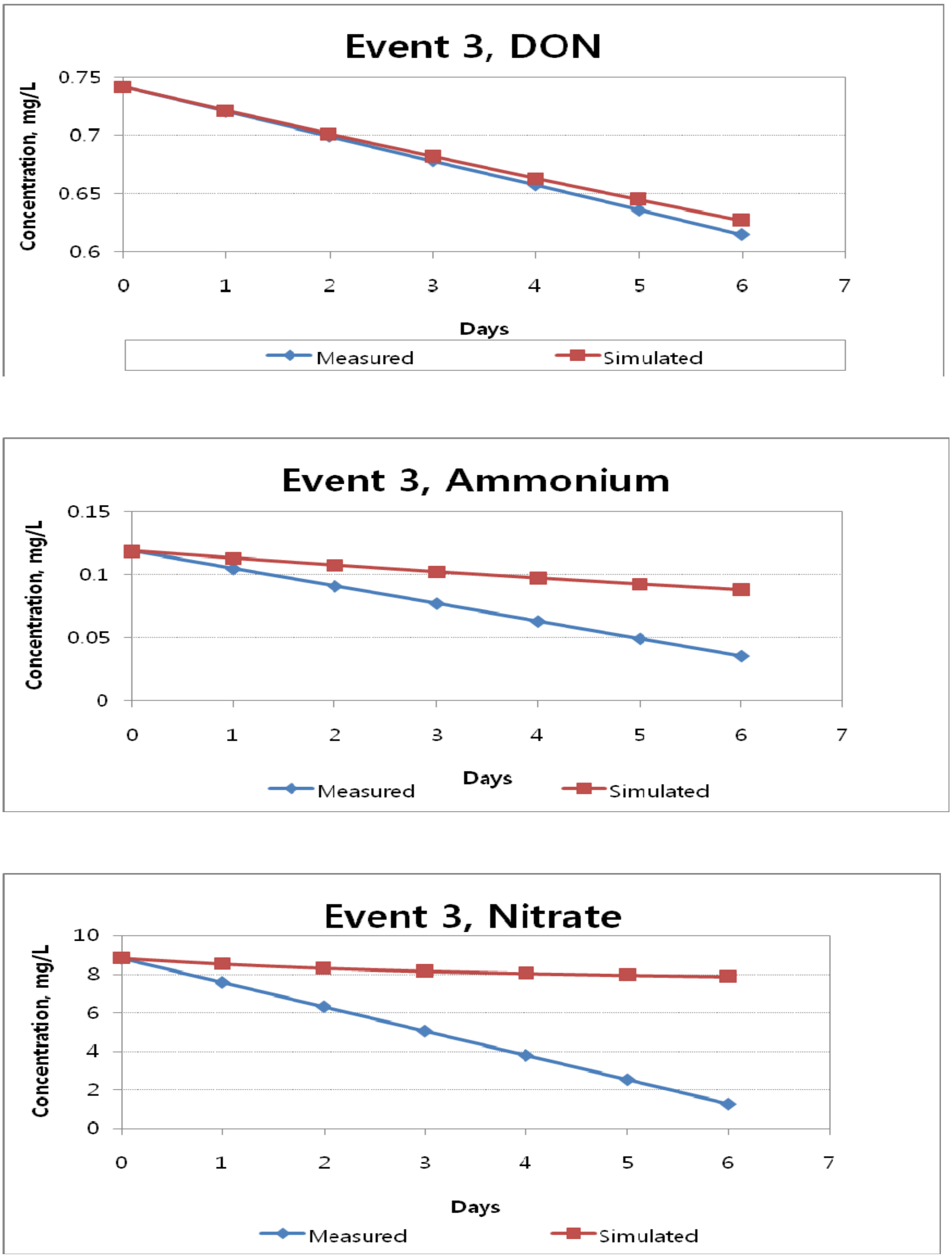


Fig. 3.6(c). Daily changes in nitrogen concentrations (measured and simulated) in water draining from the agricultural watershed for event 3 from June 5 to June 11, 2000.

