Nutrient Sources and Dynamics in the Parafield Stormwater Harvesting Facility and Implication to Water Quality Control

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1 Introduction

Stormwater is in dry countries like Australia considered as an alternative water resource, which can provide environmental, economic and social benefits to urban areas (Scott 1996; Wong et al. 1999; Environment Australia 2002). In urbanised areas around 90% of the stormwater runoff is directed into drainage systems to lead them to the nearest natural water body (Scott 1996; Wong et al. 1999; Environment Australia 2002), which was based on the management objective in form of flood mitigation (Environment Australia 2002). Estimation in 1989 of the amount of stormwater runoff in Adelaide was a total of 150,000 ML, which almost equalled the amount of Adelaide's water use (190,000 ML) (Fisher and Clark 1989; Environment Australia 2002). In 1991 estimation of the amount of total stormwater runoff deriving from the metropolitan area of Adelaide was around 210,000 ML, which is a greater volume than the total annual mains water use (Environmental Consulting Australia 1991). The identification of uncontrolled urban stormwater as mayor non-point source (NPS) pollution of surface waters (Herricks and Jenkins 1995; IWA 2000) caused a change in the strategy of stormwater management, which is currently focusing on the source control of pollutants in runoff, in-stream controls and treatment processes, measures to prevent scouring, erosion, sedimentation, enhanced infiltration and detention to reduce peak flows, design of drainage corridors and urban waterways for aesthetic, ecological and recreational purposes, water quality monitoring, measures to promote water conservation, including stormwater re-use and community education and participitation (Scott 1996; Environment Australia 2002). Therefore appropriate stormwater management is essential to consider stormwater as an alternative water resource to ensure water supply (Environment Australia 2002).

This research was carried out at a constructed wetland system located beside the Parafield Airport, SA, and was built in 1999. It belongs to a series of wetland system in order to reduce pollutants in the stormwater runoffs before it enters the Barker inlet of Gulf St. Vincent, which was degraded by uncontrolled stormwater runoffs (City of Salisbury 2003). The Parafield Stormwater Harvesting Facility is the latest project with purpose to collect and treat urban stormwater runoff. The treated urban stormwater of this wetland system is firstly supplied to the near located Mawson Lake residence, providing water for secondary uses, and secondly to G. H. Michell & Sons wool processing company. Also a significant amount of the treated stormwater is used to recharge the aquifers. The main focus of this research project is the reed bed pond in regards the nutrient dynamics of the water, sediment and macrophyte components.

1.1 Contribution of the study

The overall aim of this research is to get an improved understanding of the temporal and spatial nutrient dynamics related to the water column, sediment and plant community of the reed bed pond of the Parafield stormwater harvesting facility. In particular the nutrient interactions between the sediment and open water will be investigated, as well as the function of higher aquatic plants (macrophytes). In the past a lot of research has been performed to get a better understanding of the complex processes in wetlands linked to hydrology, soil and water plants in Australia and worldwide (Andersen 1975; Kamp-Nielsen 1975b; Blackburn and Henriksen 1983; Boström et al. 1988; Boon and Sorrell 1991; Gunnars and Blomqvist 1997; Mitchell and Baldwin 1999; Baldwin et al. 2000; Madsen et al. 2001). But the understandings of these kinds of processes have to be brought into the context to develop management strategies or plans, so that the systems can be sustained.

To get these information and understandings of the processes the following research questions have to be solved:

1. Is there in addition to allochthonous DOC and nutrient loadings a distinct amount of autochthonous loading from the wetland?

2. Is there a seasonal difference in the residence time and how does it affect the performance of the wetland?

3. What is the impact of plant harvesting in the reed bed pond regarding nutrient dynamics in the sediment and water?

4. Are the sediments functioning as source or sink?

5. How do all previously mentioned points affect the removal performance of the reed bed pond?

6. Use of HEA (Hybrid Evolutionary Algorithm) for knowledge discovery of the driving agents of the nutrients and discovering rule sets to predict nutrient concentrations

2 Literature Review (Background)

2.1 Wetlands

2.1.1 Wetland Definition

Definitions for wetlands include three components (Mitsch and Gosselink 2000): (1) the presence of water, either at the surface or within the rootzone, (2) the unique soil conditions in comparison to other ecosystems, and (3) the support vegetation adapted to wet conditions (hydrophytes)

There are different types of wetlands, for example marsh, peatland, mangrove, swamp, etc; but they are generally categorized into two categories of ecosystems: coastal and inland wetland ecosystems (Gore 1983). Wetland ecosystems are unique, because of their hydrological conditions and their role as ecotones (transition zones) between aquatic and terrestrial ecosystems (Hammer 1991; Mitsch and Gosselink 2000).

2.1.2 Wetland as Sink, Source and Transformer

Wetlands consist of water, substrates, microbiota, flora and fauna (Breen 1990; IWA 2000). The interactions between these components give wetlands the capacity to act as sinks, sources and transformers of chemicals and nutrients (Mitsch and Gosselink 2000). A wetland functions as a sink for a particular nutrient when the input is greater than the output, which means there is net retention of that nutrient. In cases where the output of a particular nutrient exceeds the input the wetland is a nutrient source. If the level of output and input of nutrients in the system is similar, but the chemical form is different, the wetland is defined as a transformer of nutrients. These functions of a wetland as a sink, source and transformer of nutrients can occur simultaneously; for example, a wetland can act as a sink for an inorganic nutrient and as a source for an organic form of the same nutrient (Mitsch and Gosselink 2000).

2.1.3 History of Wetland Research

The study of wetlands as ecosystems began in the 1940s with Lindeman's studies on Cedarbog Lake (Lindeman 1942) and with the work of Odum (Odum 1957) and Teal (Teal 1962) in the 1950s and early 1960s.

In the past, humankind thought about wetland only as wasteland, a place for release of insects and diseases, a land useless for human purposes. These misunderstandings and lack of knowledge about wetland ecosystems caused a decline of the number of wetlands worldwide, but over the past 50 years the importance of wetland ecosystems has been recognized and great efforts have been spent to restore and manage them. The functions of natural wetlands are the preservation of biodiversity, the effective removal or conversion of large quantities of organic matter, suspended solids, heavy metals and excess nutrients, and the purification of water quality (Carpenter 1980).

The concept of deliberately using wetlands for water purification has been developed within the last 20 years, but in reality human societies have used natural wetlands indirectly for waste management. Wetland plants can remove large amounts of nutrients for biomass production through absorption and assimilation during the growing season (Hammer 1995). Therefore in developed countries research on the building and use of constructed wetlands is very active (Mitchell 1978; Brix 1994b; Mitsch and Gosselink 2000). Especially *Phragmites*, *Typha* and other emergent macrophytes have high biomass and therefore a relatively high absorption rate of nutrients (Whigham et al. 1978; Lieffers 1983; Kim 1989; Ennabili et al. 1998). The growth of macrophytes functions as a motivator or activator of the flow of energy, nutrients and other inorganic or organic materials inside an ecosystem (Brinson et al. 1981). This flow of energy is also called nutrient cycling, which is complex and determined by many environmental parameters. A quantitative description of the inputs, outputs and internal cycling of materials in an ecosystem is called an ecosystem mass balance (Whigham and Barley 1979). The measuring of materials, elements such as phosphorus, nitrogen and potassium, is called a nutrient budget. In wetlands, nutrient budgets are used for the description of ecosystem functioning and to determine the importance of wetland as sources, sinks and transformers of chemicals. Nutrient budgets in wetland consider the major categories of pathway and storage, which are important in accounting for materials passing into and out of wetlands. Nutrients that are brought into a system are called inputs or inflows. In wetlands these inputs are primarily through hydrologic, such as precipitation, surface and groundwater inflow and tidal exchange. Precipitation is a major source of nutrients and ions for both terrestrial and aquatic systems (Likens et al. 1977).

Biotic pathways, like the nitrogen fixation and fixation of carbon through photosynthesis, are also noted as inputs. Hydrologic exports occurred at both surface water and groundwater unless the wetland is an isolated basin that has no outflow.

Long-term burial of chemicals in the sediments is also considered a nutrient or chemical outflow. Biologically mediated exports to the atmosphere are also noted as exports, like denitrification, ammonia volatilisation, CO_2 loss through respiration and methane release. Internal cycling involves exchanges of various pools, or standing stocks, or chemicals within the wetland. The cycling includes pathways like litter production, re-mineralization, nutrient uptake, storage and translocation (Mitsch and Gosselink 2000).

2.2 Constructed Wetlands

The decrease in the numbers of wetlands and recognition of their importance has increased the activities in research about restoration and creation of wetlands ecosystems. The term wetland restoration means the return of a wetland to its original or previous wetlands state, whereas creation involves conversion of uplands or shallow open-water system to vegetated wetlands (Mitsch and Gosselink 2000).

2.2.1 Definition of Constructed Wetlands

A constructed wetland is a wetland, that has been developed on a former upland environment to create poorly drained soil and wetland flora and fauna for the primary purpose of contaminant or pollution removal from wastewater or runoff (Hammer 1997). Constructed wetlands are also called treatment or artificial wetlands.

2.2.2 Constructed Wetlands: Utilisation of Natural Processes for Water Treatment

Constructed wetlands are designed for the purpose to improve water quality. There are three types of wetlands used to treat wastewater: natural wetlands and two types of constructed wetlands. Constructed wetlands can be classified by differed flow regimes, such as surface flow (SF) and subsurface flow (SSF) systems (Hammer 1991; Moshiri 1993; Kadlec and Knights 1996; IWA 2000).

Studies on natural wetlands, led to the usage of SF systems in North America, on the other site SSF systems were developed and used in Europe (IWA 2000; Mitsch and Gosselink 2000). In 1973 the first intentionally engineered, constructed wetland treatment pilot systems in North America were constructed at Brookhaven National Laboratory near Brookhaven, New York. These pilot treatment systems combined a marsh wetland with a pond and a meadow in series and were designated as the meadow/pond/marsh treatment system. The initial work on SSF wetland technology was done by Seidel and co-workers at the Max-Planck Institute in Germany, during

1960-80 (Seidel 1976) and (Kickuth 1977) in (IWA 2000).The treatment process was called the root zone method (RZM), and this technology was used in many European countries. In mid 1985 the British Water Research Centre used reed-bed treatment and started also to investigate the potential of horizontal-flow RZM systems, which became applied in Denmark. The first system in the UK was built in late 1985, but ten years later the number of these treatment systems in the UK was more than 400 (Cooper and Green 1995). Besides Germany and UK, SSF constructed wetlands were introduced in Austria, Denmark, France, Sweden, Switzerland, The Netherlands, North America, Australia and Africa in the 1980s. In the 1990s many SSF wetlands were built in other European countries, like Poland, the Czech Republic, Norway and Slovenia and in Asia, for example in China and India (IWA 2000).

The values of wetlands for improving water quality are often overlooked, yet wetlands can remove and transform both organic and inorganic materials, including human waste, toxic compounds and metals, from inflowing waters. Following wetland attributes improve water quality (Committee on Restoration of Aquatic Ecosystems - Science 1992): 1. Water floods into the wetlands from rivers and streams, its velocity decreases, causing an increase in sedimentation. Thus, chemicals sorbed to sediments are removed from the water and deposited in the wetlands. 2. Variety of anaerobic and aerobic processes function to precipitate or volatilize certain chemicals from the water column. 3. Accumulation of organic peat that is characteristic of many wetlands can ultimately lead to a permanent sink for many materials. 4. High rate of productivity of many wetlands can lead to high rates of mineral uptake by, an accumulation in, plant material with subsequent burial in sediments. 5. Shallow water coupled with the presence of emergent vegetation leads to significant sediment-plant-water exchange.

Sediment is playing a key role in wetland ecosystems, but not much is known about the processes in the sediments and especially their interactions between other parts of the ecosystems. So more investigation is needed to understand the processes inside the sediments (Committee on Restoration of Aquatic Ecosystems - Science 1992).

2.3 Wastewater

2.3.1 Sources and Characteristics of Wastewater

Wastewater is water that has been polluted or altered chemically and physically through human activities. The composition and contribution of pollutants in wastewater depend on the origin and nature of pollution source, which is either through point source (localised origin, PS) pollution and/or non-point source (non-localised origin, NPS) pollution. The sources for wastewater are agricultural, industrial and municipal wastewater (Kadlec and Knights 1996).

Agricultural wastewaters are generated from a wide range of agricultural endeavours including dairies, feed lots and aquaculture (Kadlec and Knights 1996) and can be of point source or more diffuse origins. Agricultural wastewater contains high biochemical oxygen demand (BOD_5), chemical oxygen demand (COD), nutrient concentration (80-200g/m^3 N and 10-25 g/m^3 P) (van Oostrom and Cooper 1990).

Industrial wastewater quality varies according to its origins. Sources for industrial wastewater include landfill leachate, mine drainage, pulp and paper industry and textile industry. Industrial wastewaters are generally point source pollutions. The major pollutants include nutrients, BOD_5, various salts, stains, organics and heavy metals (Kadlec and Knights 1996).

Municipal wastewater can be separated into two categories: stormwater and domestic sewage. Stormwater is usually generated after rain events as a result of runoff from urbanised areas such as roads, drains, car parks and roofs and is indicated as a major contributor for non-point source pollution (Kadlec and Knights 1996; Ferguson 1998; IWA 2000). Stormwater runoff contains a variety of pollutants: sediments, nutrients, trace metals and organic compounds (Herricks and Jenkins 1995; Ferguson 1998; IWA 2000). Therefore, unlike the other forms of wastewater, the production of stormwater tends to be unpredictable and intermittent (Raisin et al. 1997). Domestic sewage originates from households public buildings and is therefore a point source pollutant. Sewage wastewaters have high nutrient loads and high BOD_5.

2.3.2 Methods of Wastewater Treatment

Wastewater treatment contains physical, chemical and biological processes. Many of these processes are general in nature and can function within a variety of treatment schemes (Kadlec and Knights 1996).

Primary wastewater treatment removes gross pollutants and suspended solids through sedimentation and formation of sludge in relatively deep (3-5m) ponds. Floating pollutants such as grease, oils and plastics form the scum and are physically removed from the surface (Kadlec and Knights 1996).

Secondary wastewater treatment removes solids and dissolved organic matter through microbial activity. The aerobic conditions required for the microbial communities are provided through oxidation ponds, facultative ponds, aeration basins and trickling filters. Advanced, or tertiary wastewater treatment reduces nutrients such as nitrogen and phosphorus through a variety of chemical, physical and biological processes (Kadlec and Knights 1996).

Effluents from secondary or tertiary treatments are often discharged into fresh or marine water bodies for disposal. High concentrations of pollutants remaining in the effluent may cause detrimental changes to the receiving ecosystems, such eutrophication and subsequent algal blooms (Gersberg et al. 1983). Ensuring that ecosystems receive treated effluent of loadings no greater than that of natural systems (ca.22 g N $m^{-2}yr^{-1}$), (Mitsch and Gosselink 2000) will reduce detrimental impacts. Alternatively, the treated water can be recycled for irrigational use in parks and gardens or natural aquifer recharge (Tomlinson et al. 1993).

2.3.3 Wetlands for Wastewater Treatment

Wetlands have been reported as receiving wastewaters of varying degrees of pre-treatment. The level of pre-treatment required depends on the origin of the wastewater, but usually receives at least primary treatment before entering wetland systems. Wastewater of industrial origin may have high concentrations of toxic substances and therefore require, a high degree of pre-treatment and in extreme cases, treatment within a wetland system will not be appropriate. Is the degree of pre-treatment high, than there is a higher potential for effective treatment within the wetland system. Many pollutants from many sources of wastewater have been treated in wetlands; nutrients, heavy metals, BOD, suspended particles (Hammer 1991; Kadlec and Knights 1996), but the most attention in treatment of domestic wastewater has focused on major nutrients (nitrogen and phosphorus).

2.4 Urban Stormwater

2.4.1 Urban Stormwater?

Urban stormwater is explained as the runoff water from urbanized areas and their upstream catchments due the major flows during and following rain, as well as dry weather flows (Pavelic et al. 1992; Herricks and Jenkins 1995; Ferguson 1998; IWA 2000). Stormwater runoff originates due to a stormwater discharge, which is developed when during a rain event the storage capacity of the catchment area is exceeded. The runoff will be influenced by the intensity and duration of the rain event, the number of watersheds and land-use characteristics, such as slope, soil type and impervious area (Livingston 1991; Debo and Reese 1995; Herricks and Jenkins 1995; Ferguson 1998). These characteristics also influence the contents of pollutants and their concentrations in the stormwater runoff, which is considered to be a major contributor to non-point source pollution of surface waters. Urban stormwater runoff originate from a wide range of sources, which can be grouped into domestic, agriculture and industrial sources (Herricks and Jenkins 1995; Ferguson 1998; IWA 2000; Field and Sullivan 2003). Studies have revealed that increasing urbanisation had led to a large increase in the pollutant loads delivered to natural receiving waters (IWA 2000; Gervin and Brix 2001). Treatment of urban stormwater runoff is in many areas required for the protection or improvement of the water quality of receiving waters (Martin 1988).

2.4.2 Control and Management of Urban Stormwater

In general there are three approaches to control and manage urban stormwater, which are the use of dry ponds, wet ponds and stormwater wetlands (Schueler 1992; IWA 2000; Barr Engineering Company 2001).

2.4.2.1 Dry Ponds

Dry ponds, also called “detention ponds”, are designed for the purpose to intercept stormwater runoff and impound the water for a short period of time and then gradually release the water to the receiving system or storm sewer system (Herricks and Jenkins 1995; IWA 2000; Barr Engineering Company 2001). In most cases they are designed to completely dry out between runoff events or to empty in a time period of less than 24 hours. Therefore dry detention ponds provide a control for the runoff rather than have impact on the water quality. Due to the short hydrologic retention time (HRT) in the dry ponds it provides a low removal rate of pollutants and only limited settling of particulate matter, which can be resuspended in large portion by subsequent runoff events.

Table 2-1: Advantages and limitations of dry "detention" ponds

Advantages	Limitations
• Control of peak flow rate • Use as recreational area • Performance non-influenced by climate or season	• Marginal removal of stormwater pollutants • Potential for clogging of outlets • Prescribed for drainage areas less than 10 acres • Resuspension of sediments in the ponds when not removed between storm events • Unattractive by residents in case of poor maintenance, which can create nuisance odors, weed growth and collection of trash

For this reason dry ponds should be considered mainly as practices used to reduce the peak discharge of stormwater to the receiving systems, to limit downstream flooding and provide to some degree of channel protection. Advantages and limitations are listed in the following table (Table 2-1) (Schueler 1992; Barr Engineering Company 2001).

2.4.2.2 Wet Ponds

A wet pond, which is also called a retention basin, is a constructed stormwater pond that permanently inundated. Sedimentation is the primary pollutant removal mechanism in wet ponds.

Table 2-2: Advantages and limitations of wet "retention" ponds

Advantages	Limitations
• Capable of removing both solids and soluble pollutants • Depending on design, can have an aesthetic function • Creation of wildlife habitat, when properly planted and maintained	• Generally not prescribed for drainage area smaller than 10 acres • Require relatively large land area • Improperly designed or maintained results in stratification and anoxic condition, which leads to nutrients and metals release from the sediment • Concern for mosquitoes and maintaining oxygen levels in the pond • Unattractive by residents in case of poor maintenance, which can create nuisance odours, weed growth and collection of trash

Significant amounts of suspended metals, nutrients, sediments and organics can be removed, which mainly depends on the hydraulic retention time (HRT) where physical adsorption to bottom sediments and suspended fine sediments, natural chemical flocculation, bacterial decomposition and uptake by aquatic plants and algae are utilised.

The wet ponds have a moderate to high capacity in removing most urban pollutants depending on the ratio of the volume of wet pond to runoff from the catchment areas. Wet ponds can be built in residential, commercial and industrial areas and provide runoff treatment or flow control when they are incorporated into an extended storage or a detention pond system. They are less effective in decreasing runoff volumes although some infiltration can occur, as well as evaporation in summer months (Schueler 1992; Herricks and Jenkins 1995; IWA 2000; Barr Engineering Company 2001). The advantages and limitations are listed in Table 2-2.

2.4.2.3 Stormwater Wetland

Stormwater wetlands are constructed wetland systems designed with the purpose to maximize the removal of pollutants from stormwater runoff due to the following mechanisms, like microbial breakdown of pollutants, plant uptake, retention, settling and adsorption (Schueler 1992; Herricks and Jenkins 1995; IWA 2000; Barr Engineering Company 2001).

NOTE:
This figure is included on page 11 of the print copy of the thesis held in the University of Adelaide Library.

Figure 2-1: Different designs for stormwater wetlands (Design 1: Shallow Marsh System, Design 2: Pond/Wetland System, Design 3: Extended Wetland, and Design 4: Pocket Wetland System) Source (Schueler 1992)

Stormwater wetlands temporarily store runoff in shallow pools that support condition for the growth of aquatic macrophytes and microbial populations, which in combination can remove soluble carbon, nutrients, metals and potentially reduce BOD and fecal coliform level concentrations (Schueler 1992; Barr Engineering Company 2001).

Typically stormwater wetlands are not as complex in their ecological functions as natural wetlands, but they are designed specifically for flood control and water quality purposes (Schueler 1992; Barr Engineering Company 2001). Figure 2-1 shows different designs of stormwater wetlands exist.

The advantages and limitations of stormwater wetlands are listed in the following table (Table 2-3):

Table 2-3: Advantages and limitations of stormwater wetlands

Advantages	Limitations
• Improvement in water quality (downstream) • Settlement of particulate pollutants • Reduction of oxygen-demanding substances and bacteria from urban runoff • Biological uptake of pollutants by aquatic plants • Flood attenuation • Reduction of peak discharges • Enhancement of vegetation diversity and wildlife habitat in urban areas • Aesthetic enhancement and valuable addition to community • Relatively low maintenance costs	• Release of nutrients in fall • May be difficult to maintain vegetation under a variety of flow regime • Large land requirement • Until vegetation is established pollutant removal efficiency is lower than expected • Relatively high construction costs

2.4.2.3.1 The role of wetland vegetation

Wetlands can support a range of water quality management objectives. The processes influencing water quality in wetlands resemble those operating in better-known aquatic environments. The wetland's inflow, organic matter and nutrient loads and hydrologic regime determine the dominance of particular processes in the wetland and their relative importance. Wetland water quality is influenced by a complex array of processes including:

- biological uptake of nutrients and metals by aquatic vegetation
- formation of chemical complexes of nutrients and metals in the sediments
- coagulation of small particles
- filtration and surface adhesion of small particles by vegetation
- enhanced sedimentation of smaller particles in vegetation
- direct sedimentation of larger particles
- decomposition of accumulated organic matter

- gas losses through chemical and microbial processes (ammonia, nitrogen, methane, hydrogen sulphide)
- microbial UV disinfection by exposure to sunlight
- The three significant types of processes are:
- biological and chemical processes involving soluble materials (e.g. uptake of nutrients by epiphytes, adsorption and desorption of phosphorus onto and from particles, nitrification and denitrification)
- coagulation and filtration of small, colloidal particles (e.g. adhesion of colloids and particles on the surface of aquatic vegetation. These particles are in size density range that makes them too small to settle under all but the most quiescent conditions.)
- physical sedimentation of particles (e.g. sedimentation in wetlands due to decreased water velocity. Large plants (macrophytes) such as reeds and rushes enhance this process by further reducing turbulences and water velocity)

Wetlands vegetation creates the physical and biological conditions required for the successful removal of finely graded particles and associated pollutants. The physical conditions created by wetland vegetation that maximize the removal finely graded particles include uniform flow distribution and flow retardation, leading to increased pollutant contact with plant surfaces. Emergent vegetation minimizes wind-generated turbulences. The root system of wetland vegetation binds and stabilizes deposited particulates, protecting them against resuspension. The root zone can also modify sediment redox (reduction – oxidation) conditions and influence the stability of pollutant trapped in the sediments. Because most pollutants are transported during storm events, physical processes are more important in trapping pollutants at these times. Biological processes become more important under low flow conditions when previously trapped materials are transformed and recycled. Small suspended particles adhere to plant surfaces, which act as filters. Plants also provide a surface on which photosynthetic organisms such as algae can grow. These epiphytic algae remove both fine particles and dissolved pollutants from the water column.

2.4.3 Change in the Approach to Urban Stormwater Management

In highly urbanised areas up to 90% of urban stormwater runoff can be directed into the drainage systems to channel them to the nearest natural water body (Scott 1996;

Environment Australia 2002). The approach of stormwater management in Australia in the past focused mainly at flood mitigation during major rain events. However growths in urban areas, industrial development and failure of aging sewers have led to an increase in stormwater pollution in both local and downstream ecosystems requiring urban stormwater management. In contrast to the environmental problems stormwater is recognized nowadays as a valuable resource, in comparison to waste water, which has the potential to bring social, economic and environmental benefits to urban areas (Scott 1996; Wong et al. 1999; Environment Australia 2002). Based on estimates in 1989 the estimated amount of stormwater runoff in Adelaide was a total of 150,000ML, which just flows into the ocean, which almost equals the amount of Adelaide's water use (190,000ML) (Fisher and Clark 1989; Environment Australia 2002). In 1991 estimates of stormwater runoff deriving from the Adelaide metropolitan area was approximately 210,000ML, which is a greater volume than the total annual mains water use in Adelaide (Environmental Consulting Australia 1991).

Over the last decade the approach towards stormwater management in Australia has changed and focuses on the source control of pollutants in runoff, in-stream controls and treatment processes, measures to prevent scouring, erosion and sedimentation, enhanced infiltration and detention to reduce peak flows, design of drainage corridors and urban waterways for aesthetic, ecological and recreational purposes, water quality monitoring, measures to promote water conservation, including stormwater re-use and community education and participitation (Scott 1996).

2.4.3.1 Integrated constructed wetlands with stormwater management

Stormwater management involves the use of various techniques with different purposes and benefits, which include the following (Wong et al. 1999):

- Flood protection and flow control
- Water quality improvement
- Landscape and recreational amenity
- Provision of wildlife habitat

A typical stormwater management system can consists of the following components:

- Gross pollutant trap (GPT) – to trap artificial and natural litter and coarse particles like gravel and sand
- Pollution control pond/constructed wetland inlet zone – to trap sand – to silt – sized particles and improve water quality. This module can have some secondary benefits, including landscape aesthetics and flow attenuation
- Macrophyte zone ie. an area of plants such as rushes, reeds and sedges – to improve water quality through the trapping of fine particles and soluble pollutants. This module can have some secondary benefits, including wildlife habitat and flow attenuation.
- Lake/island – to provide passive recreation, landscape enhancement and wildlife habitat. Depending on the outlet structure, lakes can significantly attenuate flow. Lakes can also provide water quality benefits, but this function can be compromised of the lake attracts large populations of wildlife, which can degrade water quality.
- Flood retarding basin – to protect downstream areas from flooding and to control stream hydrology. This module can provide more open space within urban landscape. Stormwater treatment modules located in flood retarding basins can benefit from extra hydrological control provided by the basin

2.4.4 Stormwater Research in Australia – Who is doing what?

Stormwater research in Australia has been undertaken by many organizations like the CSIRO, CRC, Universities, State Water Agencies and private companies. Most of this research is focusing on Australia's largest cities, in particular Sydney and Melbourne (Scott 1996).

In Sydney there has been a growing awareness over the last two decades within the communities and the water authorities. Stormwater and associated sewer overflows are a major cause of pollution in urban waterways and ocean beaches. This had led to large number of projects being implemented by the Sydney Water Corporation as part of their Clean Waterways Program. Other organizations involved are, NSW Dept Land & Water Conservation, NSW EPA, University of NSW, University Technology Sydney, University of Western Sydney, Sydney Coastal Councils and The Upper Parramatta River Catchment Trust (Scott 1996).

In town other than Sydney a small number of projects were undertaken in Wollongong (Goldrick and Armstrong 1994; Simeoni and Hickey 1995), in Newcastle (Cupitt 1992; Osborne 1992), in Lismore (Kerr and Eyre 1995) and in Armidale (Southcott 1995).

In Melbourne, monitoring has shown that stormwater is a major contributor of pollution in urban waterways, such as the Yarra River and also in Port Philipp Bay (Senior 1992). This program was established in 1985 and has resulted in Melbourne Water and Melbourne Parks & Waterways initiating a series of projects with the purpose to reduce pollution by stormwater. Apart from Melbourne there are only a few projects, like the project in Ballarat to develop a flood mitigation strategy (Elshaug 1995).

In Queensland all stormwater projects are based in Brisbane, where a consortium of local councils and state water authorities have joined together to carry out "The Brisbane River and Moreton Bay Wastewater Management Strategy. This study was conducted to determine the major source of pollution and develop wastewater management strategies, which are ecologically sustainable. The Brisbane City Council has initiated a program of stormwater quality and quantity data collection from several sites within the Brisbane region (Bycroft et al. 1995). The results of this program are being used to calibrate two stormwater models AQUALM-XP and SWMM and these will be used for the long term predictions of pollutant loads (McAlister et al. 1995).

In Perth the concept of "water sensitive urban design" has been developed by Murdoch University and the Water Authority of Western Australia (WA). The concept emphasizes the need to integrate water and land-use planning at a strategic level in the urban planning process (Mouritz et al. 1994). In the Swan River Estuary an increasing incidence of algal blooms other stormwater studies have focused on monitoring the concentrations and loads of nutrients passing into the Swan River via stormwater drains (Tan 1992; Sharma et al. 1995).

In the ACT stormwater management is more advanced than in any other part of Australia. A management strategy has been developed and is constantly updated, especially when a new satellite town is planned. The implementation of this strategy required the construction of a range of water quality control structures including urban lakes, water quality control ponds, wetlands, major and minor gross pollutant traps and retention of natural creeks and drainage lines (Winsbury 1992; Lawrence and Reynolds 1995).

In the Northern Territory the main project focus on stormwater issues has been the monitoring of pollutants entering Darwin Harbour (Townsend 1992).

In Tasmania the focus of urban water management has been associated by determining the roles and functions of the state and local authorities, which are responsible for water, sewerage and drainage, with the purpose of improving the institutional and regulatory arrangements (London Economics 1995).

2.4.5 Stormwater Research in Adelaide, South Australia

2.4.5.1 Nature of water use in Adelaide (Main Waters)

In case of Adelaide, it has the most expensive water in comparison to all Australian capital cities. The water for the metropolitan area of Adelaide is supplied from the Mt. Lofty catchments and the River Murray and exports most the wastewater to the Gulf of St. Vincent. The water supplied from the Mt. Lofty catchments and the River Murray is treated (filtered and disinfected) although only a part of the water (around 40%) is available for human consumption, contact usage and some industrial applications for which the treatment is necessary. The rest of the water is used for irrigation, which counts for almost two thirds of all metropolitan water consumption. To categorize the water usage, residential users consume 70% of the total water use. About 12% is used for irrigation by councils, schools and other government departments, while the industrial sector uses 18%. Based on reports from the (Engineering and Water Supply 1989) the forecast is, that the water consumption in Adelaide will increase from 30,000 to 80,000 ML within 30 years, depending on the projected level of water use efficiency (Pavelic et al. 1992). Options identified by the report for further increase of Adelaide's water supply include harvesting urban stormwater runoff. Additional sustainable options include sewage effluent recycling, on-site detention of rainwater and recycling of mains water.

2.4.5.2 Stormwater a Resource for Adelaide, South Australia

Miles idea on the resource value of stormwater in Adelaide have been supported in reports by (Read 1978) and (Fisher and Clark 1989). The benefits of stormwater harvesting to Adelaide have been identified by (Fisher and Clark 1989) in form of improved water resource utilization, reduced pollution loads to the natural environment, aesthetic enhancement and flood mitigation. In addition to these reports a comprehensive report by (Environmental Consulting Australia 1991) discussed the development of a Stormwater Management Strategy for Adelaide. Some creative solutions involving the collection, treatment, storage, retrieval and re-use of stormwater runoff to replace the component of mains supplied water used for "second quality" purposes had emerged (Clark 1992; McIntosh 1992; Argue 1994; Bekele and Argue 1994; Argue 1995a, b).

The Salisbury Council, located 25 km north of Adelaide, occupies an area of 161 square kilometres extending from the escarpment and foothills of the Mt Lofty Ranges to the shores of Gulf St. Vincent; tried for many years to use stormwater to enhance the environment, improve and develop the amenity and recreation opportunity, which can be achieved by one or more practical means, such as:

- water flowing across wide shallow "floodways", which will assist the growth of trees and maintain green grass throughout the year
- drainage swales and "natural from" open drains, together with suitable planting of trees and shrubs, will provide natural areas for children's play, which are often lacking in our urban areas
- ponds and parks, fed by stormwater can provide fishing and yabbying as well as a visual attraction in a park of landscape
- the development of wetlands can provide a very valuable habitat for birds, which can also be sensitively used by people for recreation, bird watching and environmental education
- development of standards on water quality

These kinds of developments can be incorporated into stormwater systems at a cost which is no more or less than that of conventional piped system. Increasing knowledge and understanding of the capacity of natural and created wetlands to treat and remove pollutants from the water. Therefore if the water from a wetland is given final purification in form of simple filtration and disinfection process, it would be possible to use this water as a potable water source for supply.

The Council has, over the last two decades, implemented a large number of developments which have used stormwater to enhance the local environment and improve the amenity or recreation opportunity of the area. The Council has developed over 30 wetlands covering a total area of 250 ha with a total construction cost of $16 million (Table 2-4).

The first project of the Salisbury Council was the Para Hills Paddocks, designed and constructed in 1974-5, which consists of a large area with natural creeks and shallow drainage swales and various ponds, lakes, wetlands and flood plain areas. The Para Hills Paddocks is an excellent example of the multiple benefits to be gained from using stormwater in a total landscape development. The benefits beside the recreational areas, which can be used for a variety of activities, such as walking, fishing, bird watching, etc., it also provides habitats for many species of birds in particular water birds, with several species beginning to breed in the wetland areas. Less obvious benefits related to stormwater disposal and water pollution. Drains carrying the stormwater flows across the site, not in costly, fast-flowing, concrete pipe systems, but in natural creeks, along grassed drainage swales, through wetland areas and detained in a number of ponds and lakes, the flows are slowed and retained on-site for longer periods, reducing peak flows and hence placing less demand on downstream drainage systems. The slower flow across grassed areas and through wetlands and ponds also results in considerable improvement in water quality, as sediments, discolouration and dissolved nutrients are removed by natural processes, which would mean a decrease of pollutants, which would reach the natural systems. With the current concerns and awareness of environmental pollution and degradation, the need to conserve natural resources and the increasing emphasis on re-afforestation and land care, Councils initiative in utilizing stormwater as a resource for environmental and amenity benefit can be seen as providing a lead to changing attitudes to the issue of stormwater disposal. Serious considerations is now being given by State Authorities to using treated and purified stormwater to augment water supplies for both underground and mains supply.