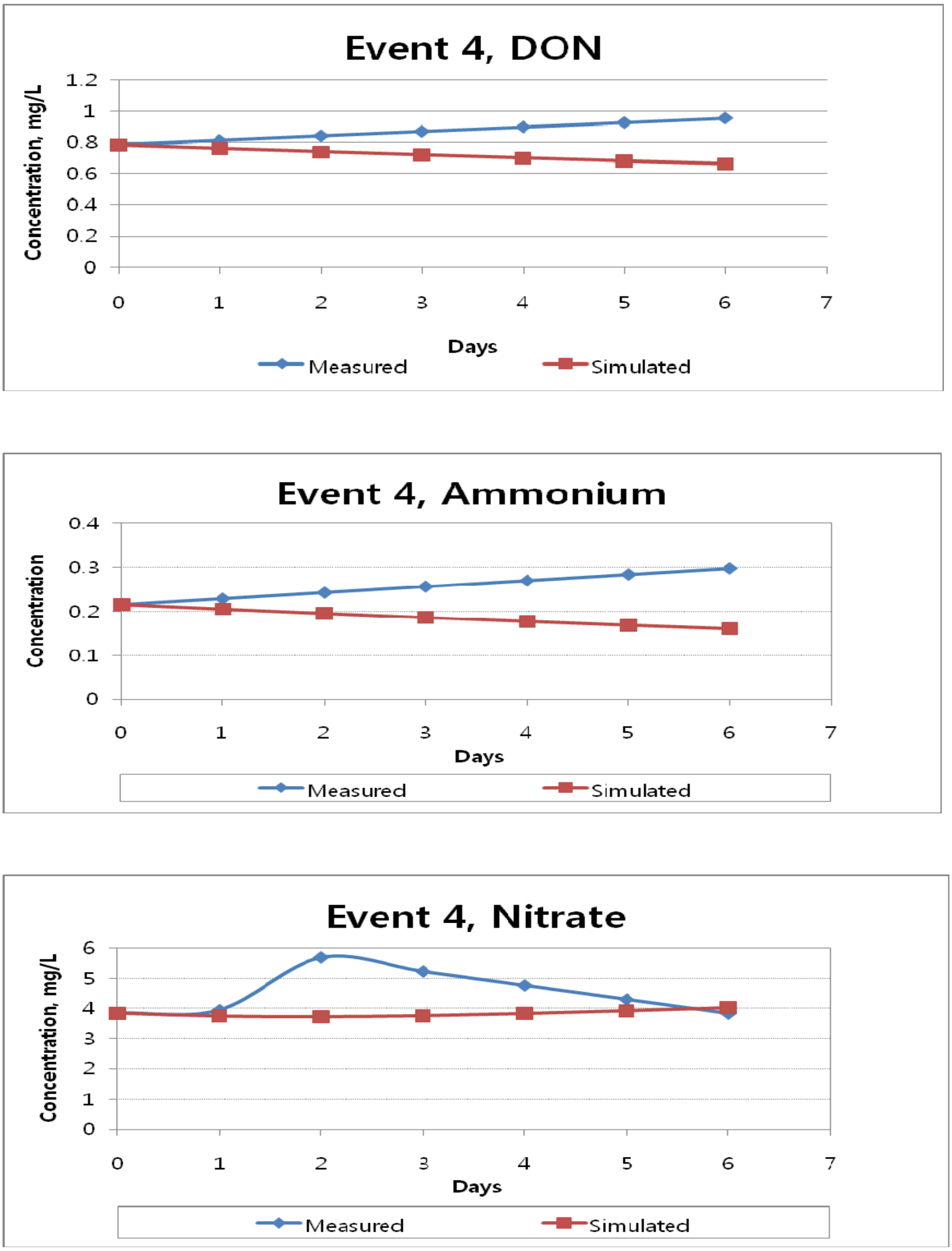
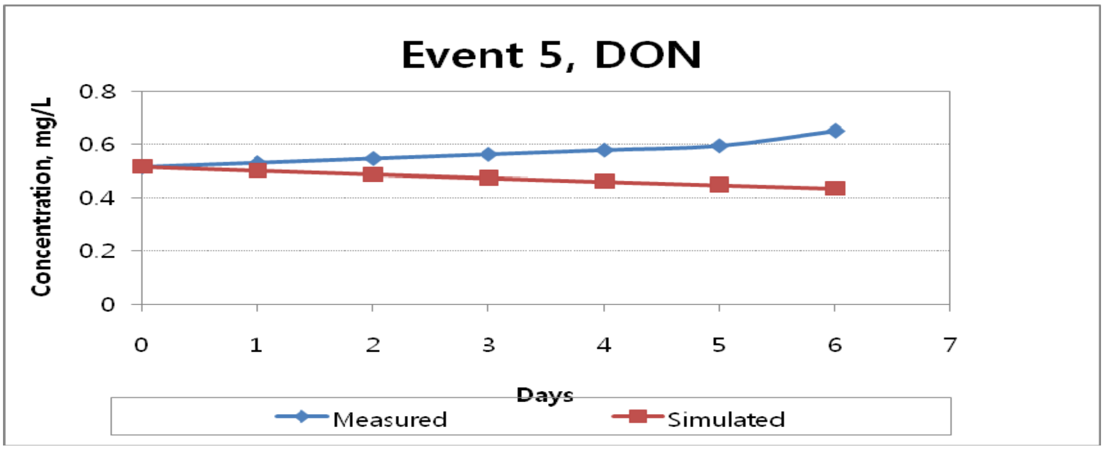
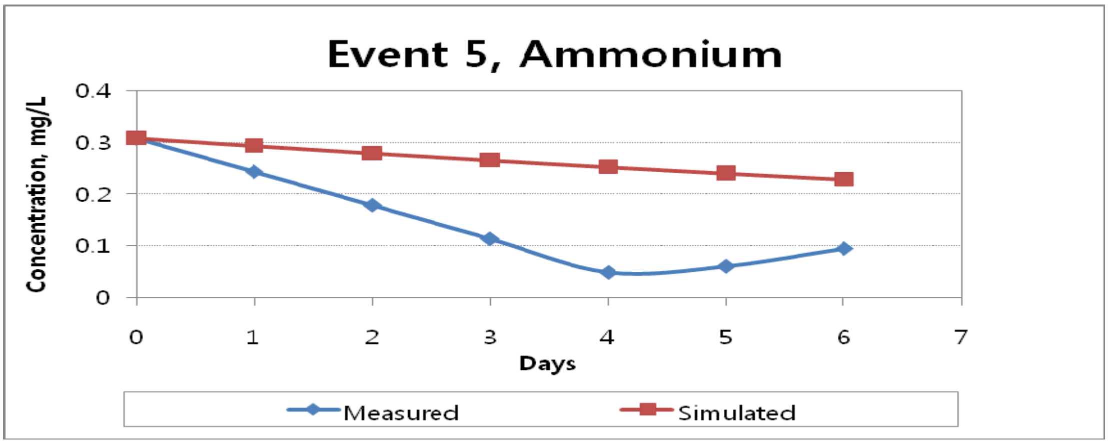