This project was followed by Lake Windemere, a formal park lake in Salisbury North, which is supplied entirely with stormwater. A number of drainage swales have been constructed in several locations in the city as an alternative to conventional pipe systems. The Little Para Overflow Drain is being progressively developed to carry excess flood flows from the Little Para and will consist of a wide shallow swale, grassed and treed, to form a linear park through residential areas. This will also contain a series of small lakes fed by local stormwater flows.

The Greenfields wetlands were one of first large constructed wetlands in Australia. In 1984, the City of Salisbury prepared and approved the initial concept of developing 42 ha of low-lying, saline land into a stormwater detention basin and wetlands habitats.

Table 2-4: Locations of implemented wetlands by Salisbury Council

The Paddocks, Para Hills **Greenfields (Stage 1, 2 and 3.)**, Greenfields **Lake Windemere**, Salisbury North **Whites Road**, Bolivar **Kaurna Park**, Burton **DSTO**, Elizabeth West **Montague Road**, Cavan **Technology Park Lagoon**, Mawson Lakes **Walpole Road**, Paralowie **Willowbrook**, Paralowie **Kensington Way**, Burton **Beadell Street**, Burton **Golf Course**, Salisbury **Pitman Park**, Salisbury **General Drive**, Paralowie **Ohio Court**, Paralowie **Little Para Overflow Ponds**, Fairbanks Drive & McQueen Court Reserves **Noak**, Noak Place, Montague Farm **Montague Farm**, Montague Farm **LangfoRoad Street**, Pooraka **Burton Road**, Burton **Little Para Outlet**, West of Port Wakefield Road **Swan Alley**, Bolivar	**Daniel Avenue**, Bolivar **Happy Home**, Waterloo Corner Reserve **Clayson Road**, Salisbury East **Pines Lakes Ornamental**, Parafield Gardens **Pine Lakes ASR**, Parafield Gardens. **Pickering Reserve**, Pooraka **Bush Park**, Pooraka **Creaser Park**, Parafield Gardens **Lower Rowe Park**, Ingle Farm **Bridge Road Plantation**, Bridge Road **Camelot Reserve**, Paralowie **Tecoma Street Reserve**, Parafield Gardens **Stanford Road Reserve**, Salisbury Heights **Norwich Road Reserve**, Salisbury East **Strowan Park**, Salisbury **Jobson Road Reserve**, Waterloo Corner **Mawson Lakes Main Lake**, Mawson Lakes **Warrendi Road**, Mawson Lakes **Alana Court**, Burton **Shearwater Lake**, Mawson Lakes **Gulfview Heights**, Salisbury East **Railway Wetlands**, Mawson Lakes **Yalumba Drive**, Paralowie

After councils approval in 1989, stage 1 (25ha) was completed in 1990, Stage 2 (17ha) was completed in 1993 and the largest stage3 (72ha) in 1995. This area of land has become home to over 160 species of birds, eight species of fish, four species of frog, yabbies, long-necked tortoise and numerous aquatic and invertebrate fauna. The development includes a natural trail with boardwalks and bird hides, facilitating environmental education and ecotourism. The GFW will use stormwater from the local catchment and Dry Creek drain to create extensive area of wetland landscape and habitat. Others projects, large and small are under consideration. These projects all have a common factor in that stormwater, instead of being conveyed away as quickly as possible in underground pipes, is utilized to benefit the environment, the landscape and the community in some way. The wetlands at the Para Hills Paddocks, which can be seen to considerably improve the quality of water flowing through the area, and the

Greenfield wetland development now under construction, are seen as ideal sites to monitor and test the performance of created wetland to improve and purify polluted water. Council is now combining with State Authorities to establish monitoring and sampling stations at these projects, so that the actual performance of wetland system in SA environment can be assessed and better understood. It is hoped that with this knowledge, it will be possible to establish wetland projects with the ultimate aim of using stormwater for water supplies. The benefits of such wetland developments are many. These include reduced pollution of our environment, improvements in flood mitigation, increased opportunities for recreation and improved amenity, the provision of valuable habitat for birds, as well as water supply.

2.4.6 Constructed Wetlands for Stormwater Treatment (CWST): Tapping for a precious Water Resource for Dry Countries

Studies have been done over the last 15 years on the use of wetlands for treatment of stormwater. The fundamental concepts of using wet settling basins for water treatment have long been accepted and practised. Design guidance for stormwater treatment was not available until the early 1980s, when the US Environmental Protection Agency's Nationwide Urban Runoff Project was completed. Results from this national research and demonstration effort provided the data needed to develop treatment basin sizing rules based on field data. The results of this project have yielded local and national guidance to maximize the removal of pollutants through the treatment systems (IWA 2000). A study of the estimated long term removal rates in the mid-Atlantic region of the USA for 60 stormwater wetlands, including constructed and natural wetlands and pond-wetland systems showed the following results (Schueler 1992): total suspended solids 75%, total nitrogen 25%, total phosphorus 45%, organic carbon 15%, lead 75%, zinc 50% and bacteria 2 log (10-2) decrease.

2.5 Sources and Pathways of Nutrients and DOC in CWST

2.5.1 Sources of Nutrients and DOC in CWST

The contents and qualities of nutrients or organic matter in wetlands and constructed wetlands vary widely to their nature and position (Pieczynska 1986). Dissolved organic matter (DOM) is the largest pool of reactive organic carbon (Hedges 1992) providing chemical energy and nutrients (Wetzel 1992). DOM will be differentiated between low molecular weight compounds, such as monomeric sugars and amino acids, which are directly utilizable by bacterio- and phytoplankton (Baines and Pace 1991; Münster 1993), and high molecular weight humic substances, which cannot be directly utilized. Humic substances accumulate and consist of 50-70% of dissolved organic carbon in most fresh and coastal waters (Aiken et al. 1985). So humic substances contain between 50-70% of carbon (Aiken et al. 1985), 3-6% of nitrogen (McKnight and Aiken 1998) and phosphorus (Jones et al. 1988; de Haan et al. 1990).

DOM originates from autochthonous (internal production) and allochthonous (external production) sources (Westlake et al. 1998). Autochthonous DOM is essentially derived from algae and macrophytes (Bertilsson and Jones Jr. 2003), where the contribution of phytoplankton is considered to be important (Baines and Pace 1991; Münster 1993), that influences both the activity and the composition of aquatic microbial communities (Pomeroy 1974),also called microbial loop. It describes the conversion of DOM to POM by bacteria. This process mediates the transfer of energy and matter from dissolved organic matter to higher trophic levels and therefore controls the productivity of aquatic systems (Azam et al. 1983; Tranvik 1992). Allochthonous DOM is a source of organic carbon, nitrogen and phosphorus to aquatic ecosystems, that is derived from the surrounding areas (Findlay and Sinsabaugh 2003). When precipitation moves through the atmosphere, washes through vegetation, infiltrates the soil and percolates downward through mineral soil horizons. DOM is typically the product of dissolved atmospheric dust and gases, through-fall, root exudate, leaf and root litter, and the primary and secondary metabolites of microorganisms (Aitkenhead-Peterson et al. 2003). Recent research also suggests that weathering of shale and consequent microbial reworking of adsorbed organic carbon is an additional allochthonous source of dissolved and particulate organic carbon to surface water (Petsch et al. 2001; Raymond and Bauer 2001).

2.5.2 Nitrogen Cycle

Nitrogen (N) compounds are the principal constituents of concern in wastewater, because of their role in eutrophication, their effect on the oxygen content of receiving waters, and their toxicity to aquatic invertebrate and vertebrate species (Mitsch and Gosselink 2000; Mitsch et al. 2000; Reilly et al. 2000). They are also of an interest as nutrient for plant growth, which stimulates the production of wildlife. The most important forms of nitrogen in wetlands are ammonium (NH_4^+), nitrate (NO_3^-), nitrite

(NO_2^-), nitrous oxide (N_2O) and dissolved elemental nitrogen (N_2). The sum NH_4^+, NO_3^- and NO_2^- is referred as dissolved inorganic nitrogen. Nitrogen is also present in wetlands in many organic forms like urea, amino acids and purines (Kadlec and Knights 1996; Dodds 2002). The transport of nitrogen is called nitrogen cycling (Figure 2-2) and is predominantly mediated by bacteria. It includes a number of pathways occurring under oxic and anoxic conditions, which are nitrogen assimilation, ammonification, nitrification, denitrification, nitrogen fixation through the atmosphere, ammonia volatilisation, nitrogen release from biomass decomposition, burial of organic nitrogen and ammonia adsorption.

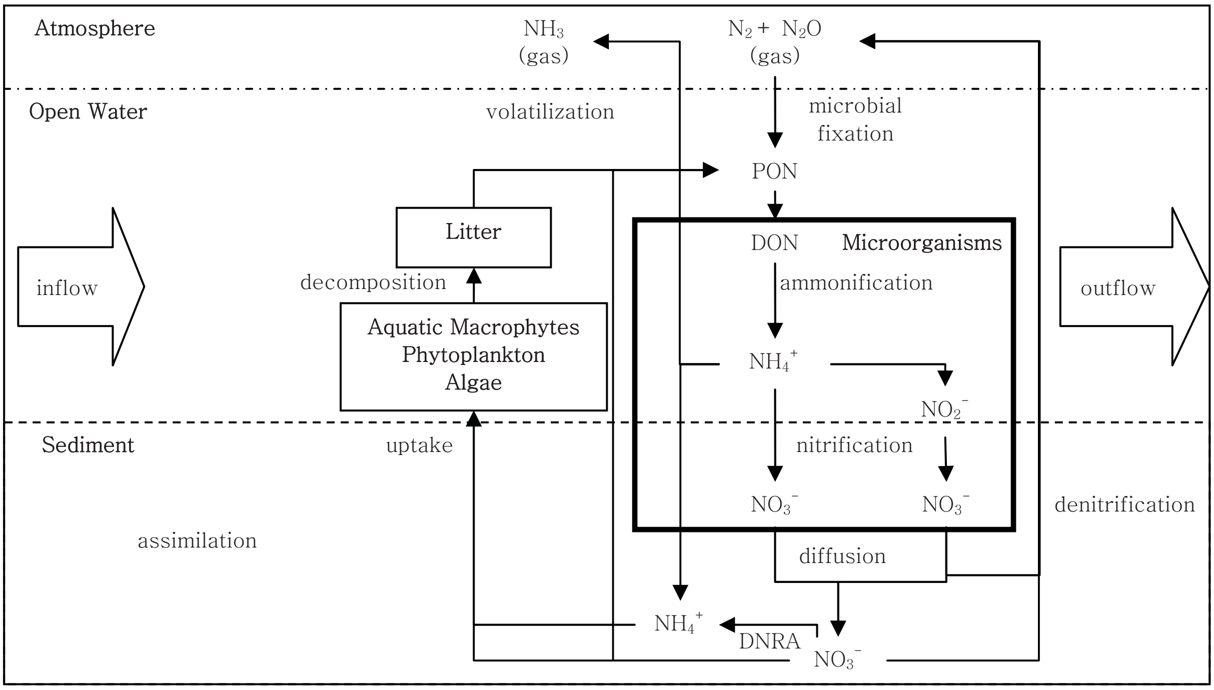


Figure 2-2: Nitrogen Cycle in Wetlands. (PON = Particulate Organic Nitrogen, NH_4^+ = Ammonium, NH_3 = Ammonia, NO_3^- = Nitrate, NO_2^- = Nitrite, N_2O = Nitrous Oxide and N_2 = Nitrogen, DNRA = dissimilatory nitrate reduction to ammonia)

Nitrogen compounds are one of the major factors responsible for the decrease of water quality, so that past and present studies on constructed wetlands focusing on the effective removal of N compounds in constructed wetlands (Arheimer and Wittgren 1994; McDowell and Asbury 1994; Reilly et al. 2000; Spieles and Mitsch 2000; Gervin and Brix 2001; Arheimer and Wittgren 2002; Braskerud 2002; Fraser et al. 2004). The reactions or pathways considered before are the major pathways for nitrogen removal. Especially the processes of nitrification and denitrification at the anaerobic–aerobic interface of the wetland substrate is thought to be the primary means of nitrogen removal (Brix and Schierup 1989b). Constructed treatment wetlands in general are not as successful as natural wetlands in removing nitrogen. (Thompson et al. 1995) found that nitrate removal via denitrification is 44 times lower in a constructed wetland than in a natural marsh.

Processes involved in reducing N are positively or negatively influenced by different environmental factors. According to (Craft 1997) nitrate removal in constructed wetlands may be limited by Eh or organic carbon availability, whereas nitrate removal in natural wetlands may be limited by nitrate availability in the anaerobic substrate. Hydrological conditions, like hydrological loadings (Moustafa et al. 1996) and residence time (Reddy et al. 1989) have also an influence on the processes for nitrogen removal. In addition to hydrology and the influent nitrogen form and concentration, specific physiochemical characteristics of a wetland have been found to influence its nitrate removal efficiency (Gersberg et al. 1983, 1984).The addition of a carbon source increased total nitrogen removal in the constructed wetlands of these studies from 25 to 95%. The availability of a carbon source increases the potential for denitrification. The rates of nitrification and denitrification depend on the presence and depletion of dissolved oxygen in flooded soil (Engler et al. 1976; Patrick and Reddy 1976) and the pH of water (Phillipps et al. 1994; van Oostrom and Russell 1994; Kadlec and Knights 1996). Temperature, one of the simplest regulating factors, becomes important in the consideration of the seasonal denitrification potential of temperate treatment wetlands. Biological removal of N is most efficient at 20-25 ºC (Sutton et al. 1975), and ambient temperatures in treatment wetlands influence both microbial activity and diffusion rates (Phipps and Crumpton 1994).

2.5.3 Contribution of Sediments and Water Plants to the Nitrogen Cycle

The contribution of sediments and water plants (macrophytes) to the N cycle is influenced by the transformation mechanisms of the N cycle, which are ammonification, nitrification, denitrification, assimilation and volatilisation. In case of denitrification and ammonia volatilisation, they are processes through which N compounds will be transferred out of the system in form of ammonia and nitrogen gas. The contribution of the sediment to the N cycle is a very important one, because most of the processes, which were mentioned before are occurring in the oxic and anoxic phase of the sediment. In oxic conditions, ammonia, produced through the ammonification of organic matter, can be converted to nitrate via nitrite by bacterial action (nitrification).

Ammonia can be either taken up by algae, or, if the pH is sufficiently high, lost to the atmosphere through volatilisation (Reddy and Patrick 1984).

In anoxic conditions nitrate can either be reduced back to ammonia (DNRA) or converted to N gas (denitrification). So denitrification is a pathway for the removal of N from aquatic systems, similar to ammonia volatilisation. This is particularly important if nitrification and denitrification are coupled, because the nitrate produced in nitrification can be quickly converted to N gas rather than assimilated into biomass (Reddy et al. 1989; Boon and Sorrell 1991). So the sediment is the main contributor to N cycle. The products of these pathways, like ammonium and nitrate will be taken up by the biota, including macrophytes, algae, phytoplankton and bacteria. The N compounds than will be translocated and stored in case of the macrophytes (Brix 1994a; Ennabili et al. 1998; Clarke and Baldwin 2002; Karunaratne et al. 2004). Under natural conditions macrophytes, algae and phytoplankton will die and will decomposed by bacteria and the stored N compounds in the macrophytes will be released, which will be considered as autochthonous source (Wetzel and Manny 1972a). In constructed wetlands this will cause problems regarding decrease of water quality, because high nitrate and ammonia concentration can stimulate the excessive growth of algae and other unwanted aquatic plants, and harm aquatic wildlife directly through toxic effects or indirect through oxygen depletion (Wang 1991; Reilly et al. 2000). On the other site macrophyte can play an important role in supplying carbon for denitrification and providing a surface for microbial attachment (van Oostrom and Russell 1994).

2.5.4 Phosphorus Cycle

Phosphorus is the key nutrient that limits primary production in many aquatic habitats. Unlike carbon and nitrogen, phosphorus is mainly found in one inorganic form, phosphate (PO_4^{3-}). Organic phosphorus occurs in a variety of organic compounds, like dissolved organic phosphorus (DOP) and particulate phosphorus (PP). The sum of DOP, PP and PO_4^{3-} is total phosphorus (TP) (Kadlec and Knights 1996; Dodds 2002).

The P dynamics (Figure 2-3) are closely related to the cycling of iron (Fe). Released P can either be sequestered by the biota or by amorphous Fe particles in either the overlying oxic sediments or the oxic part of the water column. There is still some conjecture about the mechanisms responsible for the P release in anaerobic parts of the sediments, but it is becoming obvious that the microbiota are responsible for the release of P (Baldwin et al. 1997; Roden and Edmonds 1997). A number of mechanisms have been postulated for this P release, including iron reduction, sulphate reduction, polyphosphate hydrolysis and other processes. The cycle of P in wetland ecosystems differs from that of carbon, hydrogen, oxygen and nitrogen, which have a much faster cycle. The distribution of phosphorus can be treated as a one-way traffic from rock deposits to the water and sediments. It will take millions of years to complete the cycle (Holtan et al. 1988).

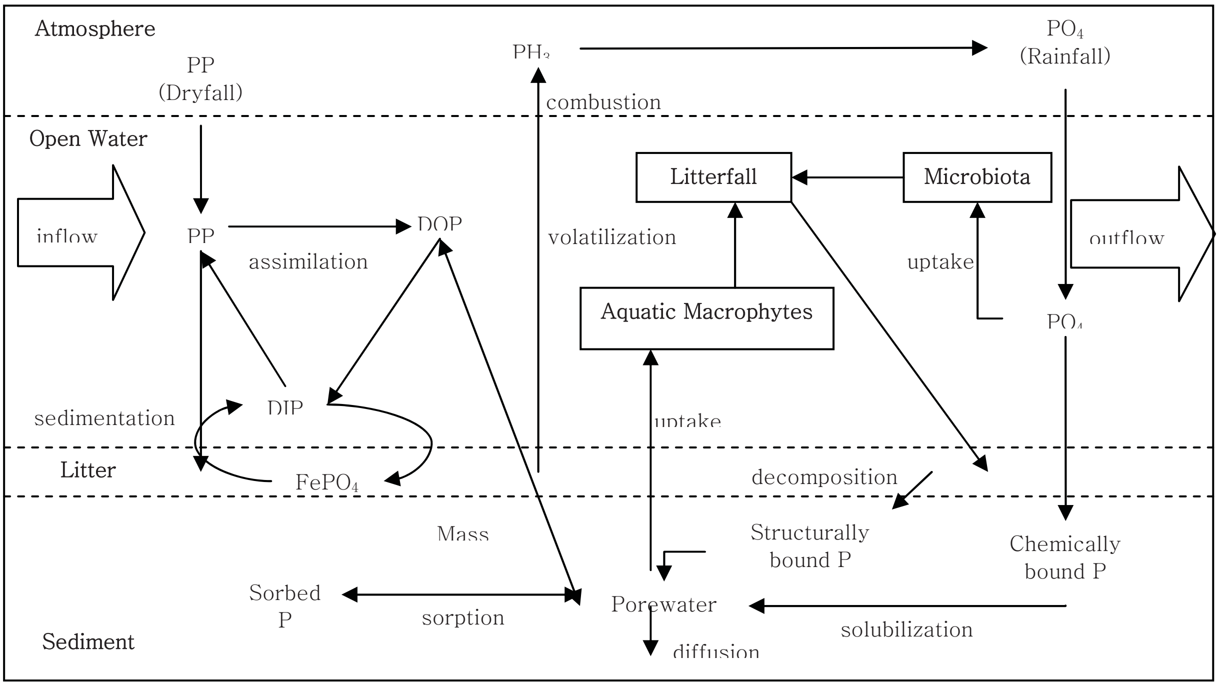


Figure 2-3: Phosphorus Cycle in Wetlands. (P = Phosphorus, PO_4 = Orthophosphate, PP = Particulate Organic Phosphorus, DOP = Dissolved Organic Phosphorus, DIP = Dissolved Inorganic Phosphorus, $FePO_4$ = Iron (III) Phosphate and PH_3 = Phosphine)

2.5.5 Contribution of Sediments and Water Plants to the Phosphorus Cycle

Sediments have a large capability to store phosphorus, which is the reason why most sediment serve as efficient “phosphorus trap”. The stored phosphorus in the sediment constitutes a vast potential nutrient source for the open water. Phosphorus release from the sediments requires that mechanisms, which transfer phosphorus to the pool of dissolved phosphorus in the pore water (1% of total sediment phosphorus). In the same time processes occur, which can transport the released phosphorus to the open water. Important mobilizations processes are desorption, dissolution, ligand exchange mechanisms and enzymatic hydrolysis. These processes are affected by a number of environmental factors, from which redox potential, pH and temperature are the most important. The regulation of the phosphorus exchange between sediment and water is based upon the interaction between iron and phosphorus during aerobic and anaerobic conditions (Kamp-Nielsen 1974, 1975b; Hosomi et al. 1981; Boström et al. 1982; Boström et al. 1988; Holtan et al. 1988; Nürnberg 1988; Jensen and Andersen 1992; Khoshmanesh et al. 1999).

During iron reduction, iron-reducing bacteria use Fe(III) oxides and oxyhydroxides as the terminal electron acceptors for anaerobic respiration. This means that these bacteria catalyse the reduction of solid ferric minerals to dissolved ferrous ions. Any phosphate ions that are associated with the solid mineral surface will be released when the surface is reduced (Figure 2-4). Fe(III) reduction mostly mediated by bacteria (Lovley et al. 1991). Recent work shows that much of the P released by the reduction of Fe(III) phases may form insoluble complexes with Fe(II) and that P can be displaced from these insoluble ferrous phosphates by sulphide (S_2^-) produced by sulphate reduction (Roden and Edmonds 1997).

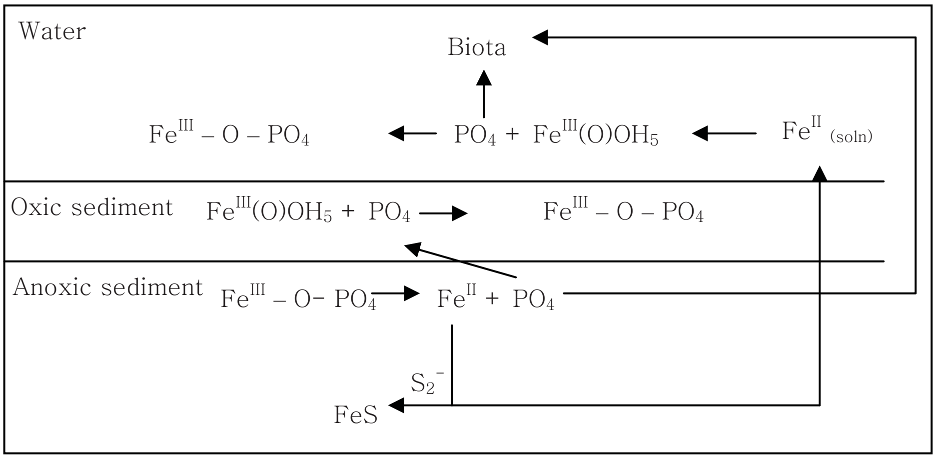


Figure 2-4: Interaction of Fe and P in aquatic sediment

Sulphate has been linked with the release of P from many aquatic systems (Boström et al. 1982; Boström et al. 1988; Caraco et al. 1989). Sulphate reducing bacteria use the sulphate ion as the terminal electron acceptor for anaerobic respiration. The respiratory end product of this reaction is hydrogen sulphide. Sulphide is a strong reducing agent to facilitate the reduction of solid ferric minerals to dissolved ferrous ions with concurrent P release (Boström et al. 1988). This reaction is favoured by the insolubility of one of the reaction products, iron sulphide. The redox conditions at the sediment-water interface are essential for the P retention capacity of the sediments (Mortimer 1941, 1942).

The iron-phosphorus and also the aluminium-phosphorus chemistry is also dependent on pH. The pH is an important controlling factor regards the P release from the sediments. An increase in positive charge of iron(III)-hydroxide with decreasing pH showed higher adsorption (Kamp-Nielsen 1974; Boström et al. 1982; Jensen and Andersen 1992). The rate of P release from the sediment will depend on pH and the initial P:Fe ratio in the sediments (Boström et al. 1982; Mortensen et al. 1994; Phillipps et al. 1994). On the other side an increase of pH in calcium-rich systems reduces the ability of the sediment to release P. Investigation showed that an increase of pH from 8 to 9 in aerobic water results in large increase in phosphate liberation (Andersen 1975). After Andersen studies the maximum net liberation was found at the pH 9-9.5 (100mg p/m^2d). When the pH values are higher than 9.5, the release rate decreased due precipitation of hydroxyapatite.

Temperature increase will influence the microbiota by increasing the bacterial activity, which increase the oxygen consumption and decreases the redox potential and pH (Kamp-Nielsen 1975a).

Essential transport mechanisms are diffusion, wind-induced turbulence, bioturbation and gas convection. All the mobilization and transport processes can theoretically contribute to the overall P release from sediments in aquatic ecosystems (Boström et al. 1982). The complex transport pathways as well as physical, chemical and biological reactions, which occur at the sediment-water interface, are important to the P cycle. A lot of studies were performed to get a better understanding of environmental factors influencing the pathways. Aquatic macrophytes, especially rooted macrophytes are acting as an intermediate link in the transport of phosphorus from the sediment to the water. Phosphorus is taken up by the roots of the plants and will be translocated to above-ground biomass from where it can be released into the water via excretion or as a result of decomposition (Boström et al. 1982; Qiu et al. 2002). The mobilization of sediment phosphorus by macrophytes can be considered quite high after (Barko and Smart 1979). The function of macrophytes as P pumps is still unclear but it is generally agreed that significant amounts of P are removed during senescence or death of old plant parts (Barko and Smart 1979; Qiu et al. 2002). But the importance of plant-mediated transport from sediment to water is dependent on the plant density and species composition. Calculations from freshwater ecosystems with a high plant density show that in such systems the phosphorus export through macrophytes is quantitatively important. In an ecosystem model of Lake Wingra the results were that harvesting removal of 50% of the macrophytes would reduce the P availability in the lake by 30%. Therefore it can be considered, that in ecosystems with rich macrophyte production, a plant-regulated transport of P and other nutrients from the sediment is an internal nutrient source. Both emergent and submersed plants are involved in this process. The type of internal loading is connected with senescence and decay of plants (Boström et al. 1982).

2.5.6 Carbon Cycle

Carbon is the currency of energy exchange in aquatic ecosystems (Dodds 2002). Inorganic carbon in water is involved in the bicarbonate equilibrium, which is connected to pH control and to acid precipitation. Organic carbon exists in various forms. The broadest classification of organic carbon is into dissolved organic carbon (DOC) and particulate organic carbon (POC). DOC is differentiated as refractory DOC and labile DOC. POC can also be divided in fine particulate organic matter and coarse particulate matter (Bolin 1981; Kadlec and Knights 1996; Dodds 2002). DOC can be divided into five major groups: Hydrophobic acids, hydrophilic acids, hydrophobic neutrals, bases and hydrophilic neutrals (Imai et al. 2001; Chow et al. 2004).

DOC is an unregulated parameter that has significant ecological and human health impacts, but has not thoroughly been studied across wetlands (Pinney et al. 2000). DOC can provide a bacterial energy source for denitrification, complex with metals and hydrophobic organics and reduce light penetration in the water. Increasingly, especially in arid regions in the world, wastewater effluent flows can represent a large fraction of the total flow in receiving water that may serve as a downstream potable water supply or recharge groundwater aquifers used as potable water supplies. Potable water treatment generally includes chemicals disinfection, and reactions between disinfectants and DOC can form carcinogenic by-products, like tri-halo-methane (Pinney et al. 2000).

DOC is produced by incomplete mineralisation of particulate organic matter or by desorption from mineral surfaces (Figure 2-5). Simultaneously, DOC production is counterbalanced by degradation, photolytic decomposition or adsorption. The mineralization proceeds slowly, so that DOC levels are typically low, but neglected mechanism of rapid DOC production can occur, which are the effects of turbulences on sediment particles, like resuspension events caused by wind or fish, during dredging of contaminated sediments or under laboratory conditions shaking of sediment (Koelmans and Prevo 2003). The concentrations of DOC in water are often greater than the concentration of POC. The DOC:POC ratios in rivers vary from 1:1 to 10:1 (Wetzel 1984). DOC commonly imparts a yellowish-brown colour to the water (Wetzel 1983; Gorniak et al. 2002), which will be considered as a decrease of water quality. The concentration of DOC (ca. 5 to >50 mgL^{-1}), (Mann and Wetzel 1995) simultaneously increases or decreases with such events, as phytoplankton or macrophyte populations decline, major precipitation events and loadings however these events are relatively short-lived and on an annual basis cause small deviations from the overall constancy. This means that the DOC:POC ratios are quite constant from year to year (Wetzel 1984).

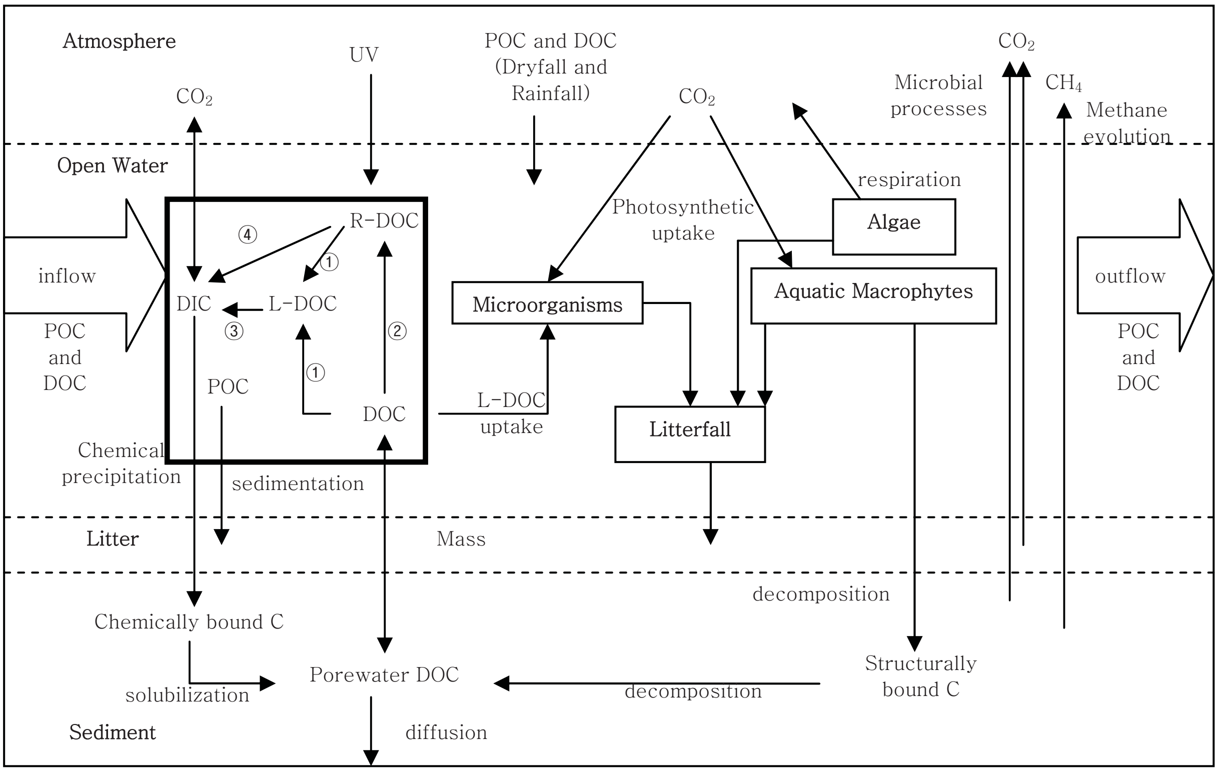


Figure 2-5: Carbon cycle in Wetlands. (DOC = Dissolved organic carbon, POC = Particulate Organic Carbon, DIC = Dissolved Inorganic Carbon, R-DOC = Refractory Dissolved Organic Carbon, L-DOC = Labile Dissolved Organic Carbon, CO_2 = Carbon Dioxide, CH_4 = Methane, C = Carbon, ① = Photodegradation of DOC, ② = DOC Immobilization, ③ = Microbial Degradation of DOC and ④ = Photoremineralization of DOC)

Precipitation and through-fall initiate the transport of allochthonous source for DOC and POC (Findlay and Sinsabaugh 2003). In the water column DOC and POC will be degraded by UV radiation (UVR). UVR has an important function in the DOC cycle, because DOC compounds, which are primarily humic substances, contain chromophores that strongly absorb UVR. While these compounds can help to shield the water column from the direct effect of UVR, they themselves will be degraded, which is considered as photochemical degradation (Strome and Miller 1978). This degradation is often accompanied by changes in the optical properties of DOC (photo bleaching). Photochemical reactions are known to reduce molecular weight of humic substances (Strome and Miller 1978; Lindell et al. 1995; Wetzel et al. 1995) and to produce inorganic carbon (Salonen and Vähätalo 1994). Although photochemical transformation of humic substances provides sources of nutrition to bacterio- and phytoplankton, the rates of photochemical reactions driven by UVR are important to other biogeochemical fluxes (Granelli et al. 1996). Increased UVR showed increased metabolic mineralization of organic carbon by bacteria (Vähätalo et al. 2003).

2.5.7 Contribution of Sediments and Water Plants to the DOC Cycle

Macrophytes are among the most productive plant communities (Wetzel 1983) and comprise a significant fraction of the total organic inputs to wetlands and many aquatic ecosystems. Macrophytes are playing a key role in regulating the C flow between sediment and water (Wilcock and Croker 2004). DOC from macrophytes can enter wetland ecosystems through the excretion of photosynthate (Wetzel and Manny 1972a) and leaching of plant materials (Wetzel and Manny 1972b). Excretion rates of 4% of net production have been estimated for aquatic macrophytes (Wetzel and Manny 1972a). Dying plants and plants cut by animals do contact the water and can leach DOC. Rhizome and root of plants also can release soluble organic carbon. Macrophytes are adapted to saturated anaerobic sediments condition and can translocate oxygen to the roots and rhizomes to support belowground respiration. Some of this translocated oxygen is released into the surrounding sediments and creates a small aerobic zone around plant roots (rhizosphere), which will lead to small changes in the sediment redox potential (Armstrong and Armstrong 1988; Boon and Sorrell 1991). Macrophyte can affect substrate utilization rates by bacteria, metabolic pathways of bacterial consortia, interstitial water chemistry and nutrient availability (Mann and Wetzel 2000). Dense stands of macrophytes in aquatic systems during summer have a major influence on water and habitat quality (Wilcock et al. 1999). Interactions between freely dissolved and bound substances, and between sediment and aqueous phases are mediated by aquatic plants through their capacity to trap fine sediments and extract nutrients from different environmental compartments (Barko et al. 1991). Plant photosynthesis and respiration cause carbon to be interchanged between inorganic and organic forms. Results have shown, that leaves from macrophytes and other plants are probably the greatest internal source for DOC (Wetzel and Manny 1972a; Wetzel and Penhale 1979; O'Connell et al. 2000), which can be utilized by bacteria. Bacterial decomposition of plant detritus has been shown to convert POC into dissolved form and cause the release of humic substances into the bulk DOC pool (Moran and Hodson 1994).

In many water quality models production of DOC is modelled as mineralization from particulate organic matter (POM), but on the other side DOC production from desiccated sediments by water turbulence may be of similar importance. Research results showed that aquatic resuspension of desiccated sediment may cause mobilization of up to 100% of indigenous particulate carbon, especially at low suspended sediment concentration. This process is fast and may out compete DOC production rates by mineralization of organic matter (Koelmans and Prevo 2003). Many studies were performed to find the sources of DOC, influences on the food web, but only little is known about the interactions between sediment and water, like exchange rates, exchange amount, factors influencing exchanges, etc. Improved understanding of the processes occurring between both compartments is needed.

2.6 Management of CWST

The management methods discussed below are methods for the improvement of water quality, i.e. methods to remove pollutants from the water so that they will not decrease the water quality.

2.6.1 Drying and Wetting

Dry and wet cycles are typical for natural shallow wetlands. In the summer seasons the wetlands dry out, and they are filled with water during the rainy seasons. Constructed wetlands have no dry and wet cycles, because the hydrological conditions can be controlled. Research on dry and wet cycles show that they influence plant germination and establishment on the edges of wetlands (Brock 1997). The effect of drying and wetting on the sediment is that partial drying of wet, (previously) inundated sediments increases the affinity for P and produces a zone for coupled nitrification/denitrification, but complete desiccation leads to a decrease of P affinity and also to a decrease of microbial activity, because through the aeration of the sediment 76% of the microbial biomass is killed (Qiu and McComb 1995; Baldwin and Mitchell 2000). After rewetting of dried sediments and soil the concentration of nitrogen and phosphorus is high, leading to an increase in aerobic bacterial activity, particularly nitrification (Qiu and McComb 1994; Qiu and McComb 1995, 1996; Baldwin and Mitchell 2000). Drying and wetting of sediments may eventually increase N loss through promotion and denitrification (Qiu and McComb 1996).

2.6.2 Drawdown

Drawdown is decline in the water level, which will change the hydrological conditions inside the treatment system. Drawdown of water levels and consequent exposure of sediment may result in partial or complete dehydration of the surface sediments of shallow wetlands. Drawdown has also been used to control macrophyte populations (Cooke 1980). The effects of drawdown are, that in some instances the drawdown produces a temporary abundance of food in the form of seeds of wetland plants, it provides suitable conditions for the establishment of emergent cover, and it may result in soil improvement and in improved aquatic plant food production upon re-flooding (Kadlec 1962). Research results have shown that this method must be used with care, because during the drawdown there will be an accumulation of phosphorus and nitrogen compounds, and during re-inundation there is an increase of the phosphorus and nitrogen concentration (Qiu and McComb 1994; Qiu and McComb 1996). This can be one reason for algal blooms after drying and refilling, also related to that may be the increased light penetration following by reduced macrophytes (Cooke 1980).

2.6.3 Plant harvesting

Nutrients removed from the inflowing water in treatment wetlands are accumulated in the sediments, released to the atmosphere (nitrogen), or taken up by aquatic plants. Macrophytes have a relatively high absorption rate of nutrients (Whigham et al. 1978; Lieffers 1983; Kim 1989; Ennabili et al. 1998). The harvesting of plants generally does not result in removal of a large quantity of chemicals, for examples nutrients; from the system unless plants like Phragmites australis are harvested several times in a growing season. The maximal removal of nutrients is through harvesting at the peak of the nutrient content and at the end of plant growth Suzuki et al. (1989)in (Mitsch and Gosselink 2000). The harvesting will stimulate the growth of plants, so that effective harvesting will remove a great amount of nutrients. Plant harvesting is necessary to control mosquitoes, reduce congestion in the water, change the residence time of the system and allow a greater plant diversity (Mitsch and Gosselink 2000).

2.6.4 Hydrological Control (Water Retention Time)

The residence time is also a management method. In most constructed wetlands the water residence or retention time is 2 to 20 days. Systems with long water residence time are generally equipped with aeration via diffusion from the atmosphere in order to support biological oxygen demand (BOD) and nitrification. Systems with short residence time receive higher quality treated wastewater (Kadlec and Knights 1996). For the treatment, of ammonia from the water, the residence time must be long. However the residence time has an impact on the release of dissolved organic carbon (DOC), which is a pollutant factor that decreases water quality. Research has shown that in systems with long residence time the concentration of DOC is greater than in systems with shorter residence time (Pinney et al. 2000).

2.6.5 Sediment Dredging

This method is a rarely used management technique in constructed wetlands. It is used for wetlands with high siltation rates, thus shortening their effective lives, and whether sediment accumulation is viewed as an undesirable feature relative to the objectives of the wetlands. Dredging is generally an expensive operation and one that should not be attempted in the life of constructed wetland. Dredging not only removes sediments, but also removes the seed bank and rooted plants (Mitsch and Gosselink 2000).

2.7 Wetland Modelling used for Decision Support

2.7.1 Computer Modelling of Ecosystems

Ecological modelling is diverse and complex. They are often difficult to investigate, especially in a quantitative manner (Kremer and Nixon 1978). Field observations and laboratory experiments are effective methods used to study ecological systems. However, it often becomes complicated, time consuming and expensive to carry out long term studies using these techniques. Computer modelling of such dynamic systems can be an alternative method of quantitatively assessing the natural environment (Kremer and Nixon 1978; Costanza and Gottlieb 1998).

2.7.2 Modelling as a Management Tool

A computer model can be a substitute for a real system and can be used for the simulation and prediction of dynamic environments (Costanza and Gottlieb 1998). The implementation of mass balance equations of biomass or concentration linked to source and sink relationships typical of ecological systems can be developed within an ecosystem model via modelling software. Using deterministic simulation and prediction models can help facilitate decision-making processes for managing freshwater systems. The value of predictive models as a decision making tool for management purposes has been widely confirmed in the literature (Straskraba et al. 1988; Mitsch and Reeder 1991; Mageu et al. 1998).

2.7.3 Wetland Modelling

Computer modelling of natural systems began in the late 1960s (Jorgensen 1988). Wetland modelling developed in the mid-1970s when sufficient data on the function of wetlands became available (Straskraba et al. 1988). Early wetland models were simple and lacked scope, as knowledge about wetland functioning was not adequate (Jorgensen 1988). As knowledge of and interest in wetlands increased, earlier computer models were improved to realistically simulate wetland function. Wetland modelling is relatively new compared to that of other aquatic areas, with the number of models published for freshwater, riverine wetlands especially low (Mitsch et al. 1988; Mitsch and Reeder 1991; van der Peijl and Verhoeven 1999).

However despite this late development, computer modelling has recently become increasingly popular for the investigation of wetland systems. Ecosystem models are widely used to simulate wetland and primary productivity functions to give predictive results for effective management decisions. Generally the simulation of phytoplankton, nutrients, macrophytes and hydrology are the basic building blocks to any wetland model. Complexity can be increased from this starting point. Many wetland models involve these basic components to predict a wide range of problems from eutrophication control to the assessment of macrophytes populations (Mitsch et al. 1988; Mitsch and Reeder 1991).

2.7.4 Non-Supervised Artificial neural networks (NSNN)

The NSNN employs non-supervised learning procedures to discover and visualize patterns, by classifying and recognising specific patterns or similarities from complex data. The internal organization of the network is totally dependant on the input variables. NSNN is in most cases a two layered network where the first layer is the input layer, with one node for each parameter. The input data then pass to every node in the second layer, which is the two-dimensional grid of nodes. Each node has a set of weights on the inputs, which are modified during the learning process (Boddy and Morris 1999). The outcome of the NSNN is the ordination and clustering of the input data. This method has the capability to discover significant patterns and features in the input data, which are comparable to the traditional Principal Component Analysis (PCA) or hierarchical clustering analysis (Jongman et al. 1987), while also having the ability to cope with non-linearity's (Boddy and Morris 1999). The patterns found in the input data are expressed by Euclidean distances, which are calculated between inputs and can be visualised as a unified distance matrix, U matrix, or as a partitioned map, K-means (Figure 2-6). The U matrix visualises the relative distances between the neighbouring data in the input data. The k-means algorithm simply partitions the input data into a specified number of clusters based on the U matrix (Recknagel et al. 2006b).

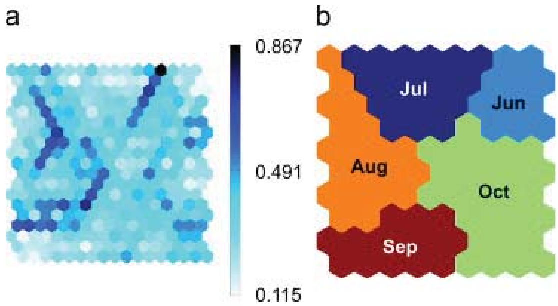


Figure 2-6: Example of unified distance map (U matrix) (a) and partitioned map (K means) (b) (from Chan et. al 2007)

2.7.4.1 Kohonnen Artificial Neural Network (KANN)

KANN (Kohonen 1982, 1984), also known as self-organising maps (SOMs), can be used to give a comprehensive view of the patterns within a data set, and it also allows a view into how the data are distributed and organized in n-dimensional data is mapped into reduced dimensional (usually 2 dimensional) space and visualized in a simplistic fashion, as patterns or clusters arranged on hexagonal lattice or grid (Giraudel and Lek 2006). KANN use competitive or self-organising learning, where neighbouring cells within the network interact and adaptively develop into detectors of a specific input pattern (Bowden et al. 2006). This adaptive nature is what gives KANN its similarity to processes within the brain and its place as an ANN (Figure 2-7).

KANN have two layers, the input layer and the Kohonen layer (Figure 2-7). The input and Kohonen layers are fully connected. Neurons in the Kohonen layer are not connected, and measure the distance of their weights to the input pattern. Competitive learning allows the neurons to compete with each other in Euclidean map space (Jeong and Joo 2003). Using this principle of competitive learning, the elements compete to respond to an input, based on distance from weights to the input pattern, and the winner, which has the smallest distance, adapt itself to respond more strongly to that input (Maier 1995). After training, data is clustered referring to certain criteria such as seasons of the year or nutrient ranges according to a calculated U matrix, and the result of this is the relational patterning of the data set, which can be graphically visualized and compared using component planes.

KANN are a very adaptable tool and can be extremely useful for studying ecological communities. KANN have shown through numerous studies that they are capable of overcoming the restrictions of conventional multivariate statistics and have been proven to be a superior tool for successful non-linear ordination and clustering of ecological data (Giraudel and Lek 2006).

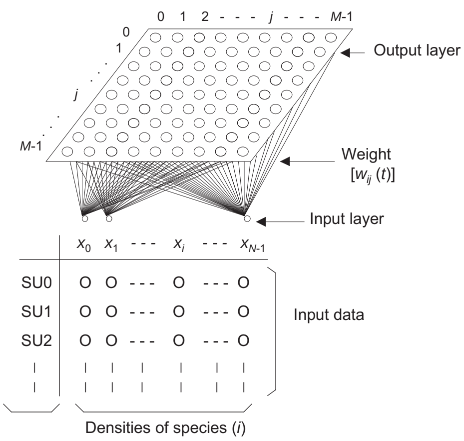


Figure 2-7: Kohonen Artificial Neural Network for non-linear cluster analysis of ecological data (From Chon et. al 1996)

Analysis and patternising of ecological community data has been successfully carried out in numerous studies (Chon et al. 1996; Lee et al. 2007). KANN has been productively applied to many different areas of aquatic ecology including fish assemblages (Brosse et al. 2001), benthic macroinvertebrate communities (Chon et al. 2000), as well as algal dynamics (Recknagel et al. 2006b). KANN can be used to make comparative assessment of periods of data, for example periods of different management regimes within a water body (Welk 2003; Whigham 2005; Recknagel et al. 2006a; Recknagel et al. 2006b; Chan et al. 2007).

2.7.5 Hybrid Evolutionary Algorithms (HEA)

The HEA is a bio-inspired machine learning model (Holland 1975), that employs principals found in processes of biological evolution, such as cross-over and mutation, natural selection and genetic variation (Cao et al. 2005a). The HEA consists of two components, which are the initial population and generation (Figure 2-8). Input variables are represented as an individual in the initial population. Individuals in the algorithm are called chromosome

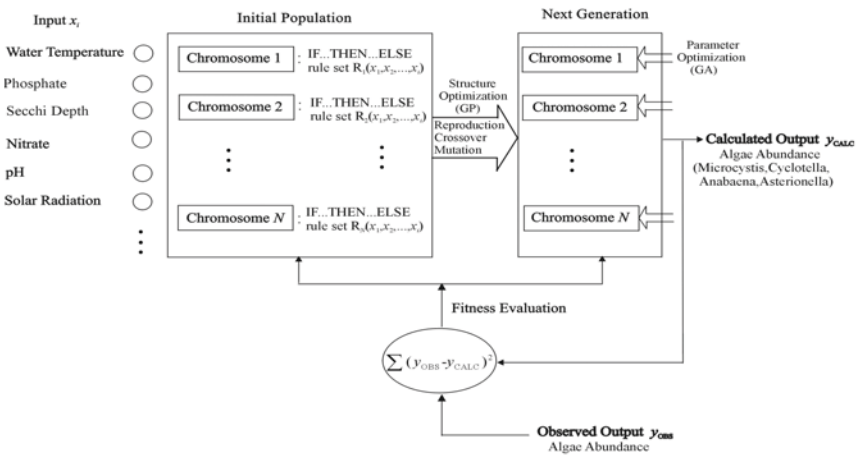


Figure 2-8: Structure of evolutionary algorithms (Cao et al. 2005b)

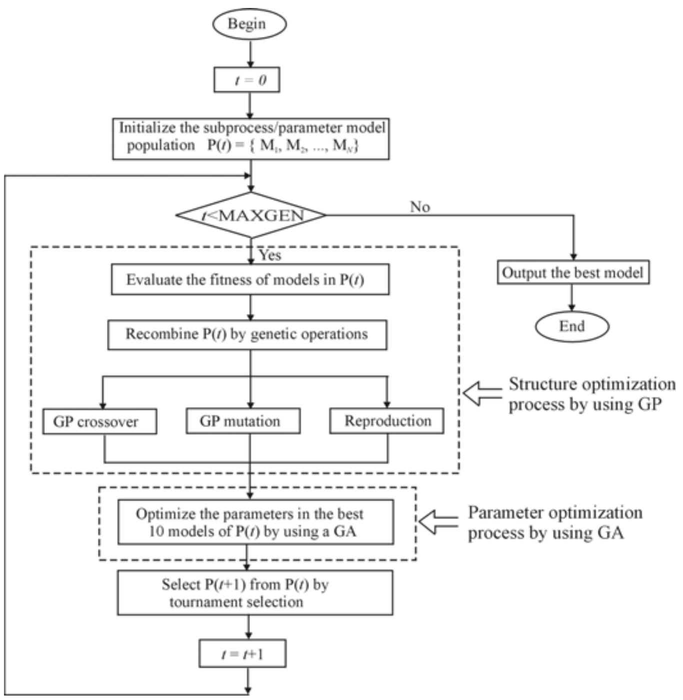


Figure 2-9: Flowchart of the hybrid evolutionary algorithm (from Cao et al. 2006)

The rules discovered by the HEA are in the form of IF-THEN-ELSE and their complexity can be controlled. The Rule set finally selected has the best fitness, which is determined by using the Root Mean Square Error (RMSE) (Figure 2-9) (Cao et al. 2006b).

In order to minimise the error (root mean square error) of output, new generations will continuously evolve based on the same approach until the defined maximum generations have been reached. The whole process started from the initial population will re-run over and over again to get the statistical significance.

Sensitivity analysis is used to assess the influence of the changes in the input variables on the output variables, which allows the examination of both the THEN and ELSE sections of the rule separately from each other to give more information about output parameters.

The HEA was applied successfully to forecast seasonal abundances of blue-green algae and diatom populations in Lake Kasumigaura (Japan) and Lake Soyang (Cao et al. 2006b). Other successful cases by applying HEA are the forecasting of phytoplankton abundance and succession in response to eutrophication control in two shallow Dutch lakes (Talib et al. 2005) and algal dynamics in Nakdong River system (Cao et al. 2006a; Kim et al. 2007)

2.8 Conclusions and aims of this study

Research on wetlands is relatively young in comparison to other ecosystems. As in the past wetland areas were thought to be not useful to humans. Many studies have been performed, including on the functions, vegetation and wildlife in wetlands. With the decrease of wetlands all over the world, humans have tried to restore and create wetlands. In many countries a great number of wetlands have been constructed and used for different purposes, but mostly the main objective of constructed wetlands is to use them as a water purifier or for water quality improvement. With the construction of wetlands another problem appears, of how to properly manage the systems. As a result many studies have been carried out to understand the processes of wetland ecosystems, such as hydrology, nutrients, sediments and vegetation. Many studies focused first of all on the vegetation and then the hydrology. The study results are sufficient to manage a constructed wetland, but more research is needed, especially on sediment and open water interactions. An important factor in regard to nutrients is the sediment and the soil, because most of the nutrients, like phosphorus, nitrogen and organic matter, which can have a huge impact on the system, are fixed in the sediments. Wetlands are mostly filled with water, and little is known about the processes or interactions of nutrients or organic matter exchange between sediment and open-water, and how they can be managed, especially in constructed wetlands receiving stormwater. These could be the main reason for many problems in constructed wetlands, so an improved understanding about the interaction between sediments and open-waters will help us to find effective management methods for constructed wetlands to improve the water quality, which is an important issue for dry countries like Australia.

The overall aim of this research is to get an improved understanding of the temporal and spatial nutrient dynamics related to the water column, sediment and plant community of the reed bed pond of the Parafield stormwater harvesting facility. In particular the nutrient interactions between the sediment and open water will be investigated, as well as the function of higher aquatic plants (macrophytes). The past a lot of research has been performed to get a better understanding of the complex processes in wetlands linked to hydrology, soil and water plants in Australia and worldwide (Andersen 1975; Kamp-Nielsen 1975b; Blackburn and Henriksen 1983; Boström et al. 1988; Boon and Sorrell 1991; Gunnars and Blomqvist 1997; Mitchell and Baldwin 1999; Baldwin et al. 2000; Madsen et al. 2001). But the understandings of these kinds of processes have to be brought into the context to develop management strategies or plans, so that the systems can be sustained.

To get these information and understandings of the processes the following hypothesis will be tested:

1. Is there in addition to allochthonous nutrient loadings a distinct amount of autochthonous loading from the wetland caused by high biomass?
The Parafield stormwater harvesting facility is a treatment wetland, therefore a great amount allochthonous loadings are to be expected of entering the system. High primary productivity in the reed bed pond in form of biomass could be the initiating source for autochthonous nutrient loadings, due to the release from the sediment caused by anaerobic conditions and re-entering of nutrients from the organic matter by decomposition (Wetzel and Manny 1972a, b; Barko and Smart 1979; Qiu et al. 2002).

2. Is there a seasonal difference in the residence time and how does it affect the removal performance of the wetland?
The residence time is been as a management method. In most constructed wetlands the water residence or retention time is 2 to 20 days. Systems with long water residence time are generally equipped with aeration via diffusion from the atmosphere in order to support biological oxygen demand (BOD) and nitrification. Systems with short

residence time receive higher quality treated wastewater (Kadlec and Knights 1996). For the treatment, of ammonia from the water, the residence time must be long. However the residence time has an impact on the release of dissolved organic carbon (DOC), which is a pollutant factor that decreases water quality. Research has shown that in systems with long residence time the concentration of DOC is greater than in systems with shorter residence time (Pinney et al. 2000).

3. What is the impact of plant harvesting in the reed bed pond regarding nutrient dynamics in the sediment and water?

Nutrients removed from the inflowing water in treatment wetlands are accumulated in the sediments, released to the atmosphere (nitrogen), or taken up by aquatic plants. Macrophytes have a relatively high absorption rate of nutrients (Whigham et al. 1978; Lieffers 1983; Kim 1989; Ennabili et al. 1998). The harvesting of plants generally does not result in removal of a large quantity of chemicals, for examples nutrients; from the system unless plants like *Phragmites australis* are harvested several times during a growing season. The maximal removal of nutrients is through harvesting at the peak of the nutrient content and at the end of plant growth Suzuki et al. (1989)in (Mitsch and Gosselink 2000). The harvesting will stimulate the growth of plants, so that effective harvesting will remove a great amount of nutrients (Mitsch and Gosselink 2000).

4. Is the function of the sediments as source or sink changing during the different seasons?

The sediments of wetlands have several functions, which are the sink, source and transformer of nutrients, but in most cases the sediment has a great potential of storing nutrients. Due to the nature of the wetland system it will be interesting to know, what the function is over the different seasons.

5. Use of HEA (Hybrid Evolutionary Algorithm) for knowledge discovery of the driving agents of the nutrients and discovering rule sets to predict nutrient concentrations.

Previously versions needed data sets of covering a certain amount of time. Due to small data sets available the HEA will be tested if it can utilize the small data set and predict the measured nutrient measurements

3 Study Site

3.1 Background

In the mid-nineties problems appeared in the Barker inlet of Gulf St. Vincent, which is a marine environment consisting of mangroves and seagrass meadows and of great importance for the State's fishing industry, caused by the uncontrolled inflows of polluted runoffs. The City of Salisbury developed a strategy to control and therefore reduce the flow of polluted water into the marine environment of the inlet. As part of that strategy wetlands were created to treat the stormwater and facilitate the rehabilitation of the Barker inlet and at the same time provide cheaper water to local industry and other users. The stormwater runoffs are now treated in a series of more than 30 wetlands along the urban stormwater paths to slow the flow and allow pollution to settle. The wetlands cover an area of 260 ha. The Parafield Stormwater Harvesting Facility, which is the study site, is the latest project to treat stormwater runoffs (City of Salisbury 2003).

3.2 Parafield Stormwater Harvesting Facility

The Parafield Stormwater Harvesting Facility (34°47'29.55"S 138°37'40.16"E) is located 25 km north of Adelaide in the City of Salisbury, South Australia. The Parafield Stormwater Harvesting Facility originated from a discussion in 1999 between the City of Salisbury and the management of G. H. Michell & Sons, Australia's largest wool processing company. This system is a constructed wetland and was built with primary purpose to collect, treat and reuse stormwater runoffs and consists of three basins: an in-stream basin, a holding storage basin and a cleansing reed bed basin (Figure 3-1).

The in-stream basin catches the stormwater runoff from the surrounding catchment area; from here the water is pumped to the holding storage basin. The water in the storage basin flows into the reed bed basin by natural gravity. The water that passes through the reed bed is pumped and stored in two adjacent aquifers. The reed bed basin is fully covered by bird netting, which prevent birds entering and contaminating the system. The total land area of this system is 11.2 ha and the catchment area is 1600 ha. The treated water is stored in an aquifer storage and recovery system (ASR). Two ASR have been installed, which allow the supply of water when the system has no flow (Table 3-1) (City of Salisbury 2003).

NOTE:
This figure is included on page 43 of the print copy of the thesis held in the University of Adelaide Library.

Figure 3-1: Schematic section of the Parafield Stormwater Harvesting Facility (City of Salisbury 2003)

Table 3-1: Snapshot of Parafield Stormwater Harvesting Facility (City of Salisbury 2003)

NOTE:
This table is included on page 44
of the print copy of the thesis held in
the University of Adelaide Library.

3.2.1 Catchment Description

The Parafield stormwater wetland system will be supplied with water from an area of 1600 ha. The catchment area is mainly residential area with a small number of parkland areas and an industrial area, which is located north of the wetland (Figure 3-2). The catchment area is serviced with a complex stormwater drainage system for the collection of stormwater runoffs, which is mainly street and roof drainage.

3.2.2 In-Stream Pond

The in-stream pond is the first pond of the three pond system. The in-stream pond functions as a sedimentation pond and therefore no vegetation was planted. The stormwater runoff will be pumped from the drainage system into the in-stream pond. Most of the suspended solids will be removed in this pond and accumulates in the pond, which will be removed. The water will be pumped then from the in-stream pond to the higher located holding (storage) pond.

3.2.3 Holding (Storage) Pond

The holding (storage) pond is the second pond and has the function to store the water and supply the water for the next located reed bed pond on demand. This pond also has no vegetation, but is different in comparison to the other two ponds, which is that the storage pond has bigger capacity. The water depth in this pond can reach up to 3.5m at its full capacity.

NOTE:
This figure is included on page 45 of the print copy of the thesis held in the University of Adelaide Library.

Figure 3-2: Parafield and Ayfield stormwater catchments showing main stormwater drains and land use (Swierc et al. 2005)

3.2.4 Reed Bed Pond

The reed bed pond is the last pond in this treatment system. The difference to the other ponds is the existence of vegetation, which has planted. The reed bed was originally planted in parallel rows of *Phragmites australis*, *Eleocharis sphacelata*, *Schoenoplectus validus* and *Baumea articulata* (Figure 3-3). Alongside the listed species *Typha orientalis* is present in the reed bed pond, which has been introduced on the natural way. The dominant species in the reed bed is overall *Phragmites australis*.

3.2.5 Water Management

The Parafield system is an example of a highly sophisticated artificial wetland. Water flow into and through this wetland is highly regulated. Stormwater can be diverted into the top of the system (the in-stream basin) via a weir. In times of very high flow, the wetland can be isolated from inflowing water. The main purpose of the in-stream basin is to slow water flow to allow sedimentation of suspended matter. Water is then pumped to the holding storage, from where it is released into the reed bed at intervals. Water flow in the reed bed is regulated to ensure a retention time of ten days and a water level of 30-70 cm. Water is seasonally (mainly in winter) pumped from near the outlet site of the reed bed into underground aquifers for storage. Water from the reed bed outlet and from the aquifer storage is supplied to the adjacent suburb of Mawson Lakes as a secondary reticulated water service and is also used in a nearby wool washing operation. Users of this water pay for it, but at a reduced cost compared to fully treated water.

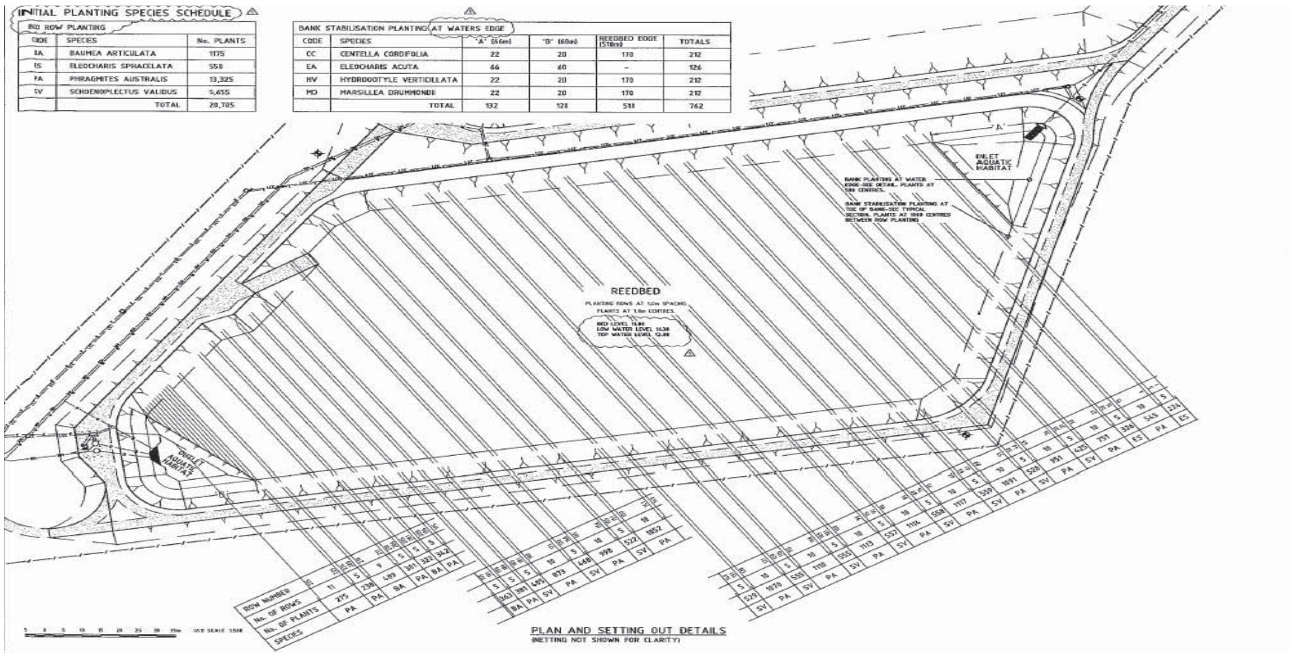


Figure 3-3: Plant species composition in the reed bed pond of the Parafield Stormwater Harvesting Facility (City of Salisbury)

4 Material and Methods

4.1 Monitoring

The monitoring of the Parafield stormwater wetland system was performed on the storage and reed bed pond with the main focus on the reed bed pond to get more information about the temporal and spatial nutrients dynamics.

Therefore the monitoring was divided into three sections, which are the monitoring of the water column, sediment and macrophytes (Figure 4-1), which will discussed in more detail in the following sections of this chapter.

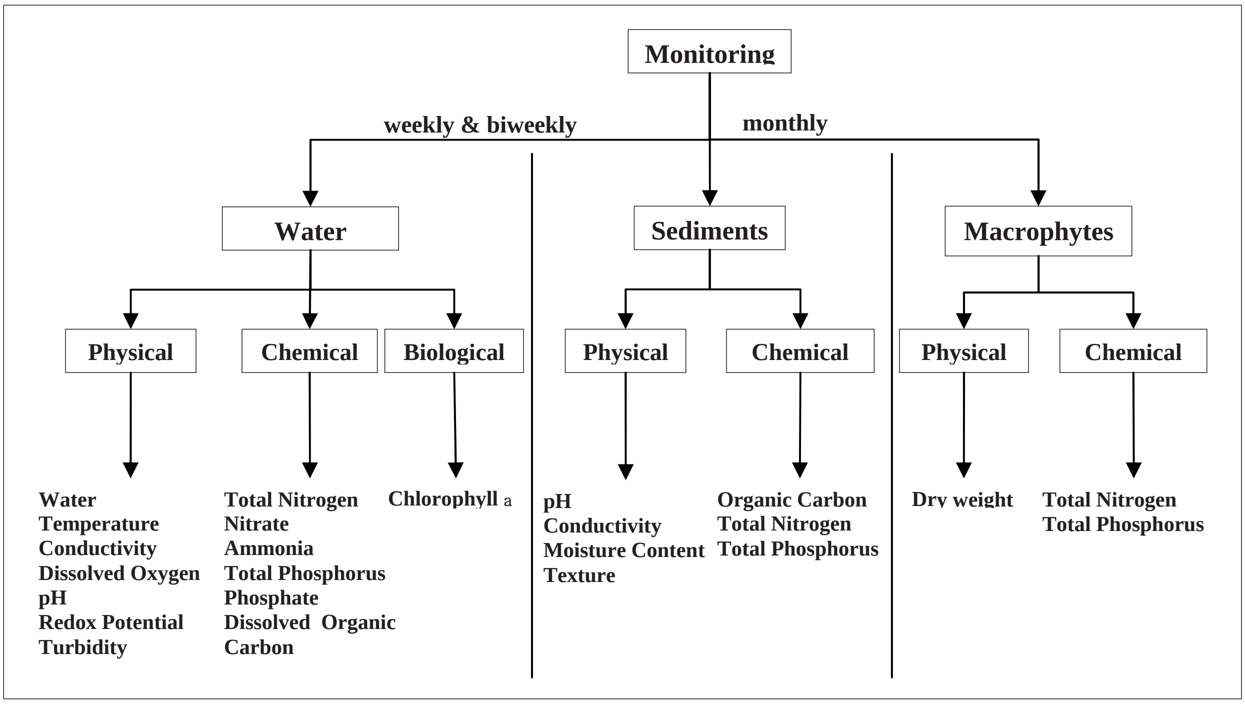


Figure 4-1: Monitoring plan for the Parafield stormwater wetland system

4.2 Sampling Sites

A total amount of seven collection sites were selected, from which six sites were located in the reed bed pond (Figure 4-2) and one collection site located at the outflow area of the storage pond, from which the water will flow into the reed bed pond.

At each collection sites monitoring and sampling was performed in order to gather information of physical, chemical and biological parameters for the water column, physical and chemical parameters for the sediment and physical and chemical parameters for the plant community (Figure 4-1).

The collection sites were selected based on their location within the system and in the direction of the water flow to get a better understanding of spatial differences in regards nutrient dynamics and removal performance. Another criteria for the selection was accessibility, because of the high plant density certain areas of the reed bed pond were difficult to access and would have also involved a lot of disturbance.

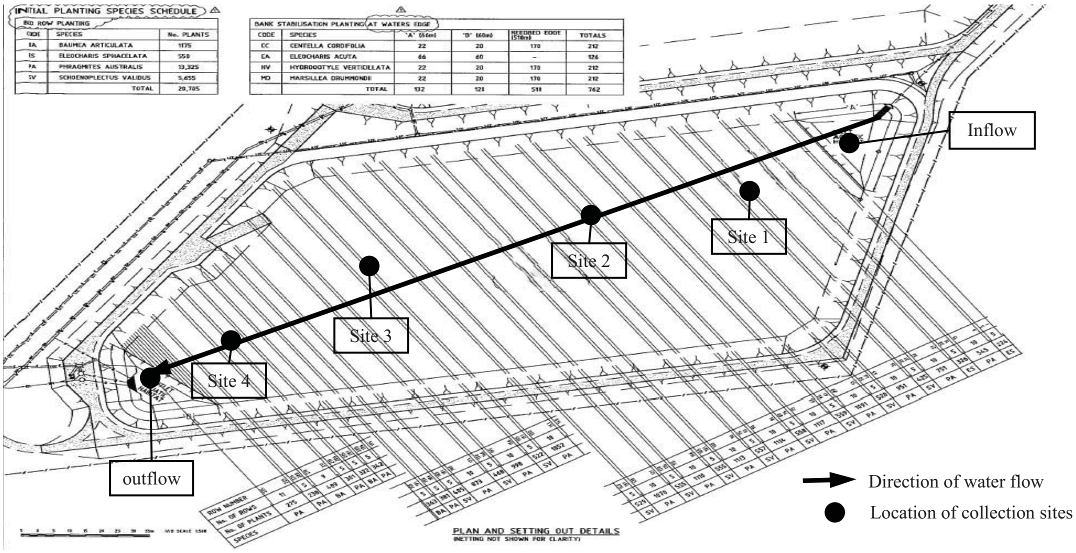


Figure 4-2: Location of collection sites in the reed bed pond

Certainly the number of sites could have been increased by creating transects running perpendicular to the flow, which would allow to gather more information of the side areas. Considering the similarity of conditions at the area of the collection sites the focus was the middle area of the wetland to get an understanding of the temporal and spatial differences of nutrient dynamics within the reed bed pond.