Fig. 3.6(d). Daily changes in nitrogen concentrations (measured and simulated) in water draining from the agricultural watershed for event 4 from July 25 to July 30, 2000.





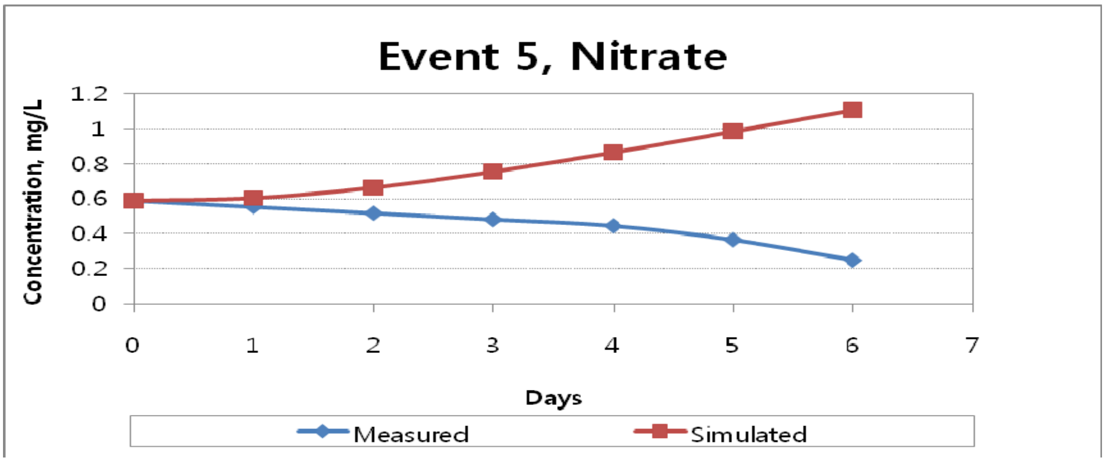
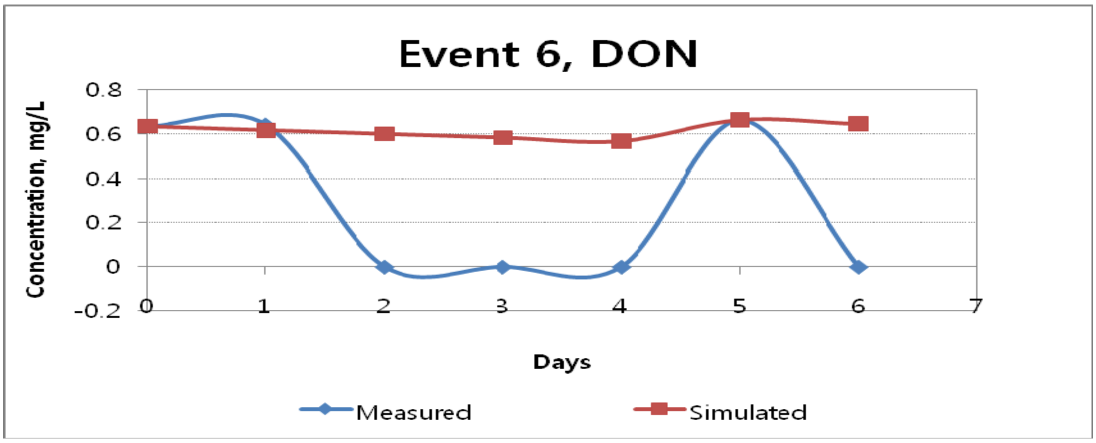
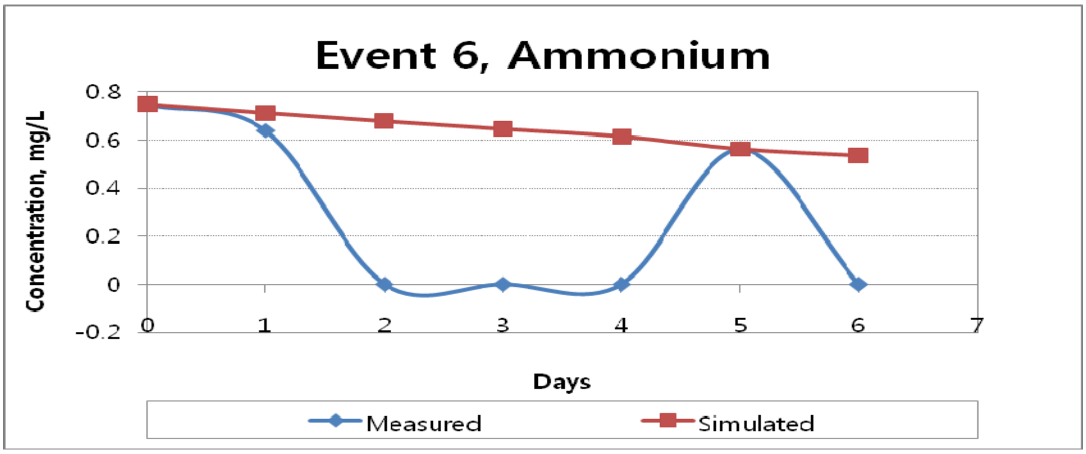