4.2.1 Inflow Site

The inflow site covers an area of around 520 m^2. The collected stormwater will enter the reed bed pond, which will be supplied from the higher located storage pond. The inflow of water from the storage to reed bed pond will be regulated by the water level condition in the reed bed. The deepest point of the inflow site has a depth of 1.5 m. It is an open water area with no vegetation.

4.2.2 Reed bed Sites

Site 1 is the site located closest to inflow, which is vegetated, consisting mainly of *Phragmites australis*, *Eleocharis sphacelata*, *Schoenoplectus validus* and *Typha orientalis*. The plant composition at site 2 is the same compared to site 1 with the exception of *Eleocharis sphacelata*. Site 3 was vegetated mainly with *Phragmites* *australis* and *Typha orientalis*, but lost its vegetation due to the harvesting procedure. Site 4 is the closest to outflow site and has a totally different plant composition compared to the other sites. The vegetation is dominated by *Baumea articulata*.

4.2.3 Outflow Site

The outflow site covers an area of around 350 m^2. The outflow site is the similar to the inflow site. The deepest point has a depth of 1.5m and like the inflow it is an open water area with no vegetation. The “treated” stormwater will leave the system to be supplied to the residential and industrial consumers or to recharge the aquifer layers.

4.3 Water Column

The monitoring of the water column included the measurements of physical, chemical and biological parameters. The monitoring of the different parameters was divided into two parts. The first part is the measurements of parameters on-site and the second is to collect samples, bring them back to the laboratory for analysis.

4.3.1 Collection of Samples

The water samples for the analysis of the chemical and biological parameters were collected in black plastic bottles, which were acid washed (12 – 24 hours in 10% HCl), rinsed with deionized water and dried prior to the fieldtrips. The water samples were collected during the first year (March – November 2005) in a weekly interval and from the second year onwards (February 2006 – August 2007) in a biweekly interval. All the water samples were taken at a depth of 50 cm at the inflow and outflow site. The water samples at the reed bed sites were collected at the sediment and water interface. The volume of the collected samples varied between 0.5L and 1.0L.

4.3.2 Preparation of Water Samples

The water samples brought back to the lab were divided into water to be filtered and the water samples to be not filtered. The water samples will be filtered by using a GF/C (Φ 47mm) filter paper.

4.3.3 Physical Parameters

The physical parameters were measured on-site by using a YSI-multiprobe (YSI model 6820) with the following parameters can be measured: water temperature (ºC), conductivity (mS/cm), dissolved oxygen (mg/L), pH, redox potential (mV) and turbidity

(NTU). The measurements for the physical parameters were taken from the sediment – water interface and total number of six measurements (n=6) were taken.

4.3.4 Chemical Parameters

The chemical parameters, which were measured, were the nutrients, like total nitrogen (TN), nitrate (NO_3^-), ammonium (NH_4^+), total phosphorus (TP), phosphate (PO_4^{3-}) and dissolved organic carbon (DOC). Except ammonium all the other nutrients had to be analysed in the laboratory.

4.3.4.1 Total Nitrogen (TN)

All forms of nitrogen (N) is found in natural waters and wastewaters, which are biochemically inter-convertible under optimal conditions. The N forms of the greatest interests are nitrate (NO_3^-), nitrite (NO_2^-) and ammonium (NH_4^+). The TN concentration of the water samples were analysed using autoanalyser. The TN concentration in the water samples were determined by using the semi-micro Kjehldahl method (Kjeldahl 1883) in combination with colorimetric determination (Rayment and Higginson 1992). Prior to the analysis water samples were boiled down at 100ºC in an oven and digested in the presence of sulphuric acid on a Tecator digestion system. The digestion steps are at 150ºC for 20 minutes, 250ºC for 20 minutes and 330ºC for 2 hours. The digestion block used is a Tecator 1016 Digester and the steps were controlled by a Tecator Autostep 1012 Controller. Three replicates for each sample was performed.

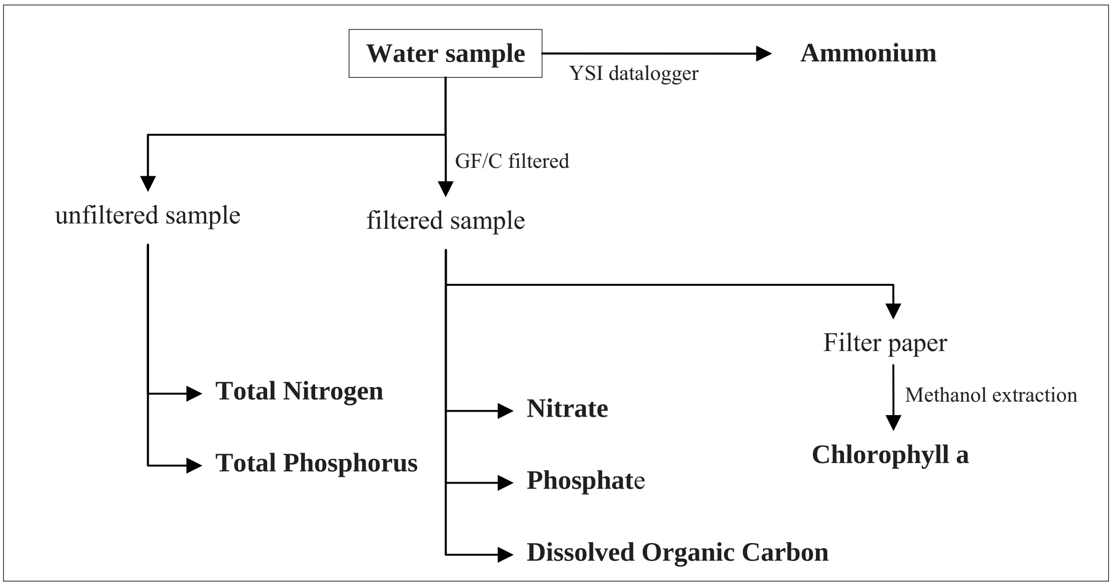


Figure 4-3: Schematic protocol for analysis of chemical and biological parameters in water

4.3.4.2 Ammonium

The ammonia concentration was measured with the YSI multiprobe (YSI model 6820). A total number of five measurements were taken from the sediment – water interface.

4.3.4.3 Nitrate

The concentration of nitrate was analysed with a HACH spectrophotometer (DR-2000) using the cadmium reduction method. 25 ml of the filtered water is used as the sample and 25 ml of deionised water is used as the control. A reagent powder pillow (Nitra Ver 5) is added to both. The cadmium metals in the powder pillow will reduce the nitrate in the samples to nitrite. The nitrite ion reacts in an acidic medium with sulfanilic acid to form an intermediate diazonium salt which couples to gentisic acid to form an ambered-colored product. The spectrophotometer will detect the colour difference due to absorbance to estimate the nitrate concentration (HACH 1997). Three replicates were performed for each sample.

4.3.4.4 Total Phosphorus

Phosphorus (P) is present in many forms, including inorganic and organic and soluble and insoluble forms. (Platell and Jack 1974) suggest that total P is the only definitely form in waters. The levels of P found in Australian natural waters are generally not of significance for domestic or agricultural uses, but is used as an indicator of pollution by runoff from agricultural areas or by domestic sewage. Levels of TP may vary from 0.01 to 10 mg L-1 depending on season, biological activity and the level of pollution (Rayment and Higginson 1992).

Prior to the analysis the water samples were boiled down at 100ºC in an oven and digested in the presence of nitric and perchloric acid on a Tecator digestion system. The digestion steps are at 70ºC for 30 minutes, 150ºC for 4hours and 180ºC for 4 hours. The digestion block used is a Tecator 1016 Digester and the steps were controlled by a Tecator Autostep 1012 Controller. Three replicates for each sample was performed.

4.3.4.5 Orthophosphate (Reactive Phosphorus)

The concentration of phosphate was analysed with a HACH spectrophotometer (DR-2000) using the ascorbic acid method. 25 ml of filtered water is used as the sample and 25 ml of the filtered water is used as the control. A reagent powder pillow (Phos Ver 3) is added only to the sample. The orthophosphate in the water sample will react with the molybdate in an acid medium to produce a phosphomolybdate complex. Ascorbic acid then reduces the complex, which will result in a molybdenum blue colour (HACH 1997). Three replicates were performed for each sample.

4.3.4.6 Dissolved Organic Carbon (DOC)

The DOC concentration was measured with a SGE ANATOC II total carbon analyser. Prior to the analysis the water samples will be filtered using GF/C filter paper and were acidified to pH 2 by using 1M perchloric acid. After that the samples will be stored at 4ºC in a dark environment to prevent breakdown of DOC by UV. Before the analysis samples were kept under room temperature conditions (20 ºC) and the pH was adjusted to a pH range between 2.5 and 2.8 by using 0.1M sodium hydroxide (NaOH) and 0.1M perchloric acid. The DOC analysis was performed in TOC mode using titanium dioxide as a catalyst in the presence of UV light. Benzoic acid was used as a DOC standard for the calibration of the analyser. Three replicate measurements of DOC were made for each sample.

4.3.5 Biological Parameters

The biological parameter was monitored in form of the chlorophyll a (chl-a) concentration in the water column.

4.3.5.1 Chlorophyll a

The chlorophyll concentration will be measured by filtering a known volume of water with a GF/C filter paper. The filter paper will put into 10 ml centrifuge tube and methanol will be added, which acts as an extraction agent. The tube will be kept in a cold and dark environment for 24 hours. After centrifuging the tubes for 10 minutes at 6000 rpm the chlorophyll will be determined by using a Hitachi Spectrophotometer 1000. To determine the chl-a concentration the absorbance at two wavelengths (665nm and 750 nm) will be measured. For the calculation of the chl-a the following equation is used (Golterman et al. 1978):

$$\text{chla}\,(\mu\text{g/ml}^{-1}) = 13.9 \times E_1 \tag{4-1}$$

$$E_1 = \text{abs}\,665\text{nm} - \text{abs}\,750\text{nm} \tag{4-2}$$

4.4 Sediment

The monitoring of the sediments included the measurements of physical and chemical parameters. All the sediment samples were collected on-site and had to be brought back to lab for analysis.

4.4.1 Collection of Samples

The sediment samples were collected based on random sampling with a sediment core sampler (Φ 50cm) from each site. Each sediment core had collected a sediment sample up to a depth of 15 cm. After collection, the sediment core was brought back to the lab for preparation and analysis.

4.4.2 Preparation of Samples

The sediment core samples brought back to the lab had to be divided into fresh sediment and air-dried sediment samples. Depending on the parameter, which has to be analysed the sediment sample had to be either fresh or air-dried (Figure 4-4). For the majority of analysis air-dried sediment was used, which had been dried for a time period of one week under room temperature conditions. After the drying procedure the sediment were sieved using a sieve with the mesh size 2mm. The fresh sediment samples were kept and stored at a temperature of 4 ºC in a cool room and were brought back to room temperature condition before the analysis.

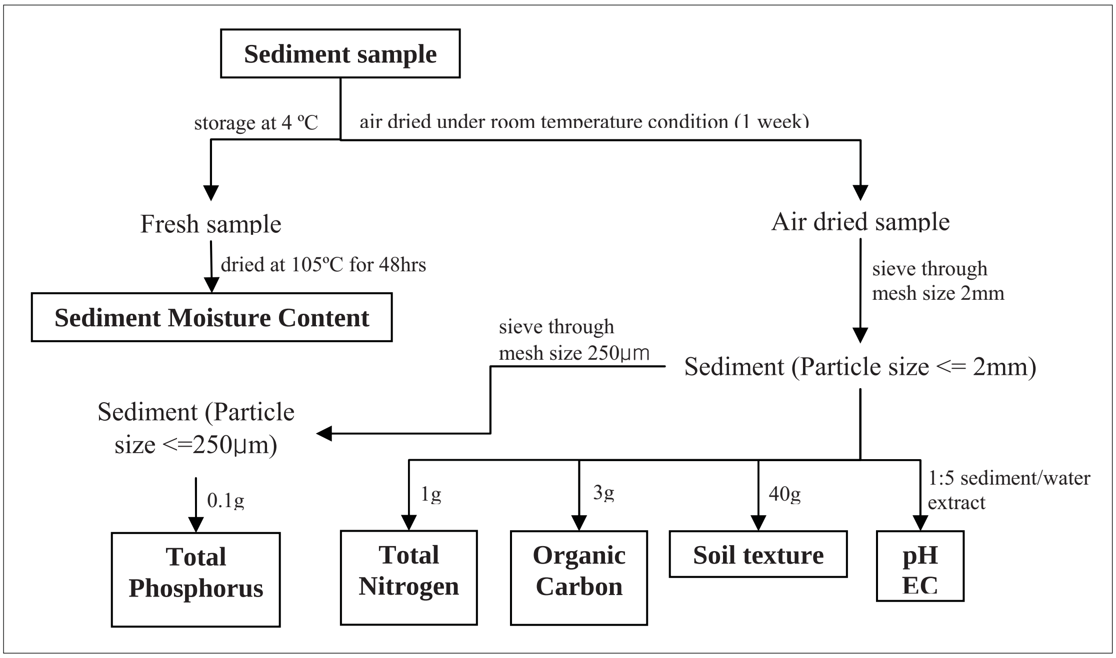


Figure 4-4: Schematic protocol for analysis of physical and chemical parameters in sediment

4.4.3 Physical Parameters

The physical parameters which were measured are: Sediment moisture content, pH, electrical conductivity and texture.

4.4.3.1 Sediment Moisture Content (SMC)

Sediment Analysis is performed on the basis of oven dry weight, so the moisture content of fresh sediment is done prior to the analysis. For the measurement of the moisture content 20-50g of fresh sediment was dried for 48 hours at 105ºC and after that the decrease of sediment weight was measured by using the following equation (Rayment and Higginson 1992):

$$\text{SMC}(\%) = \left(\frac{(\text{fresh sediment wt. - dry sediment wt.})}{\text{dry sediment wt.}}\right) \times 100 \qquad (4\text{-}3)$$

4.4.3.2 pH

The pH value of sediment indicates of acidity which influences sediment conditions and plant growth. The effects include changes in the solubility and acidity of various biologically important elements and processes. To measure the pH air-dried sediment will be mixed with distilled water in a relationship of 1:5. The next step is to shake the sediment/water mixture for one hour. After the shaking procedure allow the sediment to settle down around 20-30 minutes (Rayment and Higginson 1992). The pH was measured with pH meter (HI 8521).

4.4.3.3 Electrical Conductivity (EC)

Electrical Conductivity (EC) of sediment suspension is used to estimate the concentration of soluble salts in the sediment. The soluble salts consist mainly of cations, like Na^+, Mg_2^+ and Ca_2^+, and anions such as Cl^-, SO_4^{2-} and HCO_3^-. Therefore high electrical conductivity corresponds to high concentrations of soluble salts in the sediment (Rayment and Higginson 1992). The preparation and procedure to measure electrical conductivity is the same like pH (refer to section 4.4.3.2 pH). The electrical conductivity was measured in mS/cm (TPS 2100).

4.4.3.4 Soil Texture

Soil texture is a basic physical parameter, which describes the relative proportions of different grain sizes of mineral particles in the soil. Particles are grouped according to their size, which is called soil separates. Soil particles smaller than 2mm are classified into three major size groups: sand, slit and clay. Using the classification system used by the U.S. Department of Agriculture (USDA) particles smaller than 0.002mm are considered as clay, particles between 0.002-0.05 mm are considered silt and particles between 0.05-2mm are considered as sand. The analysis was performed and calculated after (Sheldrick and Wang 1993).

4.4.4 Chemical Parameters

The chemical parameters measured are in form of organic carbon, total nitrogen and total phosphorus.

4.4.4.1 Total Carbon

Estimates of organic carbon are being used to assess the amount of organic matter in sediment (Rayment and Higginson 1992). The method which was chosen to determine the organic carbon content is the wet oxidation (Walkley 1947) in combination with Black's rapid titration procedure, or better known as the Walkley & Black method. In this procedure, concentrated sulphuric acid (H_2SO_4) is added to sediment wetted with a dichromate ($Cr_2O_7^{2-}$) solution. Heat of the dilution raises the temperature to around 110-120 ºC which is sufficient to induce substantial oxidation. The reaction is shown in the following equation:

$$2Cr_2O_7^{2-}\text{(orange-red)} + 16H^+ + 3C \rightarrow 4Cr^{3+}\text{(green)} + 8H_2O + 3CO_2$$

A known quantity of dichromate is the source of the chromic acid and the excess of chromic acid not reduced by the organic matter of the sediment is determined by subsequent titration with standard ferrous sulphate using o-phenanthroline as an indicator. Three replicate measurements were performed for each sample.

4.4.4.2 Total Nitrogen (TN)

Nitrogen is a primary component of all living matter. The TN content in soils ranges from <0.02% in subsoils to >2.5% in peat's (Bremner 1996). Most soil N is associated with organic compounds including proteinaceous forms from plant, animal and microbiological residue. Most of soil N can undergo chemical and biological transformation under optimal conditions. Important inorganic forms of soil N involved in these reactions are nitrous oxide (N_2O), nitric oxide (NO), nitrogen dioxide (NO_2), ammonia (NH_3), ammonium (NH_4^+), nitrite (NO_2^-) and nitrate (NO_3^-).

The TN concentration in the sediment samples were analysed by following the method of wet oxidation (Kjeldahl 1883), which is also called Kjeldahl method. In our case the method of semi-micro Kjeldahl in combination with colorimetric determination was selected. During the semi-micro Kjeldahl method the TN content in the sample is converted into NH_4^+-N by digestion in the presence of a catalyst. The digestion procedure was performed in the presence of sulphuric acid at 150ºC for 20 minutes, 250ºC for 20 minutes and 330ºC for 3 hours. The digestion was performed by a Tecator digestion system having the digestion block (Tecator Digester 1016) and autostep controller (Tecator Autostep 1012 Controller). The colorimetric determination was performed by a Technicon autoanalyser. Three replicate measurements were taken for each sample.

4.4.4.3 Total Phosphorus

The TP concentration in soils is in a range from 200 to 5000 mg P kg-1. Phosphorus exists in soil as organic and inorganic P. The TP concentration in the sediment was analysed by a Technicon Autoanalyzer after a nitric-perchloric acid digestion at 160ºC for 6 hours.

4.5 Macrophyte

The monitoring of the plant composition included the measurements of physical and chemical parameters. All the plant samples were collected on-site and had to be brought back to lab for analysis.

4.5.1 Collection of Samples

The collection of the plant samples were based on random sampling from each site. At each site a 30cm x 30cm plot was constructed and all the aboveground biomass was removed by cutting the plants. The cut plants were put in plastic bags and transported back to the lab for the preparation of the samples for further analysis.

4.5.2 Preparation of Samples

The plant samples brought back to the lab were dried at 50ºC for 48hours. After the drying procedure the plant samples were divided into the different parts of the plant species, which are the leaves, stems and flowers.

4.5.3 Physical Parameters

The physical parameters were measured in form of the dry weight for the sampling plot. The dry weight was measured after the drying process, which is mentioned the section of the preparation of samples.

4.5.4 Chemical Parameters

The chemical parameters were measured in form of total nitrogen and total phosphorus.

4.5.4.1 Total Nitrogen

For the analysis of TN the method of semi-micro Kjeldahl in combination with colorimetric determination was chosen.

The TN concentration in the sediment samples were analysed by following the method of wet oxidation (Kjeldahl 1883), which is also called Kjeldahl method. In our case the method of semi-micro Kjeldahl in combination with colorimetric determination was selected. During the semi-micro Kjeldahl method the TN content in the sample is converted into NH_4^+-N by digestion in the presence of a catalyst. The digestion procedure was performed in the presence of sulphuric acid at 150ºC for 20 minutes, 250ºC for 20 minutes and 330ºC for 3 hours. The digestion was performed by a Tecator digestion system having the digestion block (Tecator digester 1016) and autostep controller (Tecator Autostep 1012 Controller). The colorimetric determination was performed by a Technicon autoanalyser. Three replicate measurements were taken for each sample.

4.5.4.2 Total Phosphorus

For the determination of TP we used colorimetric determination using vanadate and molybdate (Kitson and Mellon 1944; Hanson 1950).

Spectrophotometer was used to determine the TP concentration at a wavelength of 390 nm. Prior to the analysis the plant material had to be digested in the presence of nitric-perchloric acid at 70ºC for 30 minutes, 150ºC for 4 hours and 180ºC for 4 hours.

4.6 Data Analysis

4.6.1 Data Pre-processing

The raw data collected during the monitoring process were pre-processed before running them on NSNN and HEA. Because of the short data set the data for sediments and plants, which were collected in monthly intervals, were linearly interpolated into weekly intervals, to increase the data points and also to match the interval of data regards the water sampling. Cao (2006) found that the interpolated data did not increase the error levels when used to develop HEA rule sets, but that they produced better results than the raw data.

The classification of the different seasons was separated like in the following table:

Table 4-1: Classification of the Seasons

Seasons	Periods
Spring	1 September – 30 November
Summer	1 December – 28 February
Autumn	1 March – 30 May
Winter	1 June – 30 August

4.6.2 Non-supervised Neural Network (NSNN)

The SOM maps were used with the purpose of ordination, clustering and visualisation of the seasonality of the water quality, sediment and plant data. The analysis of the data and creation of the SOMs were performed using the SOM toolbox with the program Matlab Version 7.0.

4.6.3 Hybrid Evolutionary Algorithm (HEA)

HEA (Cao et al. 2006b) was used to develop rule equations to calculate the output parameter in form of the chemical nutrients and to find out the key parameters driving the nutrients at different site locations. All HEA experiments were performed on a Hydra supercomputer (IBM eServer 1350 Linux) using the C programming language.

The details of the parameter setting of HEA (Figure 4-5) consist of 300 initial populations, with a maximum number of 200 generations and repetitive runs of 100.

Because of the small nature of the dataset the following method, called “Bootstrap method” was used to discover the rule sets using the HEA. This means that the HEA selects the training data randomly. In this experiment the 80% of the total data set has been used for the training and 20% for the testing.

Prediction models were developed to determine the different nutrient concentrations at the outflow area, by using selected input variables from the inflow site. Based on this experiment a 7 days ahead forecasting model for the different nutrient forms have been developed.

Prediction models for knowledge discovery were developed for inflow, site1 – site 2, site 3 – site 4, and outflow for the different nutrients forms. The data for site1 – site 2 and site 3 – site 4 was merged into one matrix to produce a rule set predicting nutrient conditions of two sites.

Structure Optimization (GP)	Seed number (population) = 300 Stop number (MAXGen) = 200 F_L = {AND, OR} $F_C = \{>, <, \geq, \leq\}$ F_A = {+, -, *, /, exp, ln} MAXK = 4 $D_{IF} = D_{THEN/ELSE} = 4$ Repetitive Runs = 100 Training = 80% Testing = 20%

Figure 4-5: Parameter setting of HEA for rule set discovery (F_L = logic function set, F_C = comparison function set, F_A = arithmetic function set, MAXK = maximum size of rule set, D_{IF} and $D_{THEN/ELSE}$ = maximum tree depth for IF and THEN/ELSE tree)

4.6.4 Statistical Analysis

Statistical Analysis was performed by using Origin Pro 7.5 program.

4.6.4.1 Analysis of Variance (ANOVA)

ANOVA is a statistical tool based on F ratios which measures if a factor contributes significantly to the variance of a response. Additionally it determines the amount of variance, which is due to error. ANOVA tests the null hypothesis, which considers all populations to be equal: H0: $\mu 1 = \mu = \ldots = \mu a$ by comparing two estimates of variance (δ^2). δ^2 is the variance within each of the “a” treatment populations. The first estimate (called the Mean Square Error or “MSE”) is based on the variance within the samples. The second estimate (Mean Square Between or MSB) is based on the variance of the sample means. The MSB is only an estimate of δ^2 in case the null hypothesis is true. If the null hypothesis is false then MSB estimates something larger than δ^2. The logic by which the analysis of variance is performed, is to test the null hypothesis, which is as follows: If the null hypothesis is true, then MSE and MSB should be close to same since they are both estimates of the same quantity (δ^2); however, if the null hypothesis is false then the value of MSB can be expected to be larger than MSE since MSB is estimating a quantity larger than δ^2.

Tukey’s test was used to determine statistically significant differences among means, based on the range distribution. In the current study, if ANOVA reveals that the means are significantly different, Tukey’s test was performed to determine which of the means are significantly different from the overall mean.

5 Results

5.1 Meteorological Data

The Adelaide region is considered to be a temperate/mesothermal (Mediterranean climate) climate, based after the Köppen climate classification. The weather in winter and springs month, between June to November, tends to be wetter and there is higher chance of rainfalls. The summer months of December, January and February are generally very hot with sporadically rainfalls. The weather does not have a distinct winter as such; instead it is more like an extended autumn over a six months period (Figure 5-1).

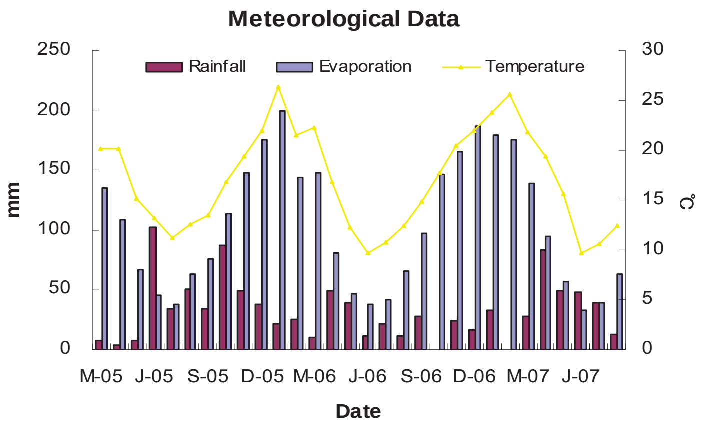


Figure 5-1: Meteorological Data for rainfall, temperature and evaporation for Parafield Airport from March 2005 until August 2007

The average temperature during the whole study period (March 2005 until August 2007) was 17ºC (max. temperature: 33.6ºC and min. temperature: 3.6ºC). The total amount of rainfall was 968.4mm (average rainfall per month: 32.3mm) and the total amount of evaporation was 4080mm (average evaporation per month: 136mm).

The source of the data for temperature and rainfall was provided by the Australian Bureau of Meteorology (www.bom.gov.au), which was downloaded as excel spreadsheet showing the daily measurements, which was averaged to a monthly interval. The data for evaporation was provided NRM-Enhanced Meteorological Datasets (www.longpaddock.qld.gov.au/silo/). The data was received as a text file converted into an excel spreadsheet and like the temperature and rainfall data the measurements were averaged into a monthly interval.

5.2 Comparison of Water Quality Conditions

5.2.1 General Physical, Chemical and Biological Characteristics

Table 5-1, 5-2 shows the comparison of the average, standard deviation (S.D.), standard error (S.E.) and the range (minimum and maximum) of measured chemical (TN, NO_3^{2-}, NH_4^+, TP, PO_4^{3-} and DOC), physical (water temperature, conductivity, DO, pH, ORP and turbidity) and biological (chl-a) water quality parameters from the different collection sites.

To compare the variances of the measured parameters an analysis of variance (ANOVA test), a statistical tool based on F ratios which determine if either a factor contribute significantly or insignificantly to the variance of a response, was performed. If the ANOVA test confirmed a significant difference between the collection sites a Tukey's test was performed. The results of ANOVA and Tukey's test (post-test) are shown in the Table 5-3 for the water quality parameters.

The analysis regards the temporal and spatial differences showed significant differences between the location of the different sites and based on if the areas were vegetated or non-vegetated. The general trend showed that for most of the measured water quality parameters were decreasing from the storage pond towards the outflow area. It also revealed significant differences between non-vegetated (storage, inflow and outflow) and vegetated sites (site 1, site 2, site 3 and site 4). Significant differences between sites near inflow (inflow, site1 and site 2) and outflow (site 3, site 4 and outflow) were observed as well (Table 5-1, 5-2; Figure 5-2, 5-3).

Dissolved Oxygen (DO), one of the important physical water quality parameters showed on average a significant difference with DO concentration higher in the non-vegetated areas than the vegetated areas. Within the vegetated area the DO concentration was significant lower at sites 1 and 2 compared to sites 3 and 4. In case of the different forms of nutrients the levels were significantly higher at the sites close to the inflow area than the sites close to the outflow area (Figure 5-1). The mean total nitrogen (TN) concentration was significantly higher in sites at the inflow area than in sites at the outflow area ($P<0.001$). The TN concentration at site1 and 2 were higher than at the storage and inflow, but showed no significant difference ($P>0.05$), but the levels compared to site 3 and 4 were significantly higher ($P<0.001$, $P<0.01$ and $P<0.05$) (Table 5-2, 5-3; Figure 5-2).

The mean nitrate concentration showed a similar pattern like TN, whereas ammonium showed a different pattern. The difference of the mean ammonium concentration showed no significance between most the sites (P>0.05), except the levels from inflow and site1 showed statistically less significant to site 4 (P<0.05) (Table 5-2, 5-3; Figure 5-2).

Table 5-1: Physical water quality parameters from the different collection sites over the study period (2005-2007)

Parameters	Site	average	S.D	S.E	Minimum	Maximum
Water Temperature (WT) ºC	**storage**	16.80	4.39	0.49	10.43	28.28
	inflow	16.16	3.23	0.36	10.73	22.80
	site 1	16.25	3.12	0.34	10.35	23.30
	site 2	15.55	2.76	0.30	10.55	21.39
	site 3	14.73	3.41	0.38	9.04	21.14
	site 4	14.19	3.38	0.37	8.10	23.42
	outflow	14.22	3.50	0.39	8.34	23.63
Conductivity (EC) mS/cm	**storage**	0.211	0.131	0.014	0.076	0.730
	inflow	0.226	0.156	0.017	0.071	1.300
	site 1	0.223	0.102	0.011	0.090	0.501
	site 2	0.249	0.095	0.011	0.100	0.510
	site 3	0.228	0.078	0.009	0.110	0.402
	site 4	0.219	0.081	0.009	0.100	0.415
	outflow	0.208	0.080	0.009	0.092	0.420
Dissolved Oxygen (DO) mg/L	**storage**	6.05	3.61	0.40	0.45	14.58
	inflow	5.78	2.85	0.32	0.56	12.13
	site 1	3.46	2.48	0.27	0.27	10.89
	site 2	2.92	2.20	0.24	0.20	9.03
	site 3	4.37	3.06	0.34	0.33	16.48
	site 4	3.93	2.48	0.27	0.22	10.64
	outflow	5.03	2.75	0.30	0.15	12.30
pH	**storage**	7.30	0.59	0.07	5.69	8.83
	inflow	7.37	0.63	0.07	5.50	9.81
	site 1	7.08	0.37	0.04	5.98	8.24
	site 2	6.96	0.41	0.05	5.83	7.98
	site 3	6.79	0.41	0.05	5.34	7.70
	site 4	6.72	0.35	0.04	5.26	7.53
	outflow	6.74	0.46	0.05	5.01	7.62
Redox Potential (ORP) mV	**storage**	71.07	76.09	8.40	-90.56	241.37
	inflow	79.36	80.33	8.87	-91.50	227.27
	site 1	-16.62	101.77	11.24	-279.85	166.35
	site 2	-40.27	89.20	9.85	-270.95	161.27
	site 3	2.80	75.60	8.35	-243.50	216.22
	site 4	30.48	64.81	7.16	-136.73	154.18
	outflow	40.13	78.66	8.69	-198.72	211.35
Turbidity (Turb) NTU	**storage**	25.37	19.98	2.21	0.30	80.44
	inflow	23.04	18.69	2.06	1.81	95.02
	site 1	51.82	35.43	3.91	4.28	200.07
	site 2	41.95	58.03	6.41	1.58	501.80
	site 3	20.93	16.81	1.86	0.70	77.22
	site 4	16.07	17.57	1.94	0.60	93.18
	outflow	16.24	14.52	1.60	0.40	76.62

S.D. = standard deviation; S.E. = standard error; n=6

Table 5-2: Chemical and biological water quality parameters from the different collection sites over the study period (2005-2007)

Parameters	Site	average	S.D	S.E	Minimum	Maximum
Total Nitrogen (TN) mg/L	storage	0.97	0.37	0.04	0.27	2.10
	inflow	0.95	0.47	0.05	0.21	2.37
	site 1	1.03	0.42	0.05	0.31	2.13
	site 2	0.91	0.35	0.04	0.22	1.89
	site 3	0.73	0.24	0.03	0.23	1.45
	site 4	0.68	0.24	0.03	0.26	1.49
	outflow	0.73	0.29	0.03	0.24	1.53
Nitrate (NO_3^-) mg/L	storage	0.41	0.21	0.02	0.03	1.36
	inflow	0.38	0.23	0.03	0.01	1.20
	site 1	0.42	0.25	0.03	0.01	1.25
	site 2	0.32	0.21	0.02	0.02	0.91
	site 3	0.22	0.15	0.02	0.02	0.83
	site 4	0.18	0.11	0.01	0.02	0.51
	outflow	0.18	0.11	0.01	0.01	0.56
Ammonium (NH_4^+) mg/L	storage	0.38	0.27	0.03	0.03	1.20
	inflow	0.47	0.46	0.05	0.06	1.59
	site 1	0.46	0.31	0.03	0.06	1.83
	site 2	0.40	0.29	0.03	0.05	1.32
	site 3	0.34	0.24	0.03	0.07	1.18
	site 4	0.31	0.23	0.03	0.07	1.25
	outflow	0.38	0.29	0.03	0.04	1.26
Total Phosphorus (TP) mg/L	storage	0.047	0.038	0.004	0.008	0.220
	inflow	0.037	0.021	0.002	0.009	0.122
	site 1	0.059	0.055	0.006	0.014	0.275
	site 2	0.047	0.038	0.004	0.011	0.172
	site 3	0.036	0.037	0.004	0.007	0.231
	site 4	0.027	0.016	0.002	0.005	0.131
	outflow	0.025	0.013	0.001	0.007	0.086
Phosphate (PO_4^{3-}) mg/L	storage	0.033	0.035	0.004	0.004	0.202
	inflow	0.023	0.019	0.002	0.005	0.113
	site 1	0.042	0.048	0.005	0.003	0.250
	site 2	0.030	0.030	0.003	0.003	0.129
	site 3	0.021	0.031	0.003	0.002	0.213
	site 4	0.015	0.012	0.001	0.002	0.081
	outflow	0.014	0.010	0.001	0.002	0.060
Dissolved Organic Carbon (DOC) mg/L	storage	14.58	6.61	0.73	5.21	40.16
	inflow	13.67	5.85	0.65	3.41	34.38
	site 1	13.33	5.59	0.62	3.78	34.14
	site 2	13.60	6.04	0.67	4.14	44.08
	site 3	13.07	5.66	0.62	3.70	33.15
	site 4	12.22	4.41	0.49	4.41	27.60
	outflow	12.24	4.62	0.51	3.98	26.26
Chlorophyll a (chla) µg/L	storage	7.10	9.44	1.08	0.37	46.70
	inflow	7.25	11.49	1.25	0.19	55.74
	site 1	22.42	26.65	2.91	0.37	123.29
	site 2	20.91	19.10	2.08	1.25	96.00
	site 3	46.22	38.83	4.24	0.28	140.48
	site 4	16.38	15.86	1.73	0.19	65.38
	outflow	29.80	20.40	2.23	0.46	79.69

S.D. = standard deviation; S.E. = standard error; n=3

The mean of total phosphorus (TP) concentration had a similar pattern like the TN with one exception, that there was no significance between the inflow and outflow ($P>0.05$).

The concentration level was significantly different between site1 and 2 and outflow ($P<0.001$, $P<0.05$) (Table 5-2, 5-3; Figure 5-2). The mean concentration of phosphate showed the same pattern of significance like TP.

The mean concentration of chlorophyll-a (chl-a) showed no significant difference between all sites with the exception of site 3, which had a significantly higher chl-a concentration in comparison to the other sites ($P<0.001$, $P<0.01$). The mean concentration of dissolved organic carbon (DOC) showed no significance between the sites.

The patterns of significance for the physical parameters showed similar patterns to chemical and biological parameters (Table 5-1, 5-3; Figure 5-3). The mean of the water temperature was significantly higher at the sites in the inflow areas in comparison to the outflow areas. Conductivity showed no significant difference between all sites, however the conductivity level at site 2 was highest, but not significant ($P>0.05$). The mean concentration of dissolved oxygen (DO) showed significance between the non-vegetated and vegetated sites, with the non-vegetated sites having significantly higher DO concentration ($P<0.001$, $P<0.05$).

Within the vegetated sites there was also a significant difference between the sites close to the inflow and outflow, with site 1 and 2 having significantly lower levels of DO (Table 5-1, 5-3; Figure 5-3). The redox potential showed a similar pattern like that of DO, by having a significantly higher level in the non-vegetated sites than the vegetated sites ($P<0.001$). The pH showed a wide range significantly differences between, which is more easily summarized that there is significant difference in the pH between the inflow and outflow ($P<0.001$). Turbidity levels were significantly different between sites 1 and 2 against the other sites by having significantly higher turbidity levels ($P<0.001$) (Table 5-1, 5-3; Figure 5-3).

Overall it can be mentioned that there are significantly differences in the levels of the measured parameters between the different locations of sites, which can be grouped into pairs like the following: non-vegetated against vegetated, and inflow against outflow.

Table 5-3: ANOVA table for physical, chemical and biological water quality parameters for different collection sites. In case the ANOVA test confirmed a significance, post-test (Tukey's analysis) was performed to compare the individual datasets

Parameters	ANOVA	Site-Site	WT	EC	DO	pH	ORP	Turb	TN	NO_3^-	NH_4^+	TP	PO_4^{3-}	chl-a	DOC
WT	***	**sto-in**	N.S.		N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	
EC	N.S.	**sto-S1**	N.S.		***	N.S.	***	***	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	
DO	***	**sto-S2**	N.S.		***	***	***	**	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	
pH	***	**sto-S3**	**		**	***	***	N.S.	***	***	N.S.	N.S.	N.S.	***	
ORP	***	**sto-S4**	***		***	***	*	N.S.	***	***	N.S.	**	***	N.S.	
Turb	***	**sto-out**	***		N.S.	***	N.S.	N.S.	***	***	N.S.	***	***	N.S.	
TN	***	**in-S1**	N.S.		***	**	***	***	N.S.	N.S.	N.S.	**	***	N.S.	
NO_3^-	***	**in-S2**	N.S.		***	***	***	**	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	
NH_4^+	**	**in-S3**	N.S.		*	***	***	N.S.	**	***	N.S.	N.S.	N.S.	***	
TP	***	**in-S4**	**		***	***	**	N.S.	***	***	*	N.S.	N.S.	N.S.	
PO_4^{3-}	***	**in-out**	**		N.S.	***	*	N.S.	**	***	N.S.	N.S.	N.S.	N.S.	
chl-a	***	**S1-S2**	N.S.		N.S.	N.S.	N.S.	N.S.	N.S.	*	N.S.	N.S.	*	N.S.	
DOC	N.S.	**S1-S3**	N.S.		N.S.	**	N.S.	***	***	***	N.S.	***	***	***	
		S1-S4	**		N.S.	***	**	***	***	***	*	***	***	N.S.	
		S1-out	**		**	***	***	***	***	***	N.S.	***	***	N.S.	
		S2-S3	N.S.		*	N.S.	*	***	*	*	N.S.	N.S.	N.S.	***	
		S2-S4	N.S.		N.S.	*	***	***	**	***	N.S.	**	*	N.S.	
		S2-out	N.S.		***	*	***	***	*	***	N.S.	***	*	N.S.	
		S3-S4	N.S.		N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	**	
		S3-out	N.S.		N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	**	
		S4-out	N.S.		N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	N.S.	

*****, Extremely significant difference (P<0.001); **, moderately significant difference (P<0.01); *, significant difference (P<0.05); N.S., no significant (P>0.05); $\alpha = 0.05$, n=82**
sto = storage pond; in = inflow; S1 = site 1; S2 = site 2; S3 = site 3; S4 = site 4; out = outflow

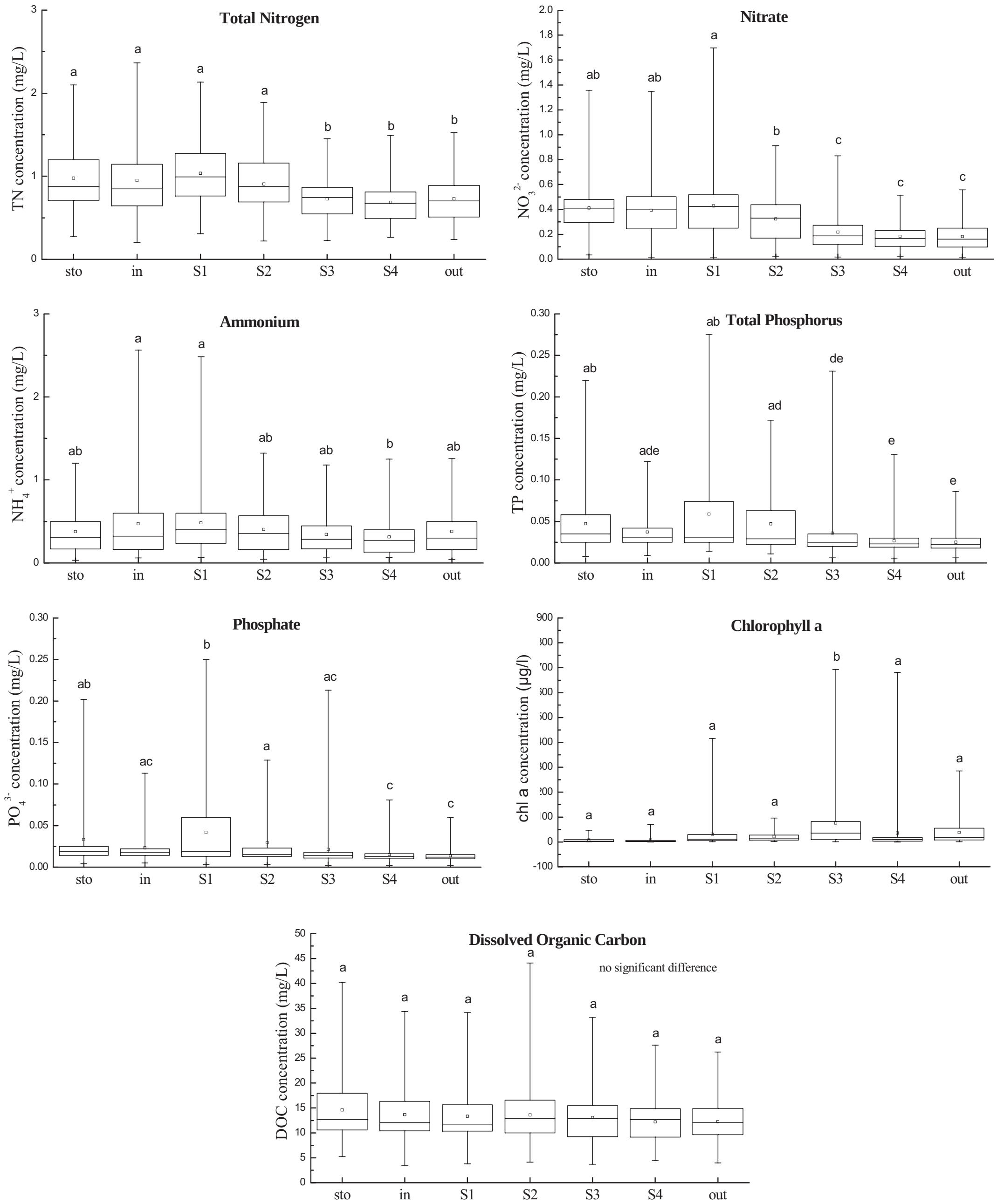


Figure 5-2: Bar and whisker graphs of chemical and biological water quality parameters for the different collection sites. The columns marked with different letters indicate significant difference according to Tukey's test (α = 0.05, n= 82). sto = storage, in = inflow, S1 = site 1, S2 = site 2, S3 = site 3, S4 = site 4, out = outflow

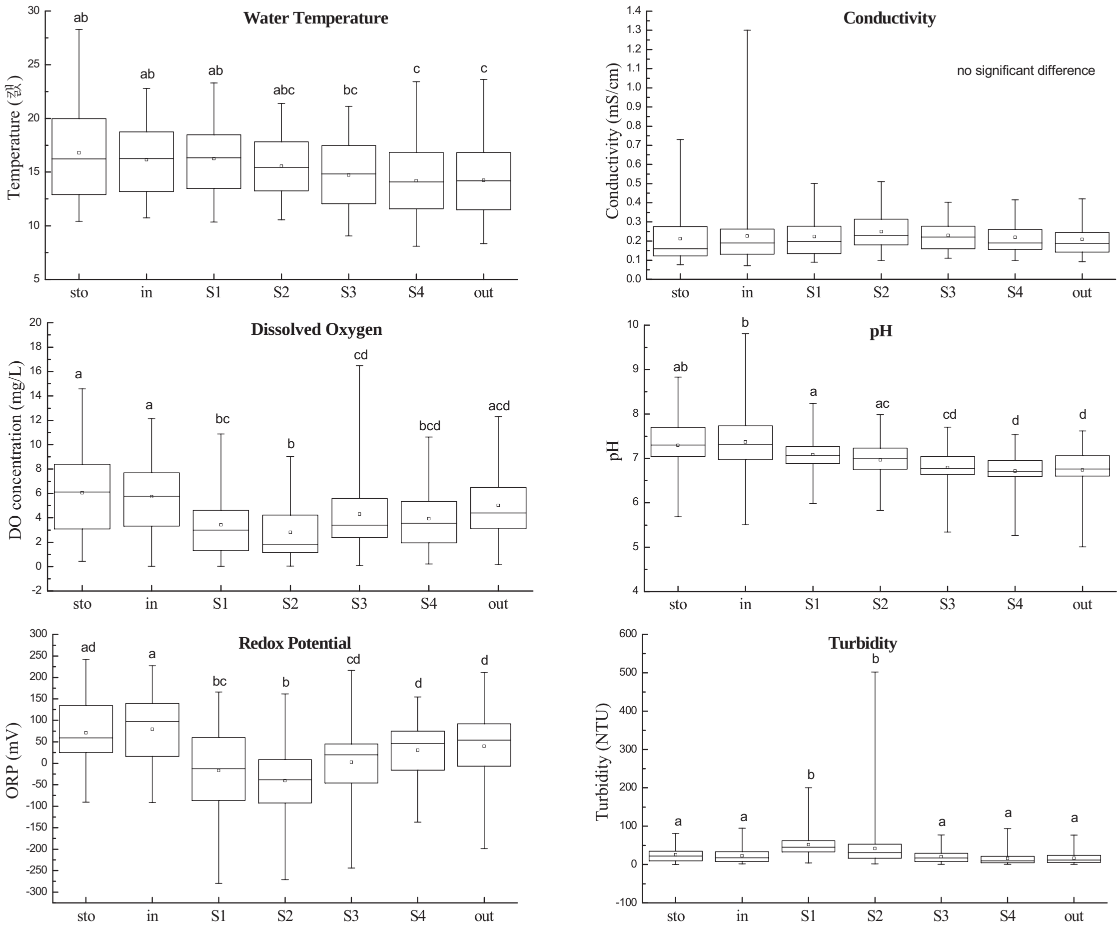


Figure 5-3: Bar and whisker graphs of physical water quality parameters for the different collection sites. The columns marked with different letters indicate significant difference according to Tukey's test (α = 0.05, n= 82). sto = storage, in = inflow, S1 = site 1, S2 = site 2, S3 = site 3, S4 = site 4, out = outflow

5.2.2 Seasonality

The seasonal patterns of the water quality parameters were distinguished between the different locations. The water temperature, which is mainly influenced by meteorological data (Figure 5-4, 5-4a, b, c, d, e, f and Table 5-4, 5-4a), showed a clear distinguished pattern during the season, which could be observed at all collection sites. The water temperature was the highest during the summer season and lowest during the winter season (Figure 5-4).

The seasonality of the levels of conductivity showed for all sites the same seasonal pattern, with low levels of conductivity in the winter season, which can be explained due to the higher rainfalls in the winter, which causes a dilution effect leading to lower levels in conductivity.

Higher levels of conductivity during the summer seasons could be explained due to the low or no rainfall in the summer. Another more reasonable explanation is that aquifer (ground) water has been fed into the Parafield system to keep the system permanently inundated. The aquifer water has higher levels of conductivity than the stormwater runoff from the catchment, because the aquifer water contains higher levels in conductivity due to the lower water table due to the close location to the ocean.

The dissolved oxygen and the correlating redox potential showed also a clear difference between the summer and winter season. The levels of the DO and ORP were relatively higher during the winter period than the summer period, which could be explained that during the summer period there is a higher DO consumption due microbial activity and also the higher water temperature and lower flow could influence the DO and ORP levels (Figure 5-4a & b).

The seasonal flow rates showed a significant difference between the summer and winter season ($P<0.05$, actual P value is 0.04867). Comparing all seasons together the results showed that there were no significant differences ($P>0.05$, actual P value is 0.052) between the seasons, however the magnitude of the flow rate during the spring and winter seasons were much higher than during the summer and autumn seasons (Figure B-11, Appendix B).

Table 5-4: ANOVA table for physical, chemical and biological water quality parameters for the different seasons. In case the ANOVA test confirmed a significance, post-test (Tukey's analysis) was performed to compare the individual datasets

		WT	EC	DO	pH	ORP	Turb	TN	NO_3^-	NH_4^+	TP	PO_4^{3-}	chl-a	DOC
sto	**ANOVA**	*	*	*	*	*	N.S.	*	N.S.	N.S.	*	N.S.	*	*
	Spring [1]	17.89^{a}	0.20^{ab}	5.30^{a}	7.68^{a}	56.31^{a}	24.00	0.97^{ab}	0.35	0.30	0.035^{a}	0.026	4.48^{ac}	15.23^{a}
	Summer [1]	23.56^{b}	0.29^{b}	5.30^{a}	7.42^{ab}	64.55^{a}	28.26	1.22^{b}	0.45	0.36	0.061^{b}	0.047	17.92^{b}	23.31^{b}
	Autumn [2]	17.37^{a}	0.21^{ab}	5.04^{a}	7.13^{b}	65.51^{a}	24.51	0.85^{a}	0.42	0.31	0.052^{ab}	0.037	9.04^{a}	14.74^{a}
	Winter [2]	12.37^{c}	0.17^{a}	7.71^{b}	7.20^{b}	114.98^{b}	18.61	0.90^{a}	0.38	0.41	0.039^{ab}	0.028	2.51^{c}	10.78^{c}
in	**ANOVA**	*	*	*	*	*	N.S.	*	N.S.	N.S.	*	N.S.	*	*
	Spring [1]	18.05^{a}	0.18^{a}	5.65^{a}	7.79^{a}	24.04^{a}	29.41	0.91^{ab}	0.33	0.29	0.027^{a}	0.016	2.94^{a}	14.30^{ab}
	Summer [1]	20.60^{b}	0.28^{b}	3.04^{b}	7.67^{a}	70.33^{ab}	23.72	1.11^{b}	0.41	0.42	0.036^{ab}	0.025	15.30^{b}	16.61^{a}
	Autumn [2]	16.98^{a}	0.23^{ab}	5.05^{a}	7.23^{b}	87.28^{b}	19.86	0.80^{a}	0.38	0.46	0.044^{b}	0.027	13.64^{b}	13.99^{ab}
	Winter [2]	12.65^{c}	0.20^{a}	7.49^{c}	7.10^{b}	117.61^{b}	23.19	1.02^{ab}	0.45	0.46	0.038^{ab}	0.027	2.58^{a}	12.75^{b}
S1	**ANOVA**	*	*	*	N.S.	*	*	*	N.S.	N.S.	*	*	N.S.	*
	Spring [1]	17.82^{a}	0.21^{ac}	2.39^{a}	7.10	-47.91^{a}	44.90^{ab}	1.05^{a}	0.32	0.43	0.029^{a}	0.019^{a}	27.04	13.12^{a}
	Summer [1]	20.64^{b}	0.31^{b}	1.67^{a}	7.05	-53.31^{a}	94.27^{a}	1.32^{b}	0.45	0.55	0.077^{b}	0.063^{b}	61.90	18.30^{b}
	Autumn [2]	16.98^{a}	0.25^{a}	2.68^{a}	7.13	-30.08^{a}	54.00^{ab}	0.87^{a}	0.41	0.47	0.071^{b}	0.051^{b}	38.52	13.73^{a}
	Winter [2]	13.10^{c}	0.17^{c}	5.68^{b}	6.94	59.32^{b}	41.16^{b}	1.04^{a}	0.49	0.41	0.041^{a}	0.025^{a}	17.69	11.55^{a}
S2	**ANOVA**	*	*	*	*	*	N.S.	*	*	N.S.	*	*	*	*
	Spring [1]	16.94^{a}	0.26^{a}	1.91^{a}	7.12^{a}	-56.33^{a}	54.04	0.92^{ab}	0.23^{a}	0.36	0.042^{a}	0.031^{a}	31.97^{a}	14.96^{a}
	Summer [1]	19.19^{b}	0.33^{b}	1.14^{a}	6.98^{ab}	-72.64^{a}	50.31	1.00^{b}	0.32^{ab}	0.38	0.067^{b}	0.054^{b}	25.68^{ab}	18.84^{b}
	Autumn [2]	16.19^{a}	0.27^{a}	2.26^{a}	6.98^{ab}	-52.61^{a}	29.47	0.79^{a}	0.29^{a}	0.35	0.048^{ab}	0.026^{a}	18.46^{b}	13.31^{ac}
	Winter [2]	12.80^{c}	0.19^{c}	4.66^{b}	6.79^{b}	20.80^{b}	39.47	0.95^{ab}	0.41^{b}	0.41	0.037^{a}	0.022^{a}	16.36^{b}	10.90^{c}

See Table 5-4a

Table 5-4 a: ANOVA table for physical, chemical and biological water quality parameters for the different seasons. In case the ANOVA test confirmed a significance, post-test (Tukey's analysis) was performed to compare the individual datasets

		WT	EC	DO	pH	ORP	Turb	TN	NO_3^-	NH_4^+	TP	PO_4^{3-}	chl-a	DOC
S3	**ANOVA**	*	*	*	*	*	N.S.	*	*	N.S.	*	*	*	*
	Spring [1]	16.55^{a}	0.22^{ac}	7.27^{a}	7.02^{a}	-3.50^{ab}	25.30	0.77^{ab}	0.22^{a}	0.26	0.051^{a}	0.041^{a}	191.60^{a}	15.81^{ab}
	Summer [1]	19.44^{b}	0.29^{b}	2.63^{b}	6.81^{ab}	-22.91^{a}	22.28	0.83^{b}	0.29^{a}	0.33	0.026^{bc}	0.014^{b}	47.25^{b}	19.01^{a}
	Autumn [2]	15.43^{a}	0.25^{ab}	3.12^{bc}	6.80^{ab}	8.51^{ab}	20.76	0.66^{a}	0.20^{a}	0.29	0.029^{bc}	0.014^{b}	29.42^{b}	12.90^{bc}
	Winter [2]	11.25^{c}	0.18^{c}	4.58^{c}	6.68^{b}	39.13^{b}	19.98	0.68^{ab}	0.20^{a}	0.37	0.031^{ac}	0.017^{b}	35.67^{b}	10.27^{c}
S4	**ANOVA**	*	*	*	*	*	N.S.	*	*	N.S.	N.S.	N.S.	*	*
	Spring [1]	15.50^{a}	0.19^{a}	3.37^{a}	6.86^{a}	47.76^{ac}	17.73	0.67^{a}	0.18^{ab}	0.25	0.026	0.015	83.62^{a}	13.42^{a}
	Summer [1]	19.27^{b}	0.29^{b}	1.77^{b}	6.72^{ab}	-14.06^{b}	26.15	0.84^{b}	0.24^{a}	0.34	0.024	0.012	20.08^{b}	17.15^{b}
	Autumn [2]	14.27^{a}	0.24^{c}	3.02^{ab}	6.70^{ab}	24.33^{a}	18.26	0.61^{a}	0.15b	0.25	0.025	0.014	12.30^{b}	12.01^{ac}
	Winter [2]	10.82^{c}	0.17^{a}	5.64^{c}	6.63^{b}	59.73^{c}	13.11	0.68^{a}	0.19^{ab}	0.35	0.026	0.014	10.82^{b}	10.82^{c}
out	**ANOVA**	*	*	*	*	*	*	*	*	N.S.	N.S.	*	*	*
	Spring [1]	15.69^{a}	0.19^{ac}	4.78^{a}	6.98^{a}	70.25^{a}	19.08^{ab}	0.80^{a}	0.18^{ab}	0.36	0.021	0.010^{a}	70.26^{a}	14.01^{ab}
	Summer [1]	19.40^{b}	0.28^{b}	3.04^{b}	6.79^{ab}	26.57^{ab}	26.67^{a}	0.86^{a}	0.25^{a}	0.37	0.023	0.010^{a}	17.45^{b}	16.52^{b}
	Autumn [2]	14.78^{a}	0.23^{a}	4.02^{ab}	6.62^{b}	24.72^{b}	18.80^{b}	0.60^{b}	0.17^{b}	0.32	0.025	0.013^{ab}	24.93^{b}	11.99^{ac}
	Winter [2]	10.82^{c}	0.16^{c}	6.36^{c}	6.67^{b}	71.85^{a}	13.03^{a}	0.72ab	0.16^{b}	0.38	0.026	0.016^{b}	28.65^{b}	10.52^{c}

***, significant difference ($P<0.05$); N.S., no significant ($P>0.05$); $\alpha = 0.05$, [1] n=26, [2] n=39 in the column followed by different letters indicate significant difference ($P<0.05$)**
sto = storage pond; in = inflow; S1 = site 1; S2 = site 2; S3 = site 3; S4 = site 4; out = outflow

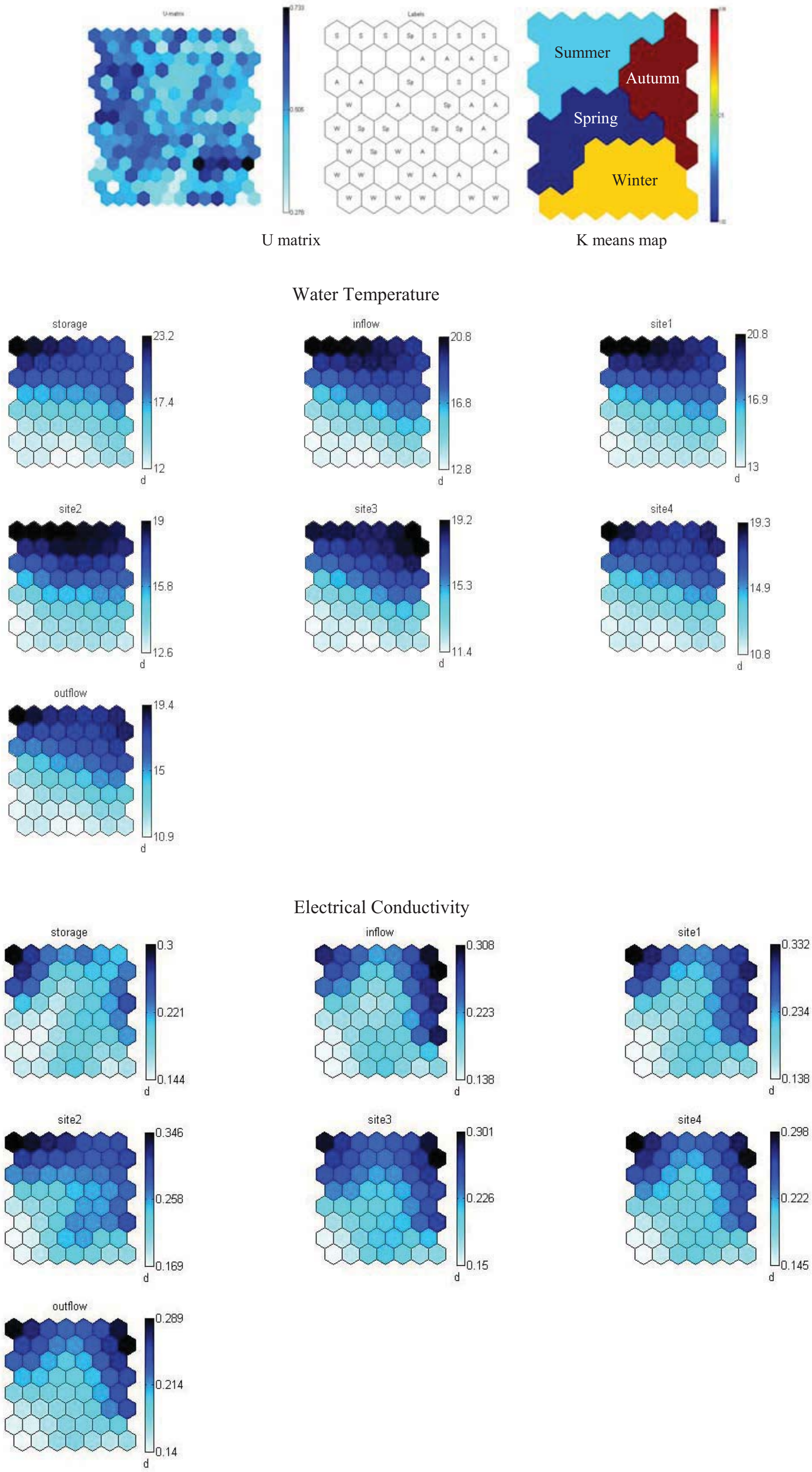


Figure 5-4: Seasonal patterns visualized as a U matrix, K means map and SOM maps for water temperature and electrical conductivity for the different sites

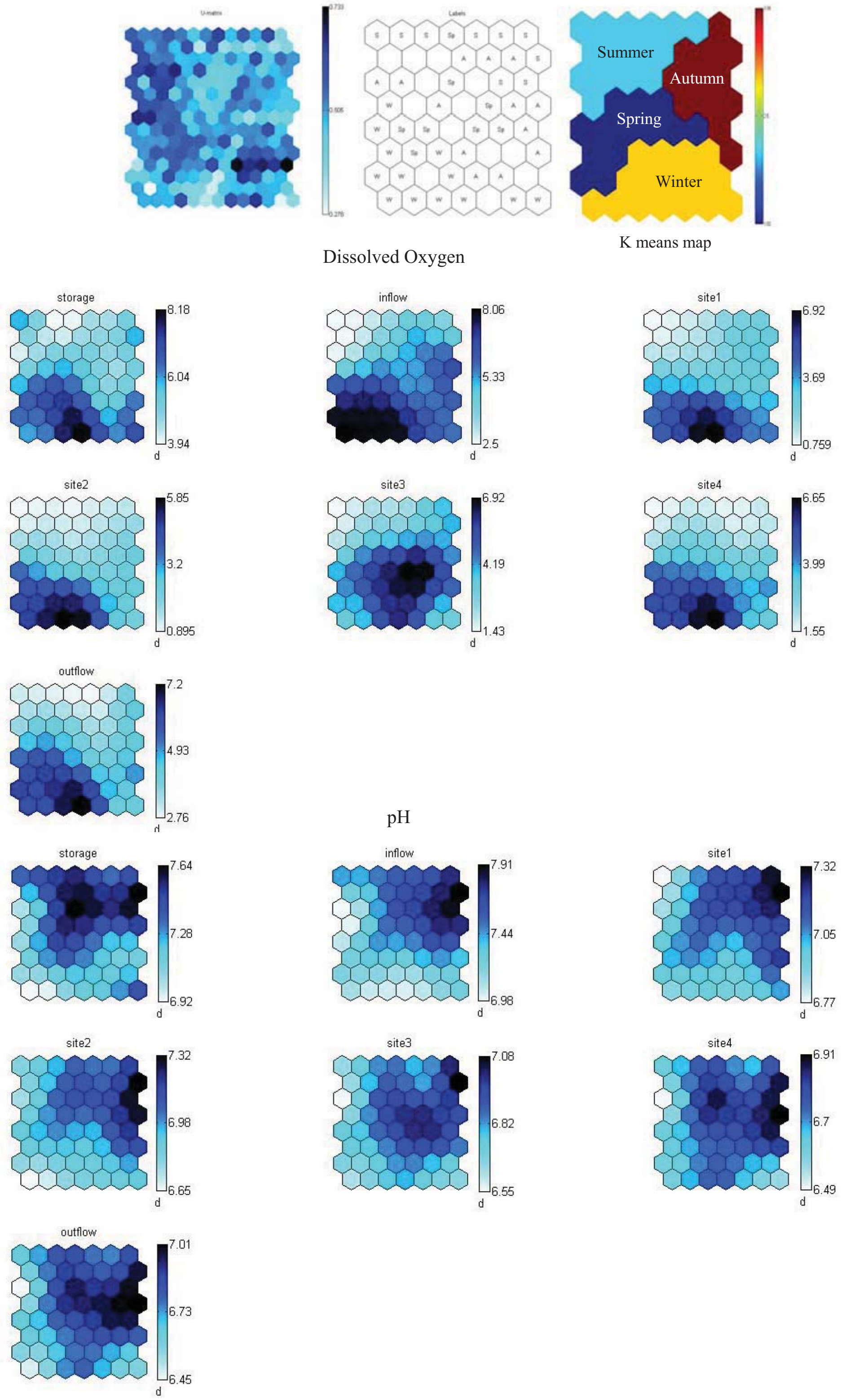


Figure 5-4 a: Seasonal patterns visualized as a U matrix, K means map and SOM maps for dissolved oxygen and pH for the different sites

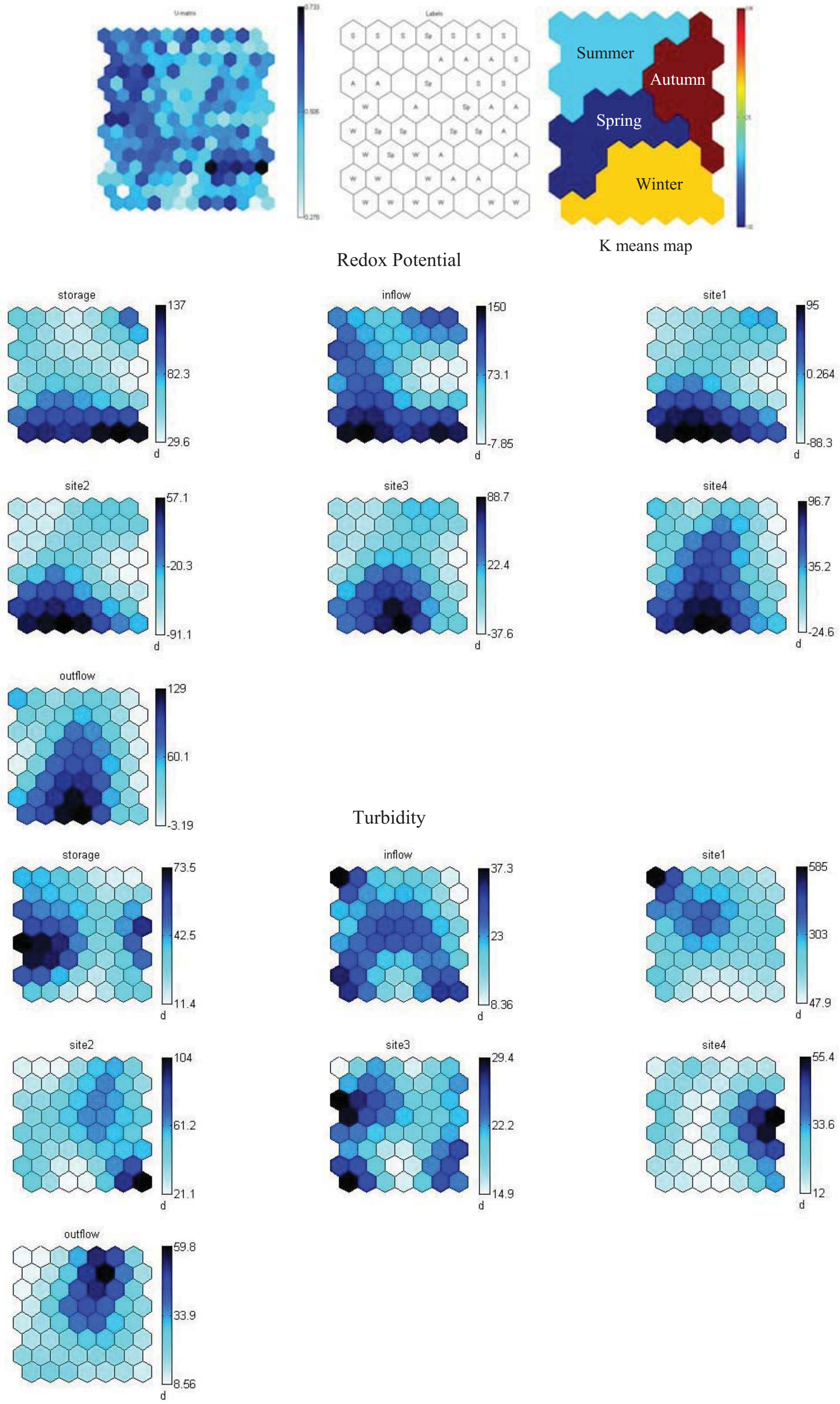


Figure 5-4 b: Seasonal patterns visualized as a U matrix, K means map and SOM maps for redox potential and turbidity for the different sites