Fig. 3.6(e). Daily changes in nitrogen concentrations (measured and simulated) in water draining from the agricultural watershed for event 5 from August 30 to September 5, 2000.





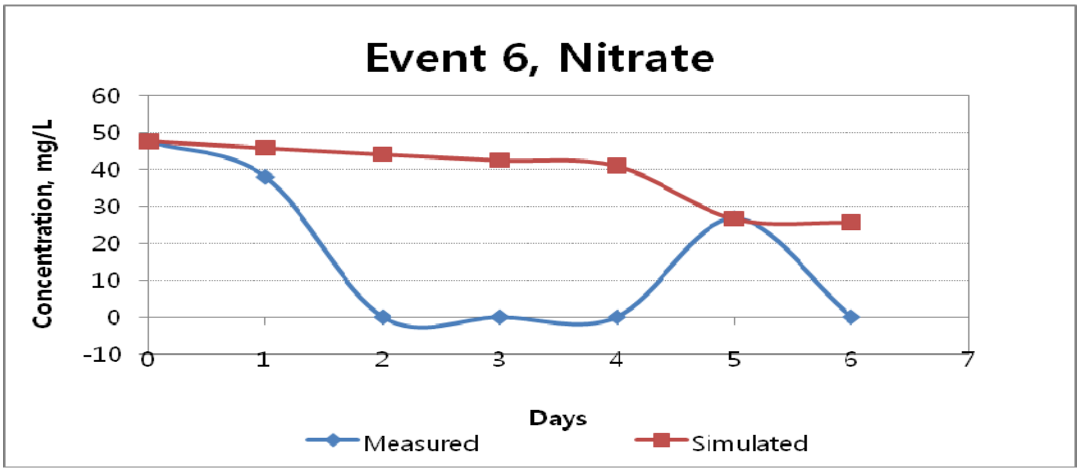


Fig. 3.6(f). Daily changes in nitrogen concentrations (measured and simulated) in water draining from the agricultural watershed for event 6 from June 24 to June 30, 2001.

3.4. Conclusion

The growth of three wetland plants including *Juncus roemerianus*, *Cladium jamaicense*, and *Spartina alterniflora* and nitrogen removal efficiency of a constructed tidal wetland system were monitored in Open Grounds Farm, Carteret County, NC. Two of the three plant species, *J. roemerianus* and *C. jamaicense*, grew rapidly during 30 months of study periods (July 1999 – December 2001). Good nitrogen removal efficiencies for this wetland treatment system were observed in almost all nitrogen species. Sequential nitrogen model proposed by Gerke et al. (2001) was applied and tested to predict the concentration changes of dissolved organic nitrogen, ammonium, and nitrate in the inlet only in selected periods due to the limitation of reliable measured flow data. The average rate constants with normalized to depth proposed by Kadlec and Knight (1996) were used for prediction. The results of statistical analysis indicated that the sequential nitrogen model did not predict the concentrations of three nitrogen species very well due to the small number of sample data, a long waterway right before the wetland cells, and non-calibrated average rate constants in this study. The prediction results can be improved with proper calibration in near future.

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APPENDICES

APPENDIX A. Part of DRAINMOD Rainfall data file – cart9901.rai

```
999991 1999 1  1 1   0 1 2   0 1 3   0 1 4   0 1 5   0 1 6   0 1 7   0 1 8   1 1 9   0 110   0 111   0 112   0
999991 1999 1  113   0 114   0 115   0 116   0 117   0 118   0 119   0 120   0 121   0 122   0 123   0 124   0
999991 1999 1  2 1   0 2 2   0 2 3   0 2 4   0 2 5   0 2 6   0 2 7   0 2 8   0 2 9   0 210   0 211   0 212   0
999991 1999 1  213   0 214   0 215   0 216   0 217   0 218   0 219   0 220   0 221   0 222   0 223   0 224   0
999991 1999 1  3 1   0 3 2   0 3 3   0 3 4   2 3 5   8 3 6  34 3 7  51 3 8   2 3 9   0 310   0 311   0 312   0
999991 1999 1  313   0 314   0 315   0 316   0 317   0 318   0 319   0 320   0 321   0 322   0 323   0 324   0
999991 1999 1  4 1   0 4 2   0 4 3   0 4 4   1 4 5   0 4 6   0 4 7   0 4 8   0 4 9   0 410   0 411   0 412   0
999991 1999 1  413   0 414   0 415   0 416   0 417   0 418   0 419   0 420   0 421   0 422   0 423   0 424   0
999991 1999 1  5 1   0 5 2   0 5 3   0 5 4   0 5 5   0 5 6   0 5 7   0 5 8   0 5 9   0 510   0 511   0 512   0
999991 1999 1  513   0 514   0 515   0 516   0 517   0 518   0 519   0 520   0 521   0 522   0 523   0 524   0
999991 1999 1  6 1   0 6 2   0 6 3   0 6 4   0 6 5   0 6 6   0 6 7   0 6 8   0 6 9   0 610   0 611   0 612   0
999991 1999 1  613   0 614   0 615   0 616   0 617   0 618   0 619   0 620   0 621   0 622   0 623   0 624   0
999991 1999 1  7 1   0 7 2   0 7 3   0 7 4   0 7 5   0 7 6   0 7 7   0 7 8   0 7 9   0 710   0 711   0 712   0
999991 1999 1  713   0 714   0 715   0 716   0 717   0 718   0 719   0 720   0 721   0 722   0 723   0 724   0
999991 1999 1  8 1   0 8 2   0 8 3   0 8 4   0 8 5   0 8 6   0 8 7   0 8 8   0 8 9   0 810   0 811   0 812   0
999991 1999 1  813   0 814   0 815   0 816   0 817   0 818   0 819   0 820   0 821   0 822   0 823   0 824   0
999991 1999 1  9 1   0 9 2   0 9 3   0 9 4   0 9 5   0 9 6   2 9 7  12 9 8   5 9 9   3 910   0 911   0 912   0
999991 1999 1  913   0 914   0 915   0 916   0 917   0 918   0 919   0 920   0 921   0 922   0 923   0 924   0
999991 1999 1 10 1   010 2   210 3   510 4   210 5   010 6   010 7   010 8   010 9   01010   01011   01012   0
999991 1999 1 1013   01014   01015   01016   01017   01018   01019   01020   01021   01022   01023   01024   0
999991 1999 1 11 1   011 2   011 3   011 4   011 5   011 6   011 7   011 8   011 9   01110   01111   01112   0
999991 1999 1 1113   01114   01115   01116   01117   01118   01119   01120   01121   01122   01123   01124   0
999991 1999 1 12 1   012 2   012 3   012 4   012 5   012 6   012 7   012 8   012 9   01210   01211   01212   0
999991 1999 1 1213   01214   01215   01216   01217   01218   01219   01220   01221   01222   01223   01224   0
999991 1999 1 13 1   013 2   013 3   013 4   013 5   013 6   013 7   013 8   013 9   01310   01311   01312   0
999991 1999 1 1313   01314   01315   01316   01317   01318   01319   01320   01321   01322   01323   01324   0
999991 1999 1 14 1   014 2   014 3   014 4   014 5   014 6   014 7   014 8   014 9   01410   01411   91412   2
999991 1999 1 1413   01414   01415   01416   01417   11418   41419   11420   11421   01422   01423   01424   0
999991 1999 1 15 1   015 2   015 3   015 4   015 5   015 6   015 7  3115 8  1915 9   01510   01511   01512   0
999991 1999 1 1513   01514   01515   01516   01517   01518   01519   01520   01521   01522   01523   01524   0
999991 1999 1 16 1   016 2   016 3   016 4   016 5   016 6   016 7   016 8   016 9   01610   01611   01612   0
999991 1999 1 1613   01614   01615   01616   01617   01618   01619   01620   01621   01622   01623   01624   0
999991 1999 1 17 1   017 2   017 3   017 4   017 5   017 6   017 7   017 8   017 9   01710   01711   01712   0
999991 1999 1 1713   01714   01715   01716   01717   01718   01719   01720   01721   01722   01723   01724   0
999991 1999 1 18 1   018 2   018 3   018 4   018 5   018 6   218 7   018 8   018 9   11810   01811   01812   0
999991 1999 1 1813   01814   01815   01816   01817   01818   01819   01820   01821   01822   01823   01824   0
999991 1999 1 19 1   019 2   019 3   019 4   019 5   019 6   019 7   019 8   019 9   01910   01911   01912   0
```