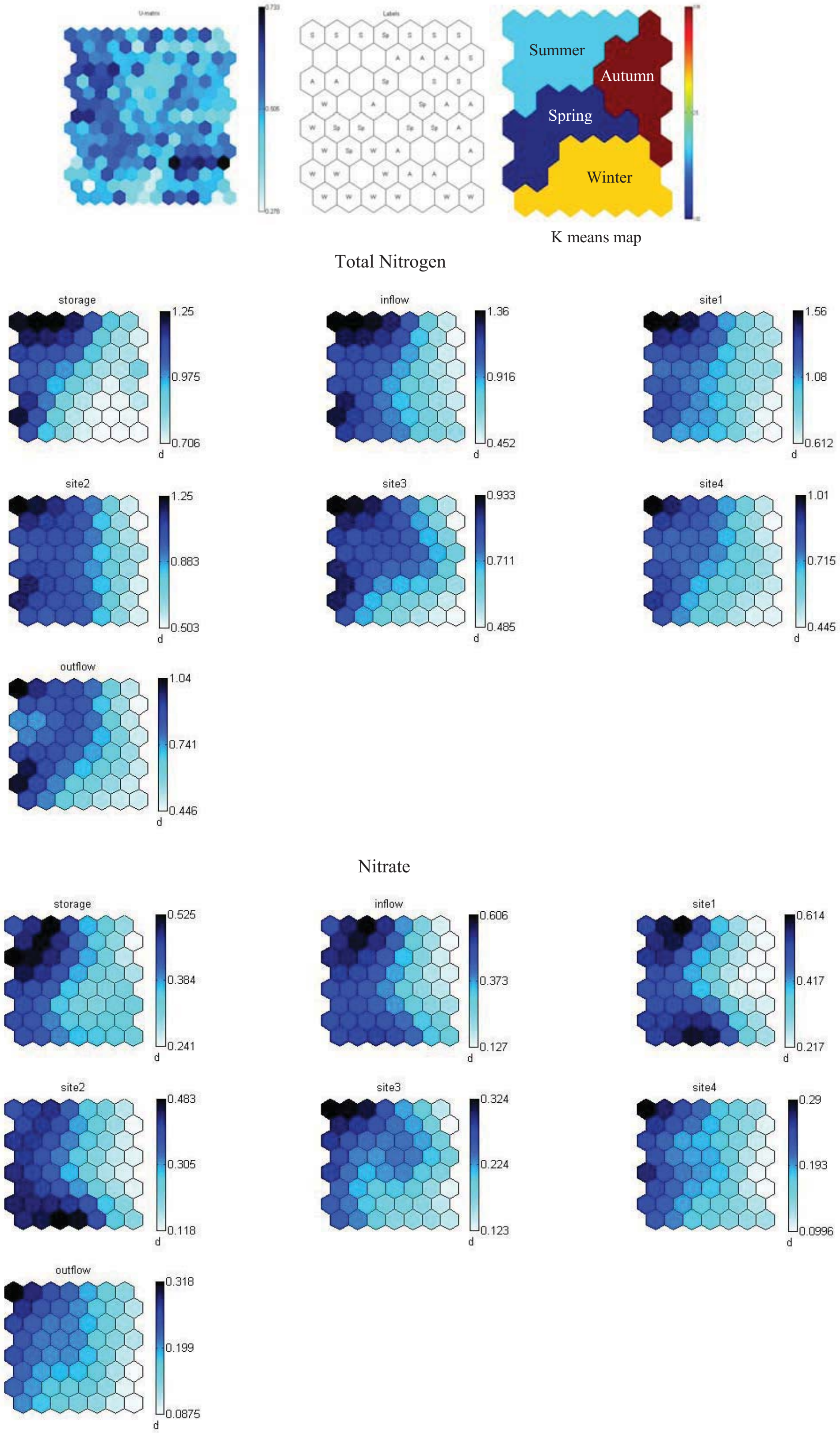


Figure 5-4 c: Seasonal patterns visualized as a U matrix, K means map and SOM maps for total nitrogen and nitrate for the different sites

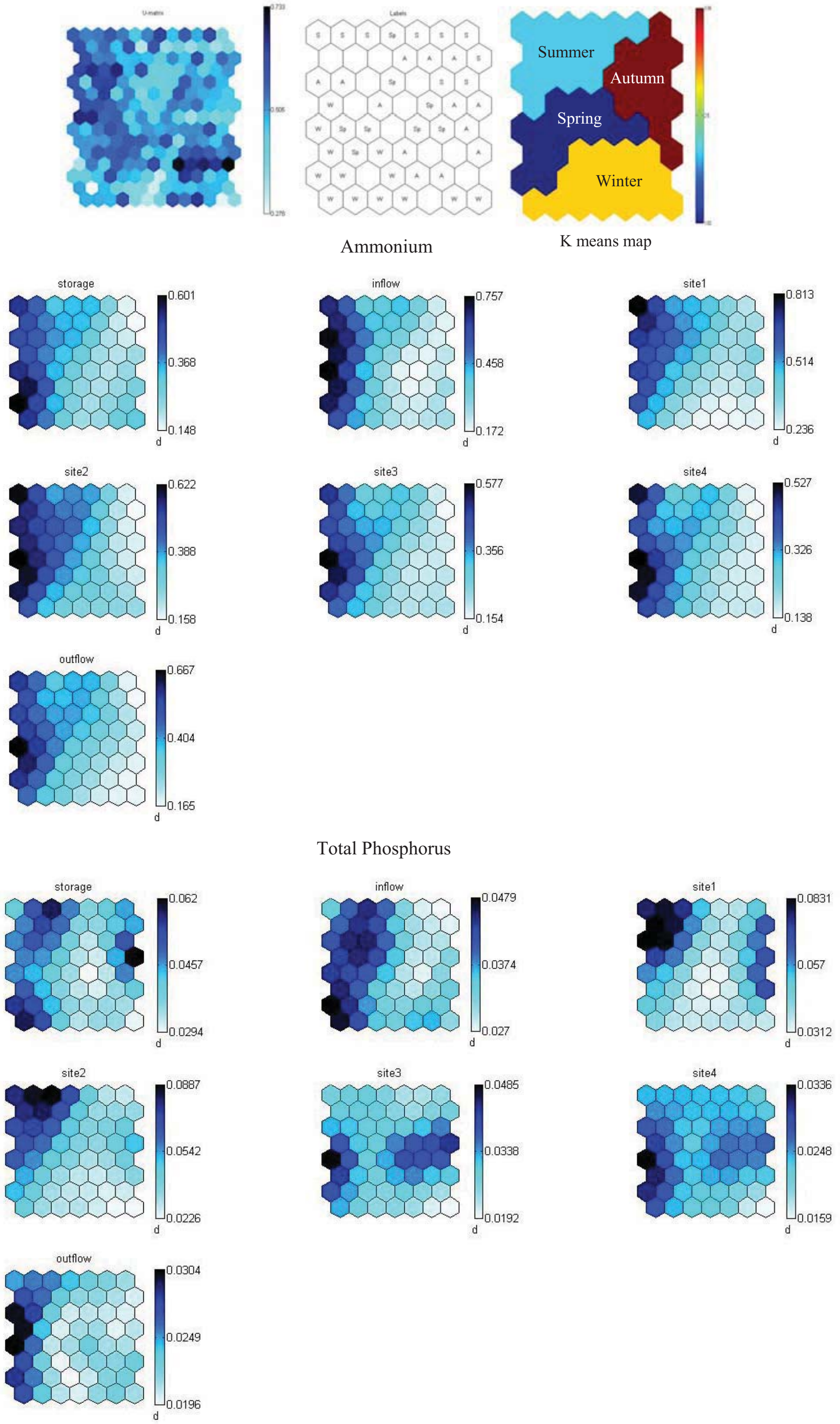


Figure 5-4 d: Seasonal patterns visualized as a U matrix, K means map and SOM maps for ammonium and total phosphorus for the different sites

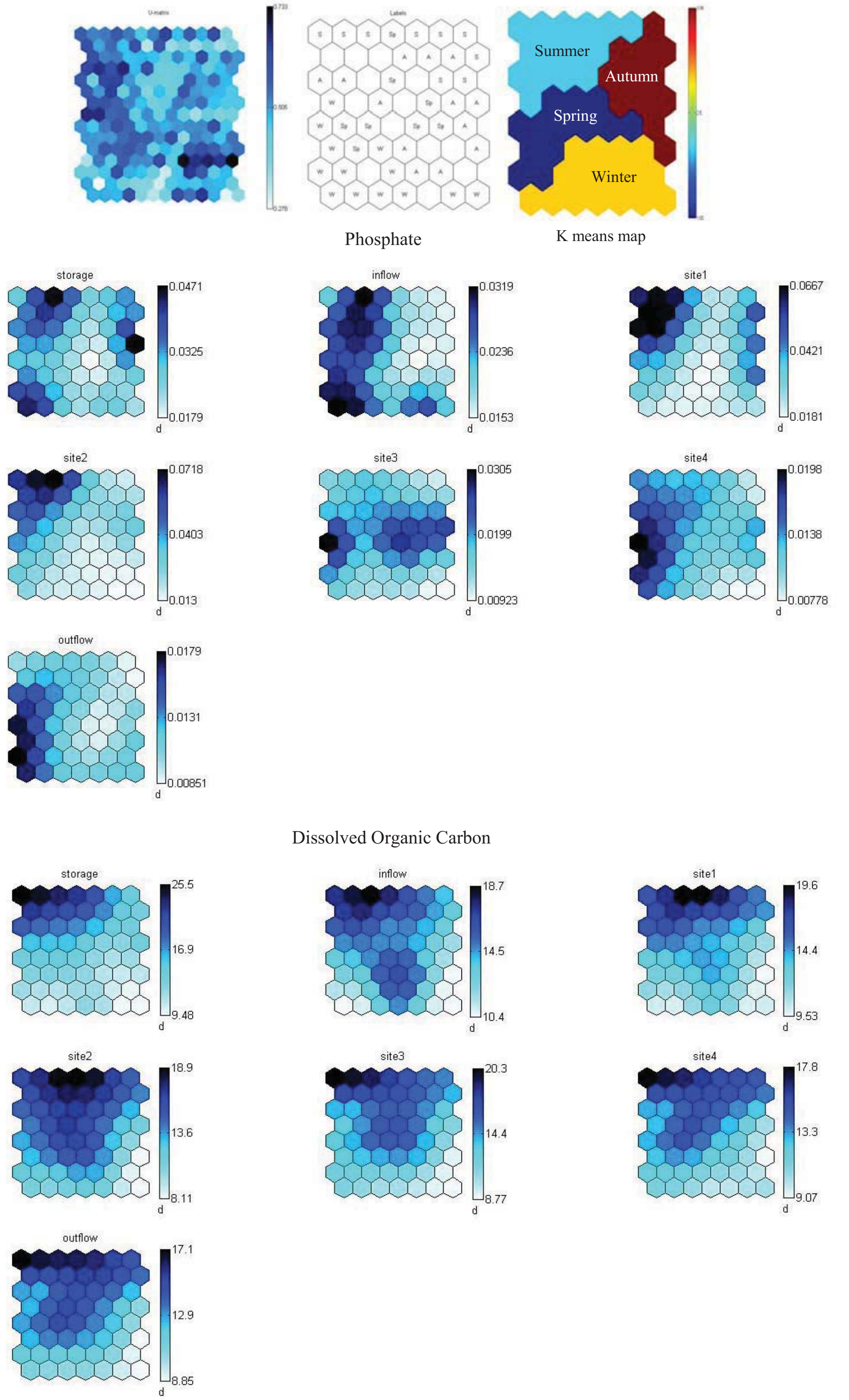


Figure 5-4 e: Seasonal patterns visualized as a U matrix, K means map and SOM maps for phosphate and dissolved organic carbon for the different sites

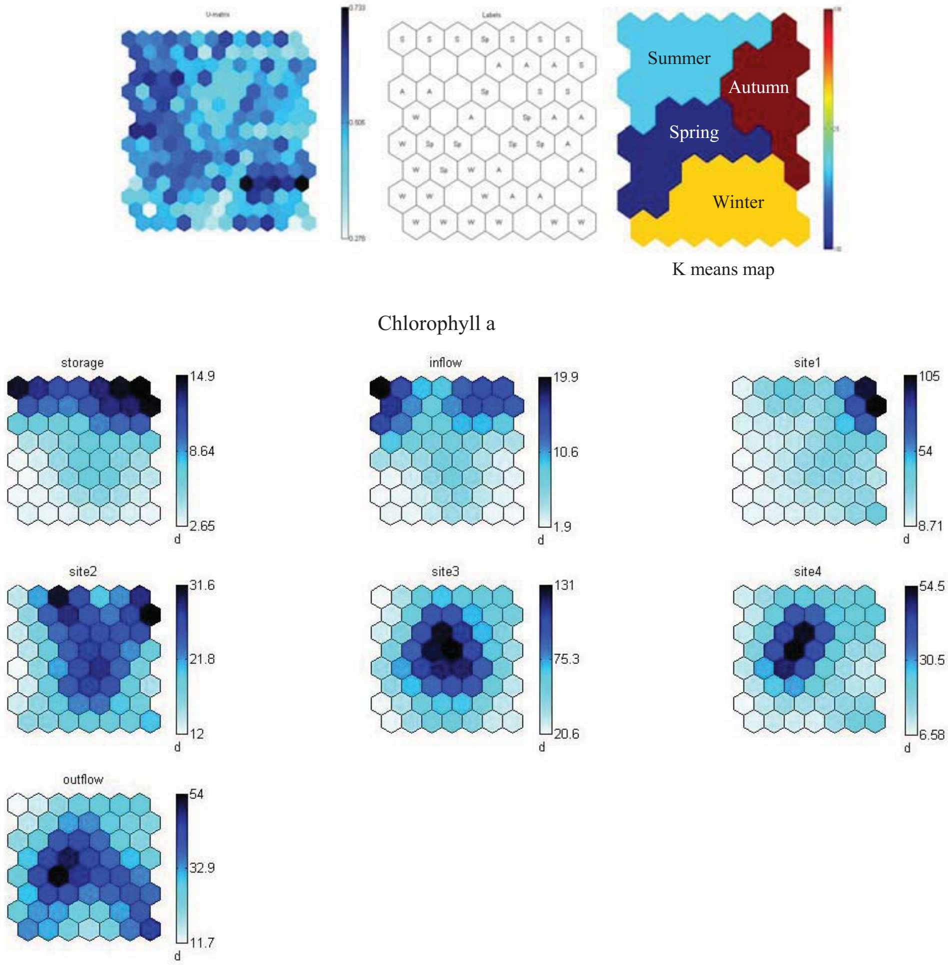


Figure 5-4 f: Seasonal patterns visualized as a U matrix, K means map and SOM maps for chlorophyll a for the different sites

The seasonal pattern of turbidity showed overall higher levels during the summer season than winter season (Figure 5-4b).

The seasonal pattern for the chemical parameters measured in form of nutrients showed a similar pattern than the physical parameters. The pattern of total nitrogen concentration showed in all sites the highest levels during the summer seasons with lower levels of concentration during the autumn and winter seasons. The fraction of TN (Figure 5-4c) measured in form of nitrate (Figure 5-4c) and ammonium (Figure 5-4d) showed a slightly distinguished seasonal pattern than TN. In case of nitrate the seasonal pattern showed two patterns, with nitrate concentrations in the sites closely located to the inflow showed the highest levels of concentration during winter, with the lowest concentrations during autumn. The pattern of sites located closely to the outflow showed a different pattern with the levels higher during the summer seasons, whereas the lowest concentration was observed during the autumn and winter seasons. The seasonal pattern of ammonium concentration showed overall higher levels in spring and summer, whereas the levels in autumn and winter were lower. The seasonal pattern of total phosphorus showed differences between the vegetated and non-vegetated sites. The levels of total phosphorus (Figure 5-4d) concentration were in the non-vegetated sites higher during the winter and spring seasons, whereas in the vegetated sites the higher levels were observed during the spring and summer seasons. Within the vegetated sites seasonal differences in the pattern concentration levels showed differences between sites located near inflow and outflow. The seasonal difference in the pattern of total phosphorus concentration within the vegetated sites showed that the sites near inflow had higher concentration during the summer seasons, whereas the sites near outflow had higher levels of total phosphorus in spring (Figure 5-4d).

The seasonal pattern of phosphate concentrations showed the same pattern like total phosphorus, with having a distinct seasonal pattern between non-vegetated and vegetated sites and within the vegetated sites. The levels of the phosphate concentration were in the non-vegetated sites higher during the winter and spring seasons. The seasonal pattern of the phosphate levels showed higher concentrations during the spring and summer seasons, with sites near the inflow having the highest concentration in summer, whereas sites near the outflow had higher concentration of phosphate in spring (Figure 5-4e).

The seasonal pattern of dissolved organic carbon showed in all sites similar levels of concentration during seasons, with having slightly higher concentrations observed during the summer season (Figure 5-4e).

Chlorophyll a was the only biological water quality parameter measured. It showed distinguished seasonal pattern between different sites. The storage pond showed high concentration of chlorophyll a during summer and autumn, with having the highest concentration in autumn. The inflow had higher concentration of chl-a in summer compared to the other seasons. Site 1 and site 2 had a similar seasonal chl-a pattern, in which the concentrations were the highest in autumn. Site 3, site 4 and outflow had the highest levels of concentration in the spring period (Figure 5-4f.

5.2.3 Annual pattern

The annual comparison of the physical, chemical and biological parameters (Figure 5-5 and B-8, Appendix B) revealed some changes between the years. The first year of the monitoring period was covering the period from March until November 2005. The second year covers all months from January until December 2006 and the third year was from January until August 2007. ANOVA was performed and in case it was successful a Tukey's test was performed. The results of the water quality parameters which didn't show any statistical significance are viewed in Figure B-8 ($P>0.05$), whereas figure 5-5 shows the results of the water quality parameters with statistical significance ($P<0.05$).

DO and ORP were the only physical parameters which showed a significant difference between the years. In case of DO the annual average, calculated based on the monthly interval showed no significant difference between year 2005 and 2006, but the DO level of 2007 was significantly lower than the previous two years (Figure 5-5 A). ORP showed that the first year (Year 2005) had a significant lower level than 2006 and 2007. The redox potential between 2006 and 2007 showed no significant difference (Figure 5-5 B). The annual comparison of the chemical parameters showed for total nitrogen no significant difference in the annual average between first two years, but 2007 showed a significant lower level of TN concentration (Figure 5-5 C). A similar pattern can be observed for nitrate (Figure 5-5 D), whereas ammonium showed a significant difference between all years, which displays a declining trend (Figure 5-5 E).

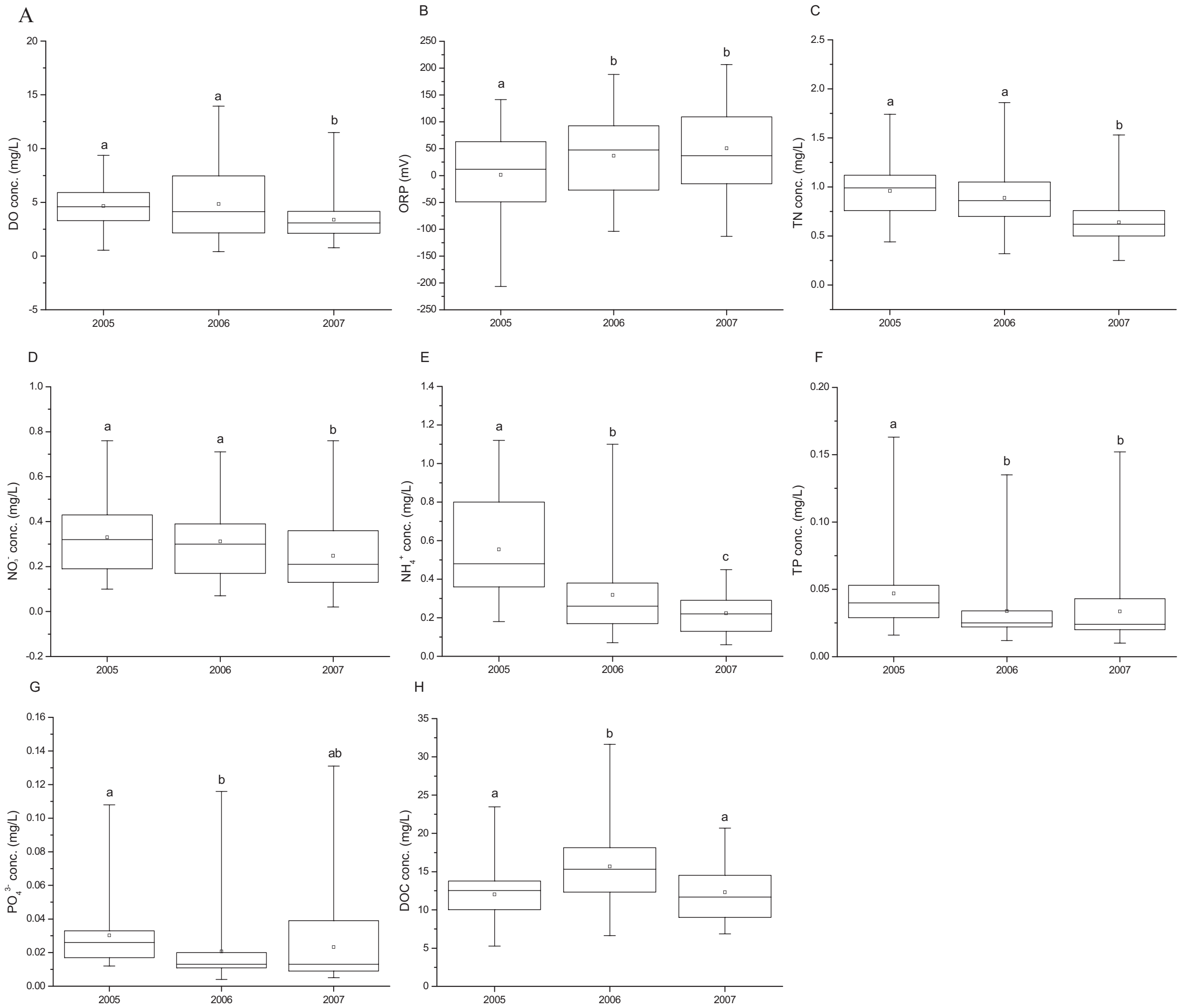


Figure 5-5: Annual comparison of water quality parameters. A. Dissolved Oxygen, B. Redox Potential, C. Total Nitrogen, D. Nitrate, E. Ammonium, F. Total Phosphorus, G. Phosphate and H. Dissolved Organic Carbon. The columns marked with different letters indicate significant difference according to Tukey's test (α = 0.05; 2005 n= 63, 2006 n = 84 and 2007 n = 56)

The annual comparison of TP and phosphate showed that there are significant differences between the years. In case of TP the average level of concentration was significantly lower in the second and third year than the first year (Figure 5-5 F). For phosphate the levels of concentration showed significant differences between the year 2005 and 2006, with the levels in 2006 significantly lower, but both years had no significant difference to year 2007 (Figure 5-5 G). For DOC the annual pattern showed a significant difference in the level of concentration, which was higher in 2006 than in comparison to 2005 and 2007. The annual average concentration of DOC levels didn't show any statistical significance (Figure 5-5 H).

5.2.4 Performance of reed bed system (Nutrient removal efficiency and removal rate)

5.2.4.1 General description

The performance of reed bed pond, which were assessed by the nutrient removal efficiency (%) and removal rate (mg/m^2/day), were summarized in Tables 5-5 for N components, 5-8 for P components and 5-11 for DOC. The nutrient removal efficiency and rate were calculated based on the total nutrient loadings and discharges on a monthly basis using the following equations:

$$RE = \frac{\Sigma (C_i Q_i - C_o Q_o)}{\Sigma C_i Q_i} \times 100 \qquad (5\text{-}1)$$

$$RR = \frac{\Sigma (C_i Q_i - C_o Q_o)}{A \times D} \qquad (5\text{-}2)$$

Where Ci and Co are the nutrient concentrations (mg/L) at inflow and outflow;

Qi and Qo are the flow rates at inflow and outflow (l)

Qo is calculated with following equation: Qo = Qi –ET (5-3)

ET = evaporation (l), A = surface area of reed bed (m^2), D = number of days (day)

5.2.4.2 Removal performance of nitrogen components by the reed bed

The performance of the reed bed related to the removal efficiency and rate of N components showed that there was a difference of removal performance within the reed bed system, which is summarized in Table 5-5. Overall the reed bed showed a removal for all N components. The performance within the reed bed showed that not all parts of the reed bed are removing nutrients. Regarding TN removal performance within the system it showed that the most efficient part was site 3. The characteristic of site 3 is that it consists of *Phragmites australis* dominantly and *Typha orientalis*, but the vegetation in form of the above-ground biomass was removed during the harvesting process, which didn’t recover until the end of study period.

The site 2 ranked second in the removal efficiency, which consists of a macrophytes composition with a dominating *Phragmites australis* and *Schoenoplectus validus* and *Typha orientalis*. The least efficient performance was shown by site 1. The plant composition consists of a very dense *Phragmites australis*, *Schoenoplectus validus*, *Eleocharis sphacelata*, and *Typha orientalis* community, which has the highest primary productivity in the system.

Table 5-5: Average N removal efficiency (RE, %) and removal rate (RR, $mg/m^2/day$) during the study period

	TN		NO_3^-		NH_4^+	
	RE	**RR**	**RE**	**RR**	**RE**	**RR**
site 1	-10.73 (5)	-0.39	-4.22 (5)	-0.07	-7.41 (4)	-0.12
site 2	13.01 (2)	0.52	22.48 (2)	0.36	17.33 (1)	0.31
site 3	20.31 (1)	0.71	31.18 (1)	0.39	14.73 (2)	0.21
site 4	5.19 (3)	0.14	19.52 (3)	0.17	7.16 (3)	0.09
outflow	-5.04 (4)	-0.13	1.33 (4)	0.01	-18.01 (5)	-0.21
Overall (inflow – outflow)	23.55	0.85	53.24	0.79	17.04	0.28

Number in the brackets are the ranking based on the removal efficiency

The order for the removal efficiency of TN was site 3 > site 2 > site 4 > outflow > site 1. For the rest of N components, NO_3^-, the order of the removal efficiency was: site 3 > site 2 > site 4 > outflow > site 1. For NH_4^+ the order was site 2 > site 3 > site 4 > site 1 > outflow. Based on the ranking of the removal efficiencies it showed that site 2 and site 3 are the sites with the highest removal efficiencies, whereas site 1 is the least efficient in the removal of all forms of nitrogen (Table 5-5). The overall removal efficiency for TN was 23.55 %, for nitrate 53.24 % and ammonium 17.04 %.

Table 5-6: Results of Tukey's multiple comparisons for the performance for: A. TN removal efficiency; B. TN removal rate

A	**site 1**	**site 2**	**site 3**	**site 4**	**outflow**
site 1					
site 2	***				
site 3	***	N.S.			
site 4	**	N.S.	N.S.		
outflow	N.S.	N.S.	*	N.S.	

B	**site 1**	**site 2**	**site 3**	**site 4**	**outflow**
site 1					
site 2	***				
site 3	***	N.S.			
site 4	*	N.S.	**		
outflow	N.S.	***	***	N.S.	

*****, Extremely significant difference ($P<0.001$); **, moderately significant difference ($P<0.01$); *, significant difference ($P<0.05$); N.S., no significant ($P>0.05$); n = 29**

The ANOVA test results showed that the removal performance had significant differences for both TN removal efficiency and removal rate (Table 5-6 and Figure 5-6). The performance was calculated based on monthly intervals. The results of post ANOVA test – Tukey's test showed that site1 area had significantly lower removal efficiencies (Figure 5-6 A) and rates (Figure 5-6 B) than the other sites with the exception of outflow. Site 2, site3 and site 4 had no significant difference in their TN removal efficiencies ($P<0.05$).

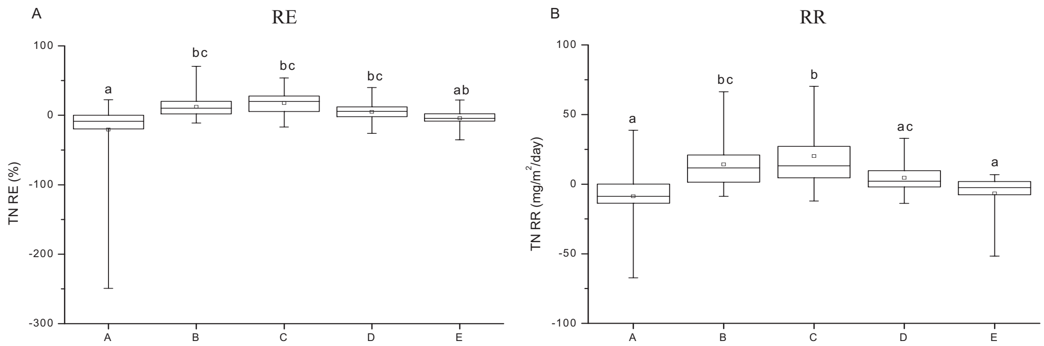


Figure 5-6: A. TN removal efficiency (RE); B. TN removal rate (RR). The columns marked with different letters indicate significant difference according to Tukey's test (α = 0.05, n= 29). A. site1, B. site2, C. site3, D. site4 and E. outflow.

The removal rate showed that there was a significant difference between site 4, and site2 and site 3, with having a slightly lower removal rate, which on the other site was significantly higher than the rates of site 1 and outflow (Figure 5-6 B).

The overall removal performance of the reed bed in regards of the different N forms were that the removal efficiencies for TN, NO_3^- and NH_4^+ were 23.55%, 53.24% and 17.04% respectively. The overall removal rates from the system during the whole study period (2005-2007) were for TN (15.7 g/day), for NO_3^- (14.62 g/day) and for NH_4^+ (5.14 g/day).

The total mass of nutrients removed by the reed bed system was for TN (13.83 kg), for NO_3^- (12.91 kg) and for NH_4^+ (4.54 kg).

5.2.4.2.1 Seasonal pattern of N removal

The seasonal comparison N removal efficiency, which are shown in Figure 5-7, showed seasonal differences, with the removal efficiencies higher during the non-growing seasons (autumn and winter) than during the growing seasons (spring and summer).

The performance of reed bed pond displayed extreme changes during the study period (Table 5-7). In site 1, for example, the removal efficiency for TN ranged from -249.25 to 22.58 %. For the other N components the highest variation occurred in site 3 for nitrate (-258.81 to 78.22 %) and site 1 for ammonium (-298.54 to 50.20 %). But in general we can see that all the other sites as well have great variations in the removal performances as well the removal rates, which are correlating with the removal efficiency.

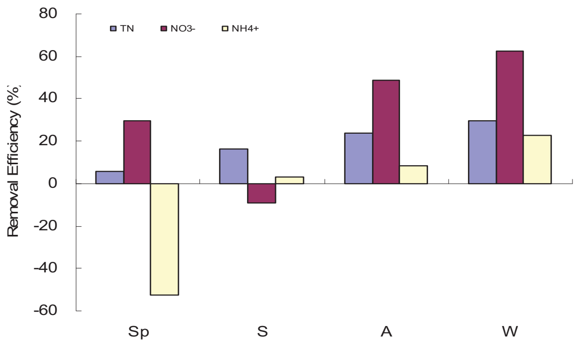


Figure 5-7: Seasonal comparison of N removal efficiency (Sp = Spring, S = Summer, A = Autumn and W = Winter)

Table 5-7: Minimum and maximum N removal efficiency (RE, %) and removal rate (RR, mg/m^2/day) for the different sites (n=29)

		TN		NO_3^-		NH_4^+	
		RE	**RR**	**RE**	**RR**	**RE**	**RR**
Site 1	**Min**	-249.25	-67.27	-66.93	-19.84	-298.54	-25.33
	Max	22.58	38.64	45.92	16.99	50.20	58.73
Site 2	**Min**	-11.11	-8.81	-11.32	-3.08	-43.97	-20.64
	Max	70.51	66.47	87.76	42.51	72.23	30.86
Site 3	**Min**	-17.07	-12.08	-258.81	-21.86	-72.46	-7.29
	Max	53.75	70.35	78.22	46.05	59.09	27.93
Site 4	**Min**	-26.26	-13.88	-48.08	-11.82	-31.94	-9.79
	Max	39.71	32.90	72.29	21.91	49.27	20.55
outflow	**Min**	-35.5	-51.69	-95.31	-8.92	-89.7	-38.54
	Max	21.74	6.78	50.00	15.08	16.54	9.24

The seasonal pattern for the N removal performances didn't show a clear trend, however it showed a distinguished trend between the sites (Figure 5-8). In case of TN removal efficiency, site 1 was for majority of the study period performing inefficient in regards the removal of any TN with a few exceptions, like the performance in July 2006 (22.58%), which was the highest removal percentage during the whole study period.

Most the removal events occurred during the autumn and winter seasons, but for most of the time site 1 was inefficient.

Outflow showed a similar pattern like site 1, with higher removal performances during the non-growing seasons. Other sites, including site 2, site 3 and site 4, showed comparably higher efficient TN removal performances, which were mostly during the spring, summer and partially autumn seasons, with being least efficient during the winter season.