APPENDIX B. DRAINMOD PET data file – cart9901.pet

```
999999 1999 1     4   3   7   4   3   4   5   5   5   4   5   6   5   2
  7   5   6   4   6   6   6   8   8   1   6   6   7  10   6   5   5
999999 1999 2     4   4   6   6   8  10   9   9   7  11  10   6   9   7
  7  10  10   5   3   7   5   7   3   4   4   9   9   5
999999 1999 3    11  10   9  10   8   9  12   9   6  10  11  12   8   4
  5  12  15  15  12  11   2  13  14  14  11   2   3  16  14  15  13
999999 1999 4     9  11  16  15  15  12  16  16  15  10   3  16  16  18
 13  19  12  16  17  14  15  19  17  15  18  15  17   4   3   4
999999 1999 5     2   3   9  13  17  10  22  20  21  20  17   9   9   5
  3   6  23  20  17  19  21  21  20  15  22  15  19  22  23  23  20
999999 1999 6    21  23  16  17  22  18  22  21  22  20  16   8  19  24
  6   7   5   9   5   8  15   5  17  19  21  18  23  19  18  11
999999 1999 7    24  21  23  22  23  24  23  18  24  25   7   8   5  16
  8  20  21  20  23  20  12  19  21  18  16  23  13  21  19  23  21
999999 1999 8    18   9   6   3  18  18  19  17   5  19  11  16  18  17
  9  14  18  18  15   7  13  15  18  15  12  15  15  13   7   4  11
999999 1999 9     9   4   4   0  17   4   7  13   7  11  18  15  11  11
  2  15  17  16  12   7   5   7  15   9   8  10   6   5  10  15
999999 199910    14  14  15   7   6  13  14  13  13  10   4  11   7  13
 12   4   1  13  11   5   1  12  11   9   9  10  12  10  10   7   5
999999 199911    11   3  11   8   9  12  10   9  10  12  11   9   8  10
 10   9   8   8   9   5   6   7   7   7   6   5   4   7   4   8
999999 199912     8   7   7   9   8   6   7   6   7   5   8   5   1   7
  7   6   5   1   1   5   2   0   5   5   6   7   7   5   7   8   6
999999 2000 1     5   6   7   7   5   5   1   5   3   1   7   5   9   5
  3   8   7   1   4   6   5   3   0   0   1   4   3   1   4   1   5
999999 2000 2     5   5   7   6   5   5   8   7   1  10   8   1   3   3
  7   7   8   4  11   8   6   8   6   8  11  11   8  10  10
999999 2000 3    12  12   6   1  11  12  12  15  11  13  11  11  10  11
 13  11   8   9  11   5   8   2   8  13  14  16  11  16  15   4  14
999999 2000 4    15  14  13  11  14  16  17  15  15  16  16  13   2   4
  3  11  14   2   6  17  15   5  16  13   3   7   7   2  15  18
999999 2000 5    19  12  19  19  19  20  20  19  20  16  19  20  21  20
 18  16  13  19  18  22  19   8  15  18  16  19  18  12   5   4  15
999999 2000 6    20  19  20   7   6  15  16  18  20  21  22  22  19  21
 20  22  20  20  19  11  22  14  19  18   8  20  23  21  11  18
999999 2000 7    19  21  20  20  19   6  16  19   8  23  18  12  10  16
  6  19  21  19  21  17   8  14   6   5   5   9  12  14  12  16  12
999999 2000 8    21  19  14   3  10  19  19  19  20  16  14  16  14  20
 18  20  21  15  14  14  18  15  16  20   7  18  14  12  11  18  11
999999 2000 9    14  14  16   8   2   3  15  17  13  17  17  18  18  12
 13  17  12   2  14  15   9   8   1  14  11   7  10  12  13  10
999999 200010     2  11  12  14  13  14   9   0   8   9  10  10  11  10
 10  11   7   8  10   8  10  11   9  10  11   6   8  10   8   8   8
999999 200011     8   9   8   6   3   3   2   3   3  10   8   2   6   0
  4   4   0   2   0   2   3   4   3   4   1   3   6   6   4   6
999999 200012     5   3   1   4   5   5   5   5   4   3   2   6   4   5
  4   2   8   5   2   5   4   6   3   4   4   3   2   2   4   4   3
999999 2001 1     3   4   3   3   4   5   4   1   2   6   6   3   7   1
  6   6   2   3   5   1   5   6   4   5   4   5   8   7   6   2   8
999999 2001 2     4   0   6   2   7   7   8   8   6   2   9   2   2   4
  8  10   5   6  -2   9   8   0   8   8   5  13   9   1
999999 2001 3     8   6   1   3   4   7   7   9   1   8  11   3  12  11
  1   7  11   7   9   5   7  10  12  15   2   7  10  12   3  15  12
999999 2001 4     3  14   3  10  14  15  18  17  17  18  15  14  12  14
  8  17   4  10  14  13  16  17  18  18   2  12  16  18  18  17
999999 2001 5    18  18  19  19  20   8  10  19  17  20  19  20  17  18
```