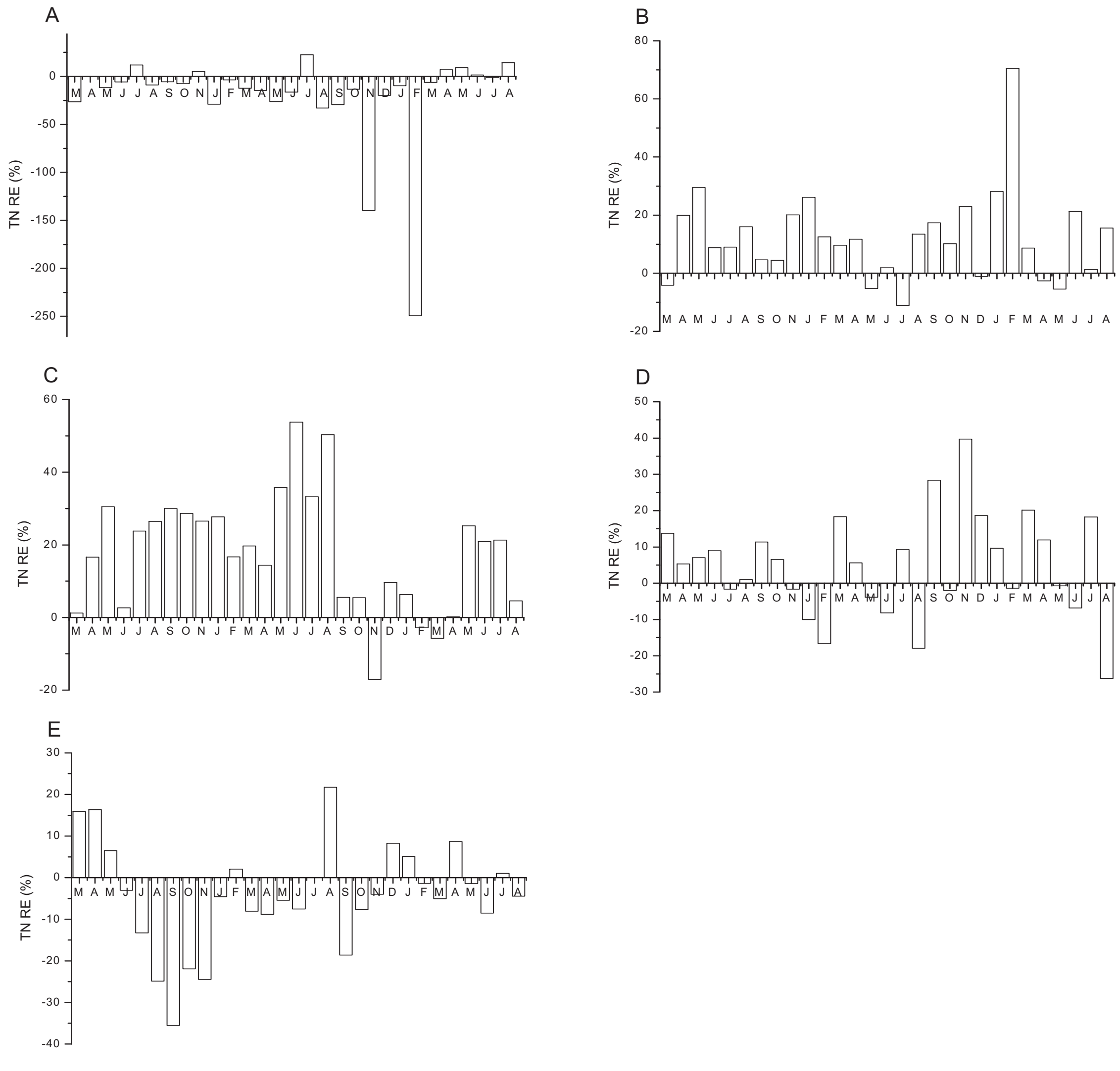


Figure 5-8: Total Nitrogen (TN) Removal Efficiency (%) for the different sites (A. site 1, B. site 2, C. site 3, D. site 4, E. outflow). Negative columns represent TN release. n = 29

The seasonal removal performance for nitrate and ammonium showed a similar pattern than TN (Figure C-1, C-2). Site 1 and outflow showed a positive removal performance occurring during the winter seasons, with being ineffective for most of the other seasons.

The rest of the sites showed higher removal performances during the growing seasons with being less effective in the non-growing seasons (Table C-1, Figure C-1 and C-2; Appendix C).

Overall it was difficult to determine a clear seasonal pattern, which could be linked to all the sites, but a reason could be that the nutrient dynamics are distinguished between the different sites, which have different characteristics.

5.2.4.3 Removal performance of phosphorus components by the reed bed

The P removal performance revealed that the reed bed system overall was successful in the removal of the different forms of P, which are summarized in Table 5-8. The overall removal performance regards TP was 34.19 % and for PO_4^{3-} it was 46.79 %. Within the reed bed the removal efficiency for TP was highest at site 4, which was also the case for PO_4^{3-}. Like at the removal performance of the N components, site 1 was the least efficient by showing negative efficiencies which means that it was more a source than a sink (Table 5-8). For both P forms the performance within the reed bed was more or less identical. The order of the average removal efficiency was for both P the same: site 4> site 3> site 2> outflow> site 1.

Table 5-8: Average P removal efficiency (RE, %) and removal rate (RR, mg/m^2/day) during the study period

	TP		PO_4^{3-}	
	RE	**RR**	**RE**	**RR**
site 1	-53.93 (5)	-0.08	-69.85 (5)	-0.06
site 2	19.23 (3)	0.04	26.89 (3)	0.04
site 3	24.04 (2)	0.04	29.12 (2)	0.03
site 4	25.64 (1)	0.03	34.12 (1)	0.03
outflow	6.29 (4)	0.01	8.25 (4)	0.00
Overall (inflow – outflow)	34.19	0.05	46.79	0.04

Number in the brackets are the ranking based on the removal efficiency

The ANOVA and Tukey's test showed the removal performance regards P had significant differences between the sites (Table 5-9, Figure 5-9). The test results revealed that for both P forms, site 1 had a significantly lower removal efficiency (Figure 5-9 A) and rate (Figure 5-9 B) than the other sites ($P<0.05$). Regards the removal efficiency, all sites with exception of site 1 showed no significant difference between each other (P>0.05). The removal rate showed that site 1 and outflow had a significantly lower rate than the other sites.

The overall removal performance of the reed bed in regards of the different P forms was that the removal efficiencies for TP and PO_4^{3-} were 34.19% and 46.79%. The removal rates from the system over the whole study period were for TP (0.88 g/day) and PO_4^{3-} (0.78 g/day). The total mass of P forms removed by the reed bed system was for TP (0.78 kg) and for PO_4^{3-} (0.69 kg).

Table 5-9: Results of Tukey's multiple comparisons for the performance for: A. TP removal efficiency; B. TP removal rate

A	**site 1**	**site 2**	**site 3**	**site 4**	**outflow**
site 1					
site 2	*				
site 3	**	N.S.			
site 4	**	N.S.	N.S.		
outflow	*	N.S.	N.S.	N.S.	
B	**site 1**	**site 2**	**site 3**	**site 4**	**outflow**
site 1					
site 2	*				
site 3	**	N.S.			
site 4	**	N.S.	N.S.		
outflow	N.S.	N.S.	N.S.	N.S.	

*****, Extremely significant difference (P<0.001); **, moderately significant difference (P<0.01); *, significant difference (P<0.05); N.S., no significant (P>0.05); n = 29**

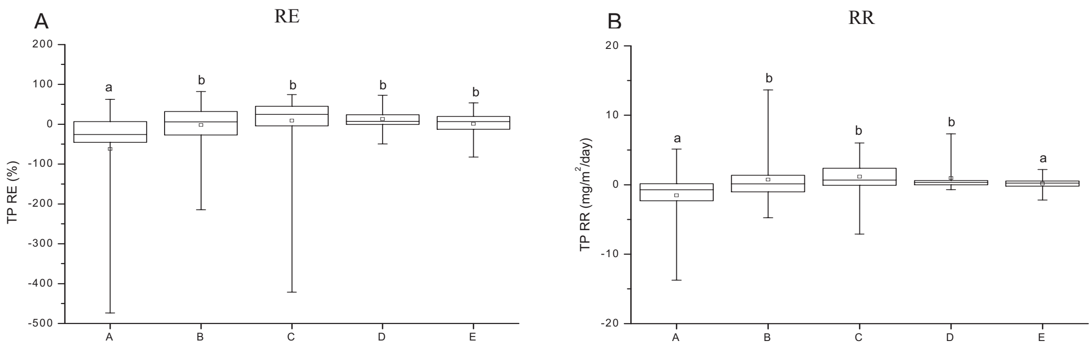


Figure 5-9: A. TP removal efficiency (RE); B. TP removal rate (RR). The columns marked with different letters indicate significant difference according to Tukey's test (α = 0.05, n= 29). A. site1, B. site2, C. site3, D. site4 and E. outflow.

5.2.4.3.1 Seasonal pattern of P removal

The seasonal performance in regards the P components for the different seasons showed that the performance seemed to be consistent (Figure 5-10). But removal performance was most efficient during the summer and autumn seasons than during the spring and winter seasons.

The performance of P removal efficiency and rate showed large variations within the reed bed during the study period (Table 5-10). For example of TP site 1 showed a removal efficiency range from -473.58 to 62.39 %. In phosphate site 1 showed an efficiency range from -956.46 to 69.58 %.

The performance of P removal efficiency and rate showed large variations within the reed bed during the study period (Table 5-10). For example of TP site 1 showed a removal efficiency range from -473.58 to 62.39 %. In phosphate site 1 showed an efficiency range from -956.46 to 69.58 %.

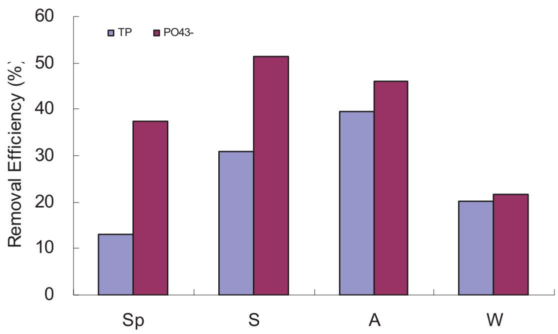


Figure 5-10: A. Seasonal comparison of P removal efficiency (Sp = Spring, S = Summer, A = Autumn and W = Winter

Table 5-10: Minimum and maximum P removal efficiency (RE, %) and removal rate (RR, mg/m^2/day) for the different sites (n=29)

		TP		PO_4^{3-}	
		RE	**RR**	**RE**	**RR**
Site 1	**Min**	-473.58	-13.76	-959.46	-12.97
	Max	62.39	5.14	69.58	5.74
Site 2	**Min**	-214.55	-4.75	-275.70	-3.96
	Max	81.91	13.65	90.05	12.90
Site 3	**Min**	-421.38	-7.10	-808.80	-6.75
	Max	74.33	6.01	85.48	5.19
Site 4	**Min**	-49.73	-0.71	-62.50	-0.65
	Max	72.66	7.32	89.97	6.83
outflow	**Min**	-82.50	-2.22	-88.89	-0.79
	Max	53.51	2.18	63.27	3.07

The different sites displayed distinguishable seasonal pattern between the different sites. Similar to the N components it was difficult to find a seasonal pattern for site 1 (Figure 5-11). For the majority of the study period the removal performance showed negative efficiency. Positive efficiency was observed during the autumn season with exception in the first year.

All other sites displayed positive removal performances during the spring and summer seasons, with a lower efficiency for autumn and winter. Most of all in comparison to site 1, the rest of the sites showed reasonable amounts of P removal (Figure 5-11). For phosphate the removal performances of all sites in regards of seasonal behaviour were similar to TP (Table C-3, Figure C-5 and C-6; Appendix C).

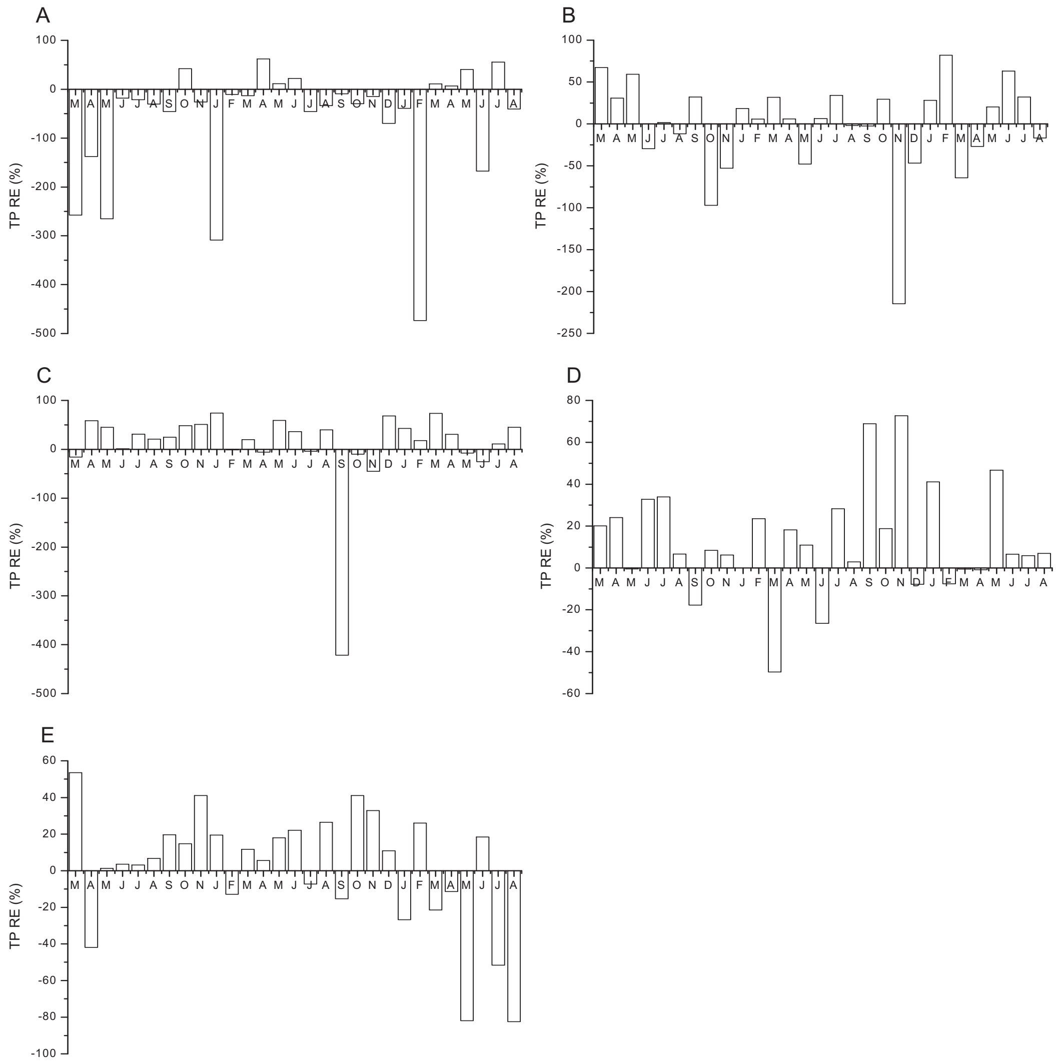


Figure 5-11: Total Phosphorus (TP) Removal Efficiency (%) for the different sites (A. site 1, B. site 2, C. site 3, D. site 4, E. outflow). Negative columns represent TP release. n = 29

5.2.4.4 Removal performance of dissolved organic carbon by the reed bed

The performance of the reed bed in relation to DOC removal efficiency and rate was minimal, but overall the system was removing DOC (Table 5-11). The overall DOC removal efficiency was 10.51 % and the removal rate was 5.75 mg/m^2/day. The performance within the reed bed pond showed that site 4 had the highest removal efficiency with 6.60 %. Site 3 was the second highest in regards removal efficiency. The total order of DOC removal efficiency was the following: site 4 > site 3 > site 1> outflow > site 2.

ANOVA and Tukey's test were performed to determine if there was a significant difference in the removal performance between the sites, but the results were that there was no statistical significance in the DOC removal performance between the sites. Removal efficiency (Figure 5-12 A) showed no significant difference between the sites (P>0.05). The removal rate showed the same result (P>0.05), but it seem to be that site 2 had a slightly lower removal rate, which wasn't statistical significant (Figure 5-12 B).

Table 5-11: Average DOC removal efficiency (RE, %) and removal rate (RR, mg/m^2/day) during the study period

	DOC	
	RE	**RR**
site 1	2.83 (3)	1.54
site 2	-1.99 (5)	-1.06
site 3	3.05 (2)	1.65
site 4	6.60 (1)	3.47
outflow	0.29 (4)	0.14
Overall (inflow – outflow)	10.51	5.75

Number in the brackets are the ranking based on the removal efficiency

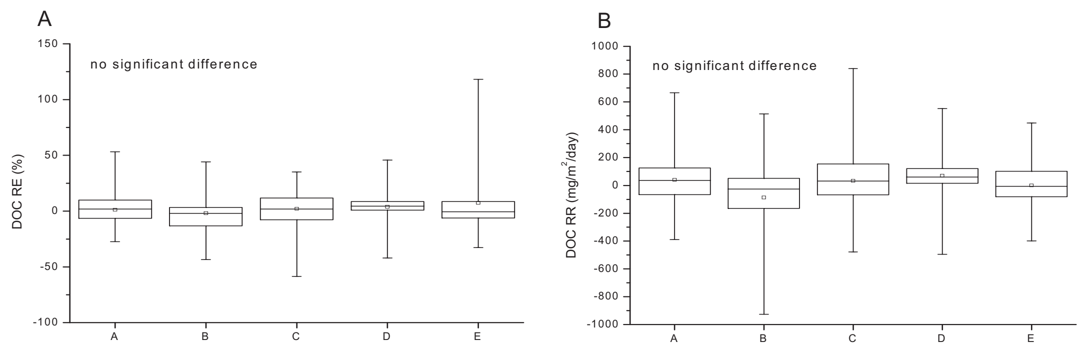


Figure 5-12: A. TP removal efficiency (RE); B. TP removal rate (RR). The columns marked with different letters indicate significant difference according to Tukey's test (α = 0.05, n= 29). A. site1, B. site2, C. site3, D. site4 and E. outflow.

The overall removal performance of DOC by the reed bed system was that the removal efficiency was 10.51%. The overall removal rate was 105.76 g/day and the total mass of DOC removed was 93.39 kg over the whole study period.

5.2.4.4.1 Seasonal pattern of DOC removal

The seasonal removal performance of DOC for the different seasons (Figure 5-13) showed that during non-growing seasons the removal performance was most efficient, whereas the removal performances during the growing seasons were minimal or close to non-existent.

The performance of DOC removal efficiency and rate showed large variations during the whole study period in study location of the reed bed (Table 5-12). For example the DOC removal efficiency at the outflow showed the highest variation between the minimum and maximum, which were ranging from -32.74 to 118.11 %. The variation of the removal rate was much more extreme. Site 2 had the highest variation in the removal rate ranging from -927.0 to 513.36 $mg/m^2/day$. In general the variation in removal efficiency and rate were high (Table 5-12).

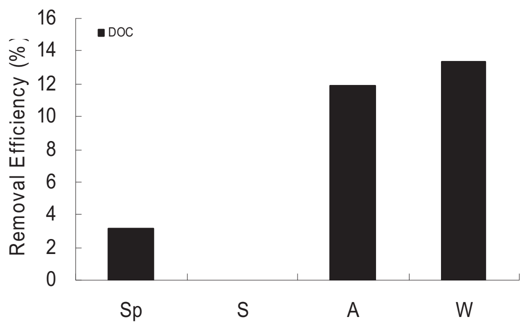


Figure 5-13: Seasonal comparison of DOC removal efficiency (Sp = Spring, S = Summer, A = Autumn and W = Winter)

Table 5-12: Minimum and maximum DOC removal efficiency (RE, %) and removal rate (RR, $mg/m^2/day$) for the different sites (n=29)

		DOC	
		RE	**RR**
Site 1	**Min**	-27.46	-389.43
	Max	53.15	666.72
Site 2	**Min**	-43.44	-927.00
	Max	44.19	513.36
Site 3	**Min**	-58.63	-478.24
	Max	35.22	839.25
Site 4	**Min**	-42.07	-496.25
	Max	45.68	551.40
outflow	**Min**	-32.74	-399.06
	Max	118.11	448.48

Similar to the removal performance of the other nutrients forms, it was difficult to find a clear seasonal pattern fitting all sites (Figure 5-14). The DOC removal efficiency at site

1 showed no clear seasonal trend (Figure 5-14 A), but most of the positive removal performances were observed during the winter and spring seasons. Sporadic positive events were also observed in autumn as well. The seasonal pattern for site 2 revealed that positive performances occurred during the autumn and winter seasons, whereas the performance during the growing seasons were mostly negative (Figure 5-14 B). Site 4 showed the most consistent performance with positive removal efficiencies occurring during the winter, spring and summer seasons (Figure 5-14 D). Site 3 and outflow showed positive removal performances scattered over different seasons, which made it difficult to determine a clear seasonal pattern (Figure 5-14 C, E).

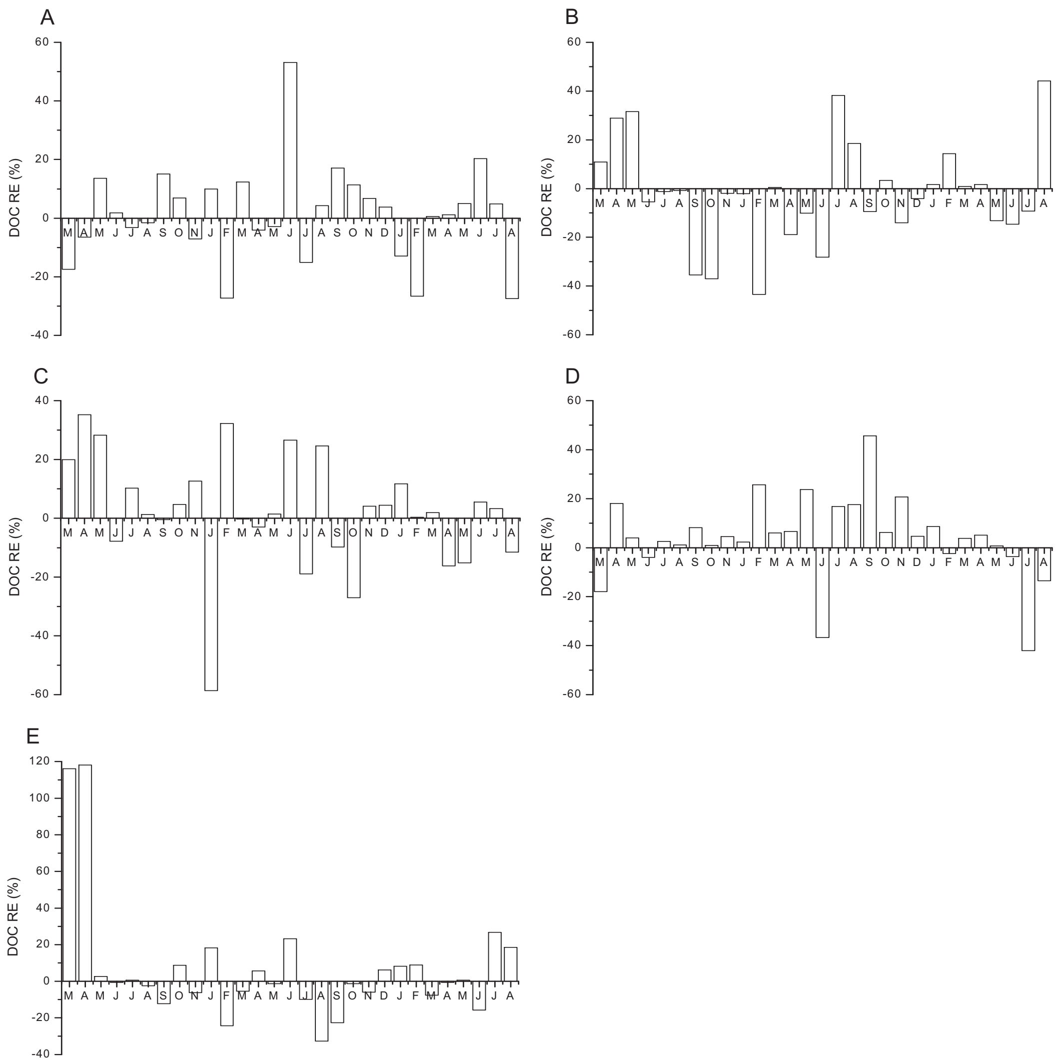


Figure 5-14: Dissolved Organic Carbon (DOC) Removal Efficiency (%) for the different sites (A. site 1, B. site 2, C. site 3, D. site 4, E. outflow). Negative columns represent DOC release. n = 29

5.2.4.5 Relationship between removal performance and residence time

The overall average residence time of the reed bed pond was around 12 days (11.81 days). The monthly residence time was calculated on a monthly basis using the following equation:

$$RT = V \div Q \tag{5-4}$$

V = wetland water volume (m^3) and Q = volumetric flow rate (m^3)

The monthly and seasonal residence time showed variation in the magnitude (Figure 5-15) or at least a difference in residence time between the growing seasons and non-growing seasons, but ANOVA and Tukey's test revealed that no significant statistical differences exist between the residence time and seasons (Figure 5-15 B, $P > 0.05$).

The residence time of the reed bed system was categorized in four groups, which are short residence time (SRT) (0-5 days), medium residence time (MRT) (5-10 days), long residence time (LRT) (10-15 days) and extreme residence time (ERT) (>15 days).

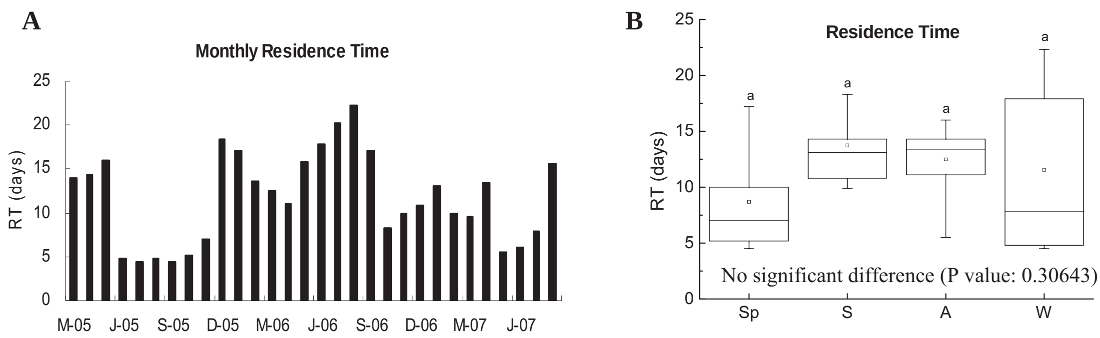


Figure 5-15: A. Monthly Residence Time; B. Bar and whisker graph comparing the residence time during the different seasons. (Sp = Spring, S = Summer, A = Autumn, W = Winter) The columns marked with the same letter indicate no significant differences according to Tukey's test ($\alpha = 0.05$, n= 11)

The influence of residence time (RT) on the removal efficiencies (RE) for the different nutrients showed, that there were spatial differences between the collection sites, which is visualized in Figure 5-16. The removal efficiency of TN showed that the overall performance was most efficient at a ERT. Comparisons between the collection sites showed that the RE was spatially distinguished. The removal performance at site 1 for the different residence time categories showed for all categories a negative performance; where the performance at a SRT was the most efficient, but still negative (Figure 5-16 A). Site 2 and 3 showed for all flow categories positive removal performances, with performances to be most effective at site 2 at MRT and at site 3 at ERT. Site 4 showed to be performing effective at MRT and LRT. At the outflow site the removal performances were all negative with the exception of performance at a LRT.

The overall removal performance for NO_3^- showed to be most effective at ERT, but it also showed high performances at SRT and LRT (Figure 5-16 B). Between the collection sites there were distinguishable spatial variations, which showed a similar trend compared to TN. Site 1 showed negative performances with the exception at MRT. Site 2 showed for all flow categories positive performances, with removal efficiencies to be highest at MRT. Performance at site 3 was highest at SRT, but was showing negative performance at MRT. Site 4 showed high removal performances at MRT and LRT, whereas the performance at SRT and ERT was negative. Outflow site showed positive performance at MRT and LRT, whereas the removal performance was negative at SRT and ERT.

For ammonium the overall performance showed to be most efficient at SRT followed by ERT, whereas the performance at MRT and LRT was inefficient (Figure 5-16 C). Spatially the variations were similar to the previous N components. Site 1 and outflow showed for most of the residence time categories a negative performance. Removal performances for sites 2, 3 and 4 showed to be most efficient at LRT, ERT and LRT.

The removal efficiency for both P components in form of total phosphorus and phosphate showed to be most effective at MRT (Figure 5-16 D & E). At site 1 the removal performances for TP and PO_4^{3-} were for all flow categories negative. For the sites 2, 3 and 4 the performances were most efficient at LRT, LRT and MRT for TP and ERT, LRT and MRT for PO_4^{3-}. The removal performance at the outflow site was most effective under SRT conditions for both P components.

The removal efficiency of DOC showed that the overall performance was most efficient at ERT (Figure 5-16 F). The removal efficiencies showed that different flow regimes influenced the removal performances at the different collection sites. The performances at sites 1, 2 and 4 were most efficient at ERT conditions, whereas the other sites showed to be most effective under LRT conditions.

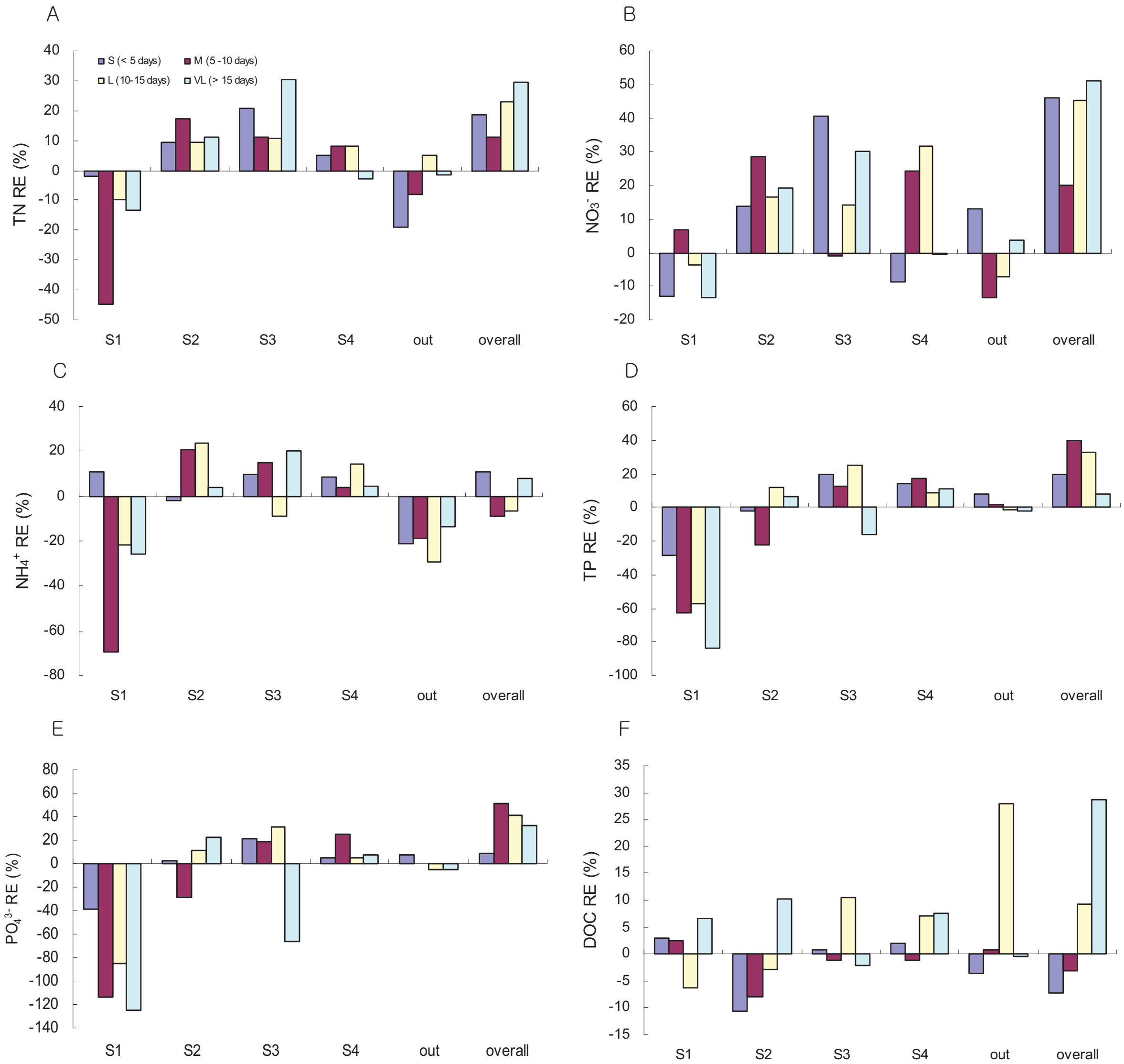


Figure 5-16: Nutrient removal efficiencies for the different collection sites at different residence time categories. A. Total Nitrogen, B. Nitrate, C. Ammonium, D. Total Phosphorus, E. Phosphate and F. Dissolved Organic Carbon

5.2.4.6 Relationship between removal performance and nutrient loadings

Figures 5-17, 5-18 and 5-19 are showing the relationships between the removal performances of the different nutrient forms and the inflow concentrations.