5 14 2 17 24 18 13 15 8 19 21 10 21 6 4 19 19
999999 2001 6 20 20 23 22 21 19 15 6 16 18 20 21 22 5
9 18 9 22 15 19 20 20 14 10 23 15 20 20 20 23
999999 2001 7 23 7 13 18 9 6 21 7 18 16 15 15 8 20
20 18 20 20 7 17 20 15 14 17 21 12 6 9 10 9 22
999999 2001 8 19 18 20 20 18 13 21 21 21 19 15 15 19 2
17 18 20 15 19 5 11 17 19 7 16 15 18 12 8 7 8
999999 2001 9 7 16 12 6 12 15 15 15 16 15 14 16 10 9
12 15 13 13 10 4 11 11 10 9 6 9 14 12 12 11
999999 200110 12 12 13 13 12 10 11 11 11 11 12 10 10 10
12 10 10 9 10 7 9 10 9 11 10 11 8 8 8 9 8
999999 200111 3 6 9 9 8 8 8 8 8 8 8 8 7 7
5 6 7 8 5 4 7 6 1 4 1 5 3 7 4 7
999999 200112 3 1 6 6 6 6 6 5 5 5 1 3 1 1
6 5 4 8 5 6 5 4 5 4 2 3 4 5 4 4 4

APPENDIX C. DRAINWAT input file 1 – ogf31.f01

```
Subwatershed Modeling with 8 fields at Open Grounds Farm, Carteret Co.,
NC - 1999-2001
2
input/ogf31base.f11
input/ogf31.f16
input/junkres.f12
input/avlobpet.prn
'1999',0
3,1,0
1
1
0,0
5
365
```

APPENDIX D. DRAINWAT input file 2 – ogf31.f11

```
*** Job Title ***
Simulation of subwatershed at Open Grounds Farm, Cateret County, NC
8 fields, 3 SCS soils, ditchcanal modeling, 1999-2001.
*** Printout Control ***
 1 2
*** Climate ***
input/d1base.rai
input/car9901.pet
999991 999999 1999  1 2001 12 3547  75
 1.00 1.00 1.00 1.00 1.00 1.00 1.00 1.00 1.00 1.00 1.00 1.00
8,3
input/belh100.dat   01 01  4.65  0.00  240.0
000.0    0129.50 input/iuhdata.dat   1 0 0 0 0 2 4003.0
input/ponz100.dat   02 02  4.65  1.50  240.0
000.0    0129.50 input/iuhdata.dat   1 0 0 0 0 2 4003.0
input/wasd100.dat   03 03  4.65  0.00  240.0
000.0    0129.50 input/iuhdata.dat   1 0 0 0 0 2 4003.0
input/ponz100.dat   04 02  4.65  0.00  240.0
000.0    0129.50 input/iuhdata.dat   1 0 0 0 0 2 4003.0
input/belh100.dat   05 01  4.65  0.00  240.0
000.0    0129.50 input/iuhdata.dat   1 0 0 0 0 2 4003.0
input/belh100.dat   06 01  4.65  0.00  240.0
000.0    0129.50 input/iuhdata.dat   1 0 0 0 0 2 4003.0
input/belh100.dat   07 01  4.65  0.00  240.0
000.0    0129.50 input/iuhdata.dat   1 0 0 0 0 2 4003.0
input/belh100.dat   08 01  4.65  0.00  240.0
000.0    0129.50 input/iuhdata.dat   1 0 0 0 0 2 4003.0
```

APPENDIX E. DRAINWAT input file 3 – ogf31.f16

```
031
1,0
1,0
1,0
3,2
1,0
3,1
1,0
3,5
1,0
1,0
1,0
3,6
1,0
1,0
1,0
1,0
3,7
1,0
1,0
1,0
1,0
1,0
1,0
3,3
3,4
1,0
1,0
1,0
3,8
1,0
1,0
0,1.0,5.0,1,0.1,-10
1.0,1.0
0,0.005,0.01,5,0
0.00,0.00,0.0,0.0,50
1 ,   33,0.003,0.040, 8.0, 0.58, 2.00, 0.01,0.0, 0, 0.,0.,0,2.50, 92.9,
0,8.0
2 ,   66,0.007,0.040, 8.0, 0.58, 2.00, 0.01,0.0, 0, 0.,0.,0,2.50, 92.9,
0,8.0
3 ,   33,0.003,0.040, 7.0, 0.58, 2.00, 0.01,0.0, 0, 0.,0.,0,2.50, 92.9,
0,8.0
4 , 2608,0.261,0.040, 6.0, 0.58, 2.00, 0.01,0.0, 0, 0.,0.,0,2.50, 92.9,
0,8.0
5 , 2640,0.264,0.040, 6.0, 0.58, 2.00, 0.01,0.0, 0, 0.,0.,0,2.50, 92.9,
0,8.0
6 , 5248,0.525,0.040, 6.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
7 ,   66,0.007,0.040, 6.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
8 , 2608,0.261,0.040, 5.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
```

```
9 , 5215,0.522,0.040, 5.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
10, 5215,0.522,0.040, 5.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
11, 2608,0.261,0.040, 5.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
12, 2608,0.261,0.040, 5.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
13,   66,0.007,0.040, 7.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
14, 2608,0.261,0.040, 6.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
15, 5248,0.525,0.040, 6.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
16, 5248,0.525,0.040, 6.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
17,   66,0.007,0.040, 6.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
18, 2608,0.261,0.040, 5.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
19, 2608,0.261,0.040, 5.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
20, 2640,0.264,0.040, 5.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
21, 2640,0.264,0.040, 5.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
22, 2640,0.264,0.040, 5.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
23, 2640,0.264,0.040, 5.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
24, 2640,0.264,0.040, 6.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
25, 5248,0.525,0.040, 6.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
26,   66,0.007,0.040, 6.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
27, 2608,0.261,0.040, 5.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
28, 5215,0.522,0.040, 5.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
29, 5215,0.522,0.040, 5.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
30, 2608,0.261,0.040, 5.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
31, 2608,0.261,0.040, 5.0, 0.58, 2.00, 0.01,0.0, 0.0, 0.,0.,0,2.50,
92.9, 0,8.0
32
1   2   1   0   0   1
2   0   2   1   0   1
4   7   4   3   2   0
7   0   5   4   0   1
9   0   6   5   0   1
11  0   7   6   0   1
13  7   9   8   7   0
16  0  10   9   0   1
```

```
18  1  10   0   0   0
19  0  11   8   0   1
21  0  12  11   0   1
23  1  12   0   0   0
24  0  13   3   0   1
26  7  15  14  13   0
29  0  16  15   0   1
31  0  17  16   0   1
33  7  19  18  17   0
36  0  20  19   0   1
38  0  21  20   0   1
40  1  21   0   0   0
41  0  22  18   0   1
43  0  23  22   0   1
45  1  23   0   0   0
46  0  24  14   0   1
48  0  25  24   0   1
50  0  26  25   0   1
52  7  28  27  26   0
55  0  29  28   0   1
57  1  29   0   0   0
58  0  30  27   0   1
60  0  31  30   0   1
62  1  31   0   0   0
 1, 3, 5, 7,
9,11,13,15,17,19,21,23,25,27,29,31,33,35,37,39,41,43,45,47,49,51,53,55,
57,59,61
1
1,3,0.10
0.0,1.3,0.2
31,30,29,28,27,26,25,24,23,22,21,20,19,18,17,16,15,14,13,12,11,10,9,8,7
,6,5,4,3,2,1
32,31,30,29,28,27,26,25,24,23,22,21,20,19,18,17,16,15,14,13,12,11,10,9,
8,7,6,5,4,3,2,1
9
25,12
39,12
40,12
42,12
45,12
47,12
50,12
51,12
52,12
```

APPENDIX F. DRAINWAT input file (soil and crop data included) – parbelh.dat

```
*** Job Title ***
BELHAVEN, 22.50
CONVENTIONAL DRAINAGE
*** Printout and Input Control ***
 3 0 0
*** Climate ***
wilson.hou     WILSON.TEM     319476 319476 1950  1 1979 12 3547  75
 1.00 1.00 1.00 1.00 1.00 1.00 1.00 1.00 1.00 1.00 1.00 1.00
*** Drainage System Design ***
 1
     90.00    175.25  10000.00      0.50      2.50      0.25     3.79
50.00
      50      0.10
 0140 0140 0140 0140 0140 0140 0140 0140 0140 0140 0140 0140
*** Soils ***
    270.00    0.0000
  30.20.00  80. 1.00 270. 0.01
99
7 9
   0.73000       0.0
   0.71000     -10.0
   0.69000     -25.0
   0.67000     -50.0
   0.66000    -100.0
   0.65000    -200.0
   0.45000  -15000.0
    0.0000    0.0000    0.0600
   15.0000    0.5000    0.0400
   30.0000    1.3000    0.0162
   60.0000    2.0000    0.0079
   90.0000    2.7000    0.0017
  120.0000    3.3000    0.0009
  150.0000    4.2000    0.0001
  200.0000    6.0000    0.0000
 1000.0000  100.0000    0.0000
 5
      0.00      0.00      0.00
     25.00      1.70     18.30
     50.00      1.90     11.00
    150.00      2.50      3.70
    500.00      3.00      3.70
*** Trafficability ***
 3171226 820              2.0       1.2       1.0
12311231 820              2.0       1.2       1.0
*** Crop ***
 0.480
 3151220       30.00
 410 818
13
 1 1  3.0  326  3.0  420  6.0  430 15.0  531 25.0  622 30.0  9 2 30.0
912 10.0
 927  3.0 10 7  3.0 1027 10.0 1126 15.0 1231 15.0
*** Wastewater Irrigation ***
```

```
  0    0 0 368   1  6
  0 0       0 0       0 0       0 0
    7.00000   1.00000 1.00 1.00 1.00 1.00 1.00 1.00 1.00 1.00 1.00 1.00
 1.00 0.00
```