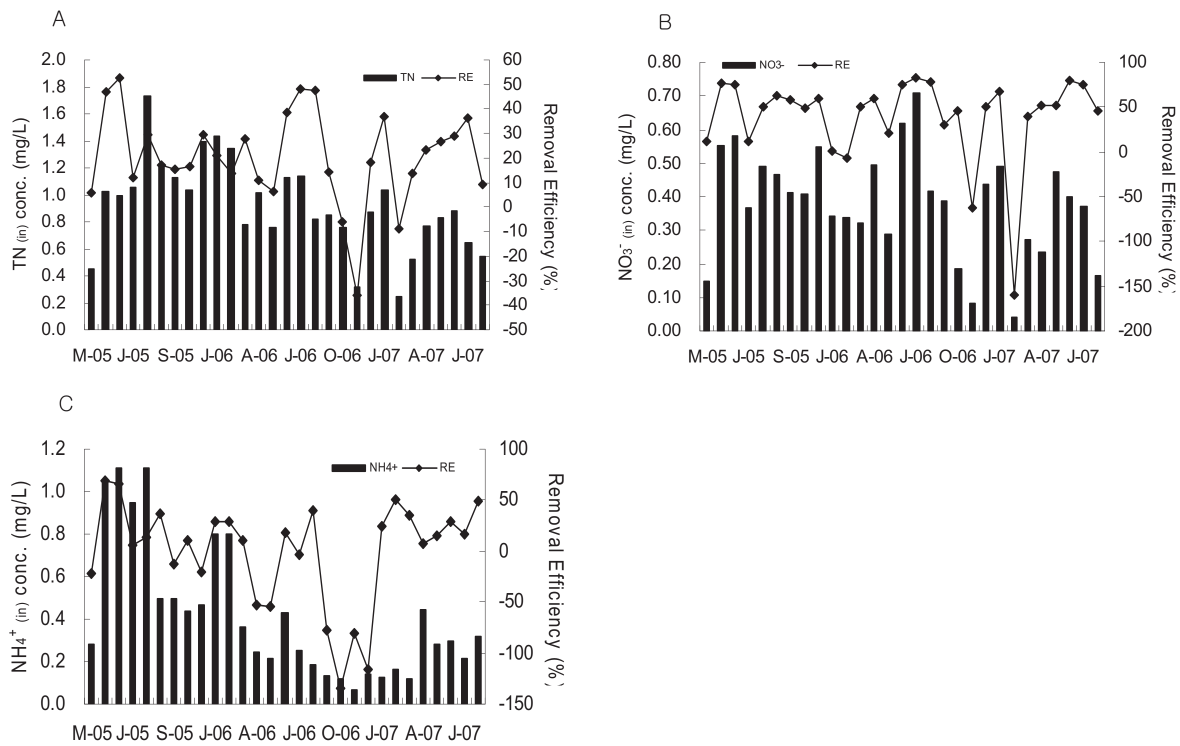


Figure 5-17: A. Relationship between TN inflow concentration (mg/L) and removal efficiency (%), B. Relationship between NO_3^- inflow concentration (mg/L) and removal efficiency (%) and C. Relationship between NH_4^+ inflow concentration (mg/L) and removal efficiency (%)

The removal for N components, which can be seen in Figure 5-17 showed, that the removal performance showed positive relationship between the nutrient load and the removal efficiency. The nutrient concentration at the inflow site seemed to have a positive effect on the removal efficiency, which means that higher nutrient concentrations at the inflow corresponded with an increased removal performance and vice versa. For P components the relationship between the nutrient concentration at the inflow site and removal performance showed a similar pattern as for the N components. Higher P nutrient concentrations seemed to be linked to an increased P removal performance of the reed bed system (Figure 5-18).

For DOC the concentration at the inflow didn't seem to correlate with the removal performance like the previous nutrients (Figure 5-19).

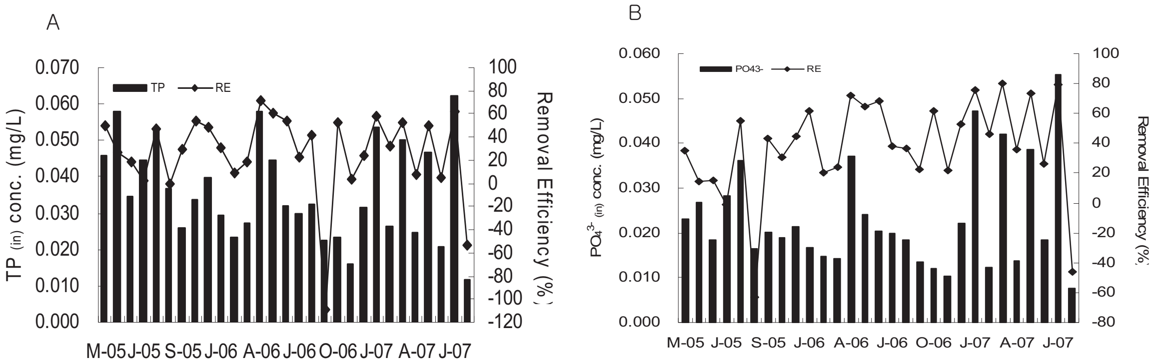


Figure 5-18: A. Relationship between TP inflow concentration (mg/L) and removal efficiency (%) and B. Relationship between PO_4^{3-} inflow concentration (mg/L) and removal efficiency (%)

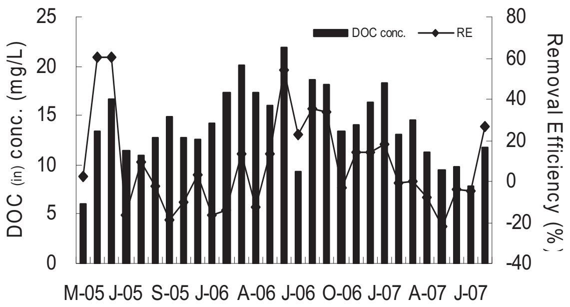


Figure 5-19: DOC inflow concentration (mg/L) and removal efficiency (%)

5.2.5 Hydrological Nutrient Budgets

The hydrological nutrient budgets represent the different nutrients in form of weight.

The budgets were calculated using the following equation: $B_N = V \times C_N$ (5-5),

where B_N is the nutrient budget, V is the water volume (L) and C_N is the nutrient concentration (mg/L).

The budgets were calculated on a monthly basis. The water volume for the different collection sites are shown in Figure 5-20. The inflow and outflow sites had a much smaller volume in comparison to the reed bed (Figure 5-20 A & B).

The water volume in the reed bed showed to be relatively constant, with slightly greater volumes during the winter seasons. Whereas the water volume of the storage pond showed in comparison to the water volume in the reed bed high fluctuation in the water volume, influenced due to the rainfall amount during the whole study period. In general the water volume in the storage pond was the highest during the autumn and winter seasons, which are the rainy seasons in Adelaide and usually the water volume dropped towards the summer seasons.

The average (Table 5-13) and monthly (Figure 5-21, 5-22 and 5-23) for the nutrient budgets for the different sites showed great variations, which are mainly due to the difference in the water volumes for the different sites (Figure 5-20). In general the ANOVA and Tukey's test revealed that the magnitudes of the budgets were significantly higher in the storage than the other sites. However budgets of the inflow and outflow were significantly lower than the rest (Table 5-13).

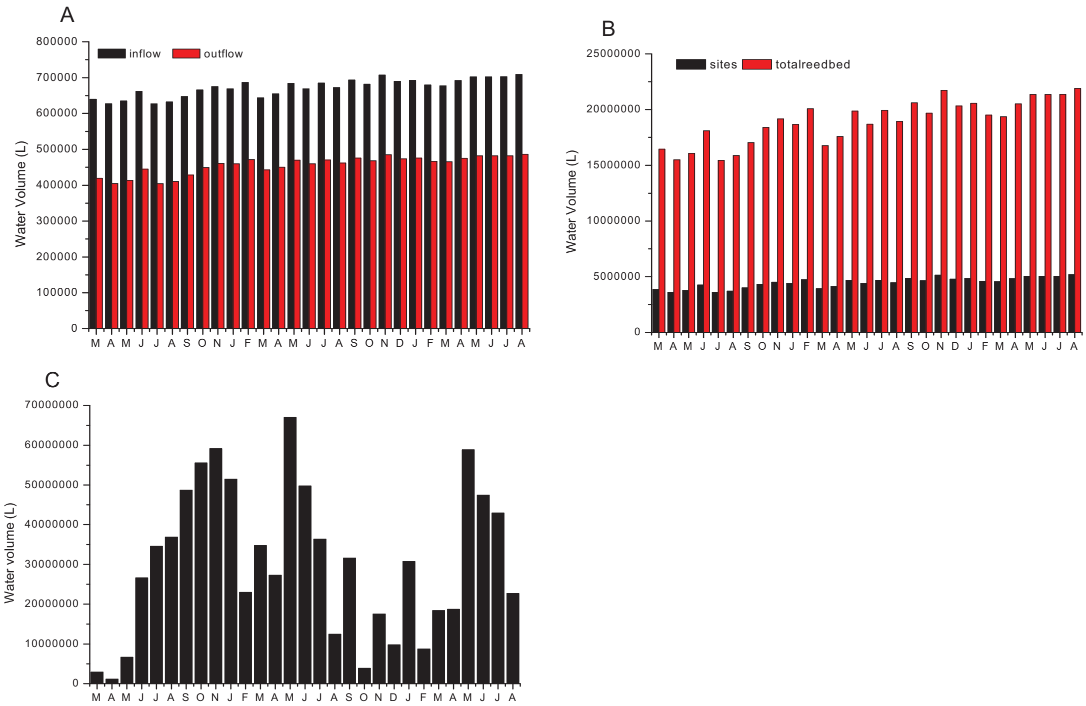


Figure 5-20: Water volume (L) of the reed bed pond and storage pond. A. Water volume of inflow and outflow area; B. Water volume of collection sites and the total reed bed and C. Water volume of the storage pond

Table 5-13: Average Nutrient Budgets (kg) for the different sites

	TN	NO_3^-	NH_4^+	TP	PO_4^{3-}	DOC
storage	29.60^{a}	12.40^{a}	11.12^{a}	1.28^{a}	0.92^{a}	451.23^{a}
inflow	0.62^{b}	0.25^{b}	0.28^{b}	0.02^{b}	$0.02^{b,e}$	9.37^{b}
site 1	4.50^{c}	1.80^{c}	1.94^{c}	0.24^{c}	0.17^{c}	60.52^{c}
site 2	3.92^{c}	1.40^{d}	1.60^{c}	0.19^{c}	$0.13^{c,d}$	61.84^{c}
site 3	3.18^{d}	0.99^{e}	1.39^{c}	0.15^{d}	$0.09^{d,e}$	60.98^{c}
site 4	3.01^{d}	0.82^{e}	1.30^{d}	0.11^{d}	$0.06^{b,e}$	56.84^{c}
outflow	0.32^{b}	0.08^{b}	0.16^{b}	0.011^{b}	$0.006^{b,e}$	5.80^{b}

Average (n=29) in the column followed by different letters indicate significant difference (P<0.05)

Within the reed bed the budgets at sites 1 and 2 had higher magnitudes for most of the measured nutrients, with exception PO_4^{3-} and DOC. In case of the DOC budget there was no significance between the sites 1 to 4. PO_4^{3-} budget showed a declining trend from site 1 to site 4, which also was significantly different (Table 5-13).

The monthly variation of the nutrient budgets in the storage pond showed higher magnitudes in the autumn and winter seasons, due to the greater amount of rainfall. In the spring and summer seasons the nutrient budgets were much smaller in magnitudes (Figure 5-21 A., 5-22 A., and 5-23 A.) The sites located in the reed bed pond showed fewer variations in magnitudes for the nutrient budgets. An explanation for that could be that the water volume was reasonably constant over the whole study period, because the reed bed pond was inundated for the whole year, so that the water volume had less influence on the budgets in comparison to the storage pond.

The monthly budgets for nitrogen compounds showed a decline of the budgets over the whole study period. The nutrient budgets of the reed bed sites showed great magnitudes during different seasons, which didn't show a clear picture compared to the storage pond. A possible explanation was mentioned above, which is that the water volume was relatively consistent, but the concentrations were changing (Figure 5-21 C-F). The monthly budgets for the P compounds showed small variations in the magnitudes for most of the time; however greater magnitudes appeared mostly during the summer seasons and sporadically in the spring seasons (Figure 5-22 C-F). For the DOC budget greater magnitudes were calculated in the summer seasons with having lower budgets during the winter time. The DOC budget didn't show much variation during the whole study period (Figure 5-23 C-F).

The total TN budget for the storage pond over the whole study period was 858.5 kg. The TN budget for the reed sites were respectively inflow (17.9 kg), site 1 (130.5 kg), site 2 (113.6 kg), site 3 (92.3 kg), site 4 (87.3 kg) and outflow (9.4 kg) (Figure D-2 A). The total NO_3^- budget was for the storage pond (359.6 kg), inflow (7.4 kg), site 1 (52.1 kg), site 2 (40.6 kg), site 3 (28.8 kg) site 4 (23.6 kg) and outflow (2.4 kg) (Figure D-2 B). The NH_4^+ budget for the whole study period was for the storage pond (322.6 kg), inflow (8.0 kg), site 1 (56.1 kg), site 2 (46.5 kg), site 3 (40.4 kg), site 4 (37.6 kg) and outflow (4.58 kg) (Figure D-2 C).

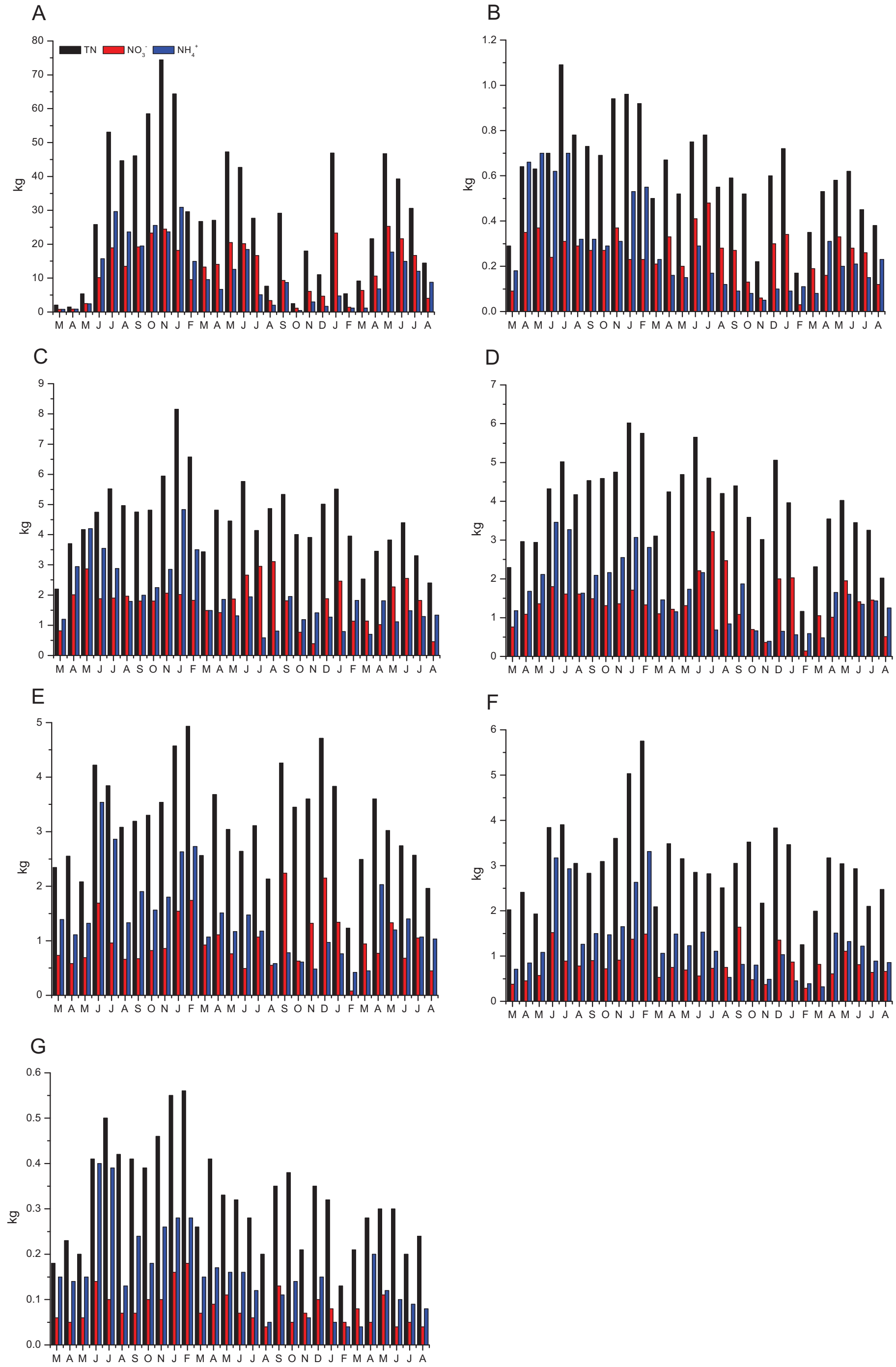


Figure 5-21: Monthly Nitrogen (TN, NO_3^- and NH_4^+) budgets for the different sites in kg. A. storage, B. inflow, C. site 1, D. site 2, E. site 3, F. site 4 and G. outflow.

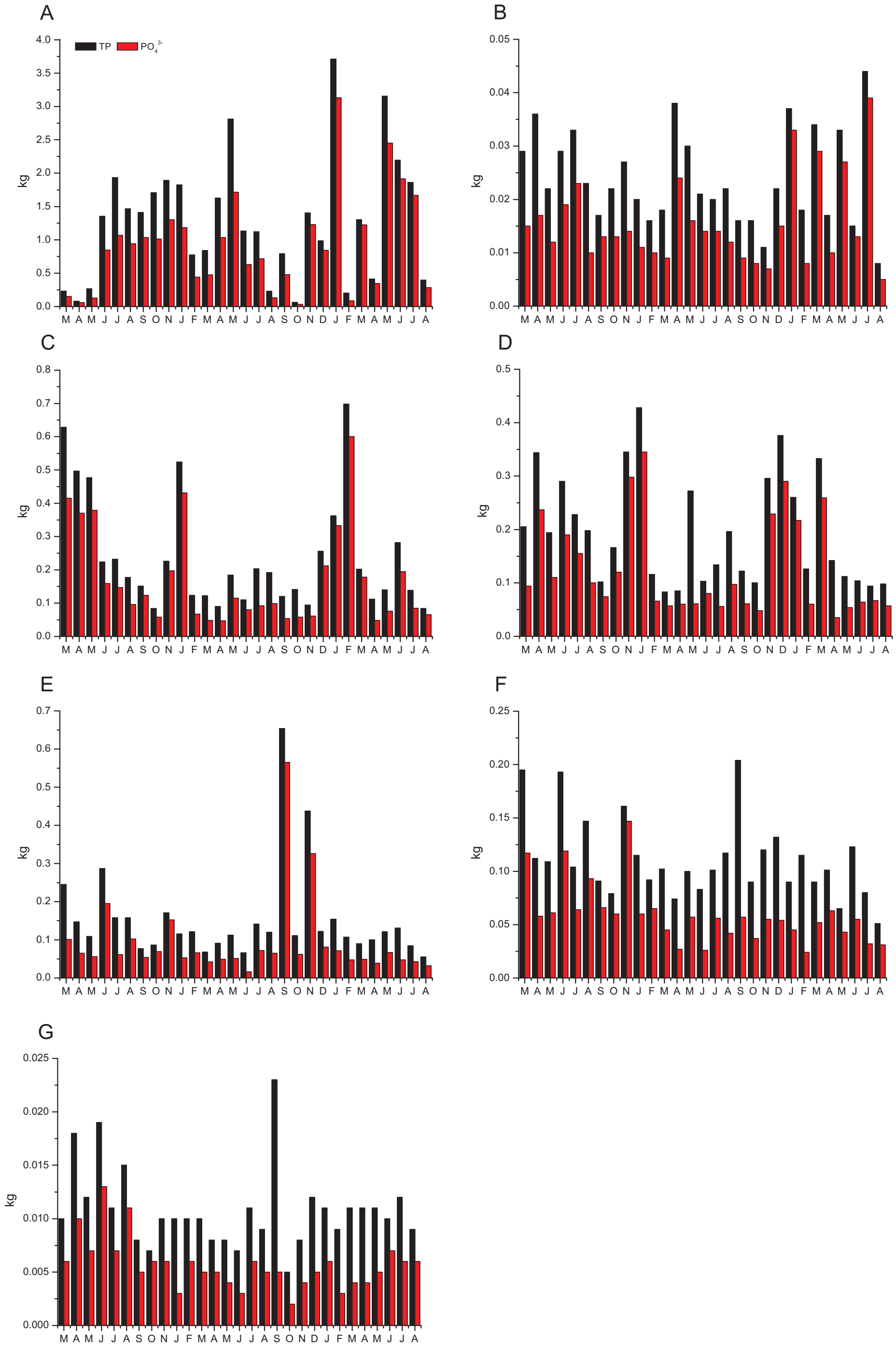


Figure 5-22: Monthly Phosphorus (TP and PO_4^{3-}) budgets for the different sites in kg. A. storage, B. inflow, C. site 1, D. site 2, E. site 3, F. site 4 and G. outflow.

The total TP budget for the storage pond (37.2 kg), inflow (0.7 kg), site 1 (6.9 kg), site 2 (5.7 kg), site 3 (4.4 kg), site 4 (3.2 kg) and outflow (0.3 kg) (Figure D-2 D). The PO_4^{3-} budgets were for the storage (26.6 kg), inflow (0.5 kg), site 1 (4.9 kg), site 2 (3.6 kg), site 3 (2.7 kg), site 4 (1.7 kg) and outflow (0.2 kg) (Figure D-2 E).

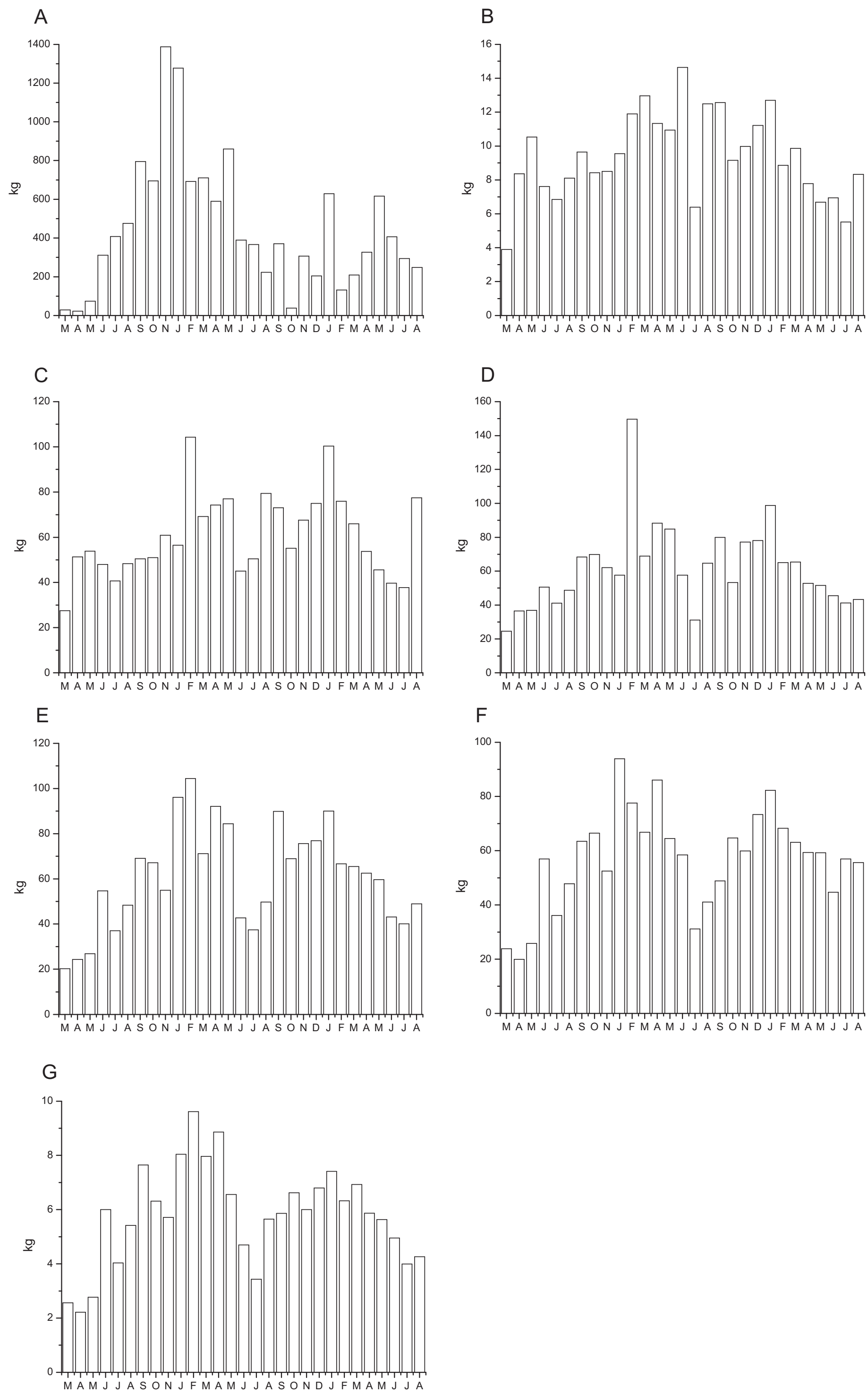


Figure 5-23: Monthly Dissolved Organic Carbon (DOC) budgets for the different sites in kg. A. storage, B. inflow, C. site 1, D. site 2, E. site 3, F. site 4 and G. outflow.

The DOC budget for the storage (13085.7 kg), inflow (271.5 kg), site 1 (1755.2 kg), site 2 (1793.4 kg), site 3 (1768.5 kg), site 4 (1648.3 kg) and outflow (168.1 kg) (Figure D-2 F).

5.3 Macrophytes

5.3.1 General Chemical and Biological Characteristics

The plant species composition in the reed bed pond, which is dominated by *P. australis*, showed changes in the composition. In case of *P. australis* the composition changed from 64.4% to 54.9%. *S. validus* composition changed from 27.3% to 13.0%, *E. sphacelata* composition changed from 2.7% to 1.3%, *B. articulata* composition changed from 5.7 % to 15.5%, and finally the composition of *T. orientalis* changed from 0% to 31.7%. *T. orientalis* originally wasn't planted in the reed bed pond, but invaded the system by natural causes, possibly due to the seed bank in the sediment or due to natural invasion. The plant species composition of the whole reed bed can be seen in Figure 5-24 A.

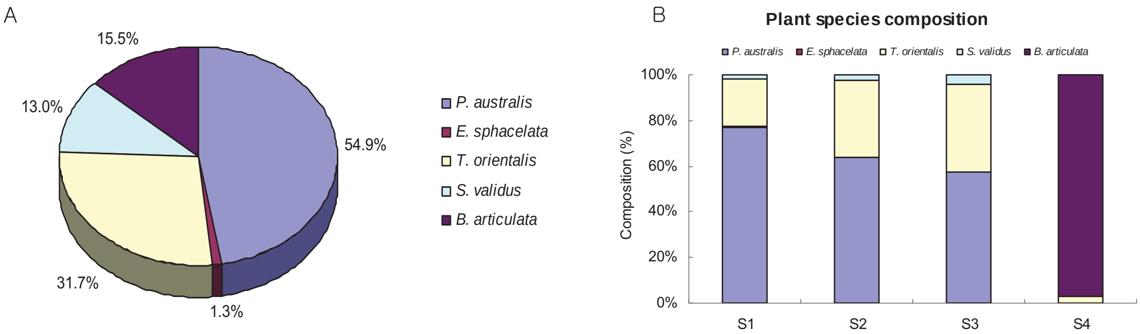


Figure 5-24: A. Plant species composition (in %) for the whole reed bed pond; B. Plant species composition of different collection sites. (S1 = site 1, S2 = site 2, S3 = site 3 and S4 = site 4)

Table 5-14: Biological and chemical plant parameters from the different collection sites over the study period (2005-2007)

Parameters	Site	average	S.D.	S.E.	Minimum	Maximum
Dry weight (DW) g/m^2	**Site 1**	3768.2	1222.4	235.3	1979.1	6622.4
	Site 2	3083.9	1024.4	197.1	1493.8	4789.0
	Site 3	1282.2	524.7	174.9	644.4	1971.0
	Site 4	2291.8	592.8	114.1	1366.4	3518.2
Total Nitrogen (TN) mg/kg	**Site 1**	20617.7	8881.1	1709.2	7865.0	34205.5
	Site 2	18256.6	6408.7	1233.4	8112.5	31806.6
	Site 3	14793.4	5082.6	1694.2	5800.9	21236.5
	Site 4	13147.0	2139.0	411.6	7770.8	18138.6
Total Phosphorus (TP) mg/kg	**Site 1**	4667.9	2956.1	568.9	1329.3	10898.5
	Site 2	4775.2	3142.5	604.8	1381.2	12937.4
	Site 3	2330.1	1384.5	461.5	252.1	5512.7
	Site 4	2843.2	1530.3	294.5	565.5	6153.8

S.D. = standard deviation; S.E. = standard error

The plant species composition for the different collection sites (Figure 5-24 B.) showed that the site 1 to 3 were dominated by *P. australis*, which is ranging between 50% - 75%, followed by *T. orientalis* (20% - 40%). Site 4 was completely dominated by *B. articulata*, which is over 90%. The "biggest loser" seems to be S. validus, where the plant species composition decreased by 50% compared to original composition. ANOVA and Tukey's test were performed on the measured plant parameters to determine possible significant differences between sites (Table 5-14, 5-15; Figure 5-25). The biological and chemical parameters were measured based on the most dominant species at the different sites. The spatial differences in the measured plant parameters showed high significantly differences between the different sites (P<0.001, P<0.01 and P<0.05).

Table 5-15: ANOVA table for biological and chemical plant parameters for the different collection sites. In case the ANOVA test confirmed a significance, post-test (Tukey's analysis) was performed to compare the individual datasets

Parameters	ANOVA	Site-Site	DW	TN	TP
DW	***	**S1-S2**	*	N.S.	N.S.
TN	***	**S1-S3**	***	N.S.	N.S.
TP	**	**S1-S4**	***	***	*
		S2-S3	***	N.S.	N.S,
		S2-S4	*	*	*
		S3-S4	*	N.S.	N.S.

*****, Extremely significant difference (P<0.001); **, moderately significant difference (P<0.01); *, significant difference (P<0.05); N.S., no significant (P>0.05); α = 0.05, n=27, only for site 3 α = 0.05, n = 9; S1 = site 1; S2 = site 2; S3 = site 3; S4 = site 4**

The result of the statistical analysis on the means of dry weight (DW) showed significant differences between all four sites, with site 1 having a significantly higher dry weight than the all the other sites (Table 5-15, Figure 5-25). In case of site 3 the analysis includes data only from the first year of the monitoring. From the second year until the end of the monitoring no data were collected due to plant harvesting and no re-growth of the macrophytes. In general the trend showed that the sites close to the inflow had significantly higher dry weights than the sites close to the outflow (P<0.001 and P<0.05) (Table 5-14, 5-15; Figure 5-25).

The means of TN concentration showed a similar pattern than the DW. The sites close to the inflow showed significantly higher levels in the TN concentration than the sites close to the outflow (P<0.001 and P<0.05). The TN concentration between site 1 and site 2 showed no significant differences (P>0.05). The results for TP was the same like

TN, with a lower significance, however the levels of TP was significantly higher in the sites close to the inflow than the sites close to the outflow (Table 5-14, 5-15; Figure 5-25).

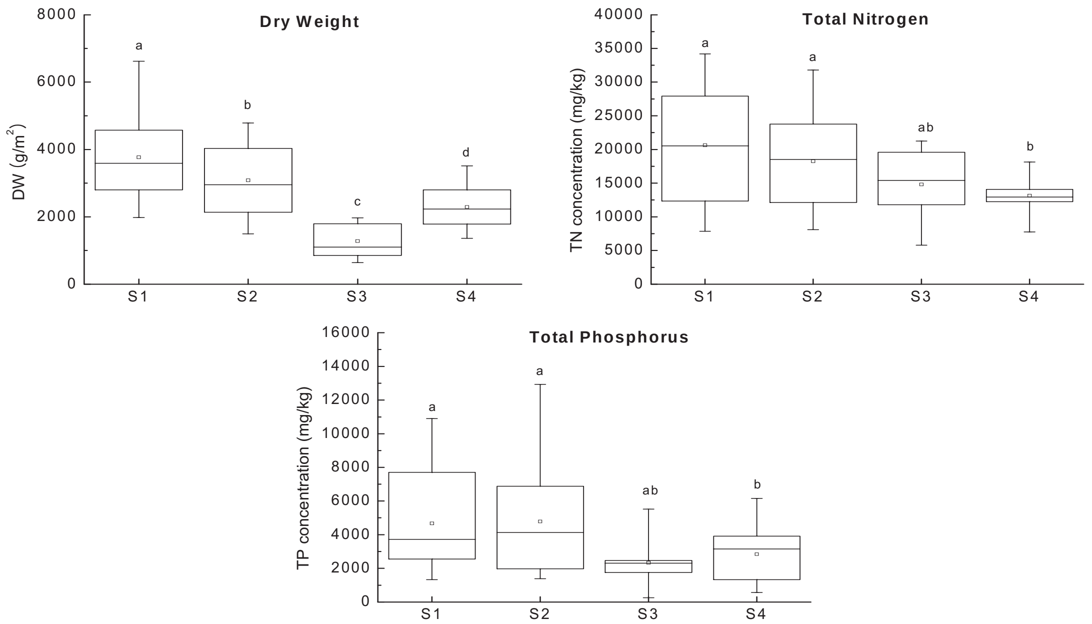


Figure 5-25: Bar and whisker graphs of biological and chemical plant parameters for the different collection sites. The columns marked with different letters indicate significant difference according to Tukey's test (α = 0.05, n= 27, only for site 3 α = 0.05, n= 9). S1 = site 1, S2 = site 2, S3 = site 3, S4 = site 4

Based on the monitoring results of the dry weight and retention of the different nutrients of the macrophytes for the reed bed pond at the Parafield Stormwater Harvesting Facility (Figure 5-26) showed that the dry weight was highest between February and March during the study period, which also corresponded with the nutrient concentrations for TN and TP. The productivity measured in form of the dry weight showed that site 1 and 2 had significantly higher production than site 3 and 4. The dry weight for site 1 was ranging between 1979 g/m^2 and 6622.4 g/m^2. Dry weight at site 2 was ranging between 1493.8 g/m^2 and 4789.0 g/m^2. For site 3 it was ranging between 644.4 g/m^2 and 1971.0 g/m^2, and for site 4 it was ranging between 1366.4 g/m^2 and 3518.2 g/m^2. The nutrient concentrations regards TN and TP were highest at site 1 and 2 than site 3 and 4. TN at site 1 was ranging between 16.9 g/m^2 and 226.5 g/m^2, for site 2 it is ranging between 12.8 g/m^2 and 149.7 g/m^2, for site 3 it is ranging between 3.8 g/m^2 and 40.0 g/m^2, and for site 4 it is ranging between 10.6 g/m^2 and 63.8 g/m^2. TP at site 1 was ranging between 2.9 g/m^2 and 60.6 g/m^2, for site 2 it is ranging between 2.1 g/m^2

and 56.7 g/m^2, for site 3 it is ranging between 0.2 g/m^2 and 8.4 g/m^2, and for site 4 it is ranging between 0.8 g/m^2 and 19.2 g/m^2.

For site 3 only data were available from March 2005 until November 2005, which doesn't give a clear picture of the variations and dynamics in regards productivity and nutrient concentrations.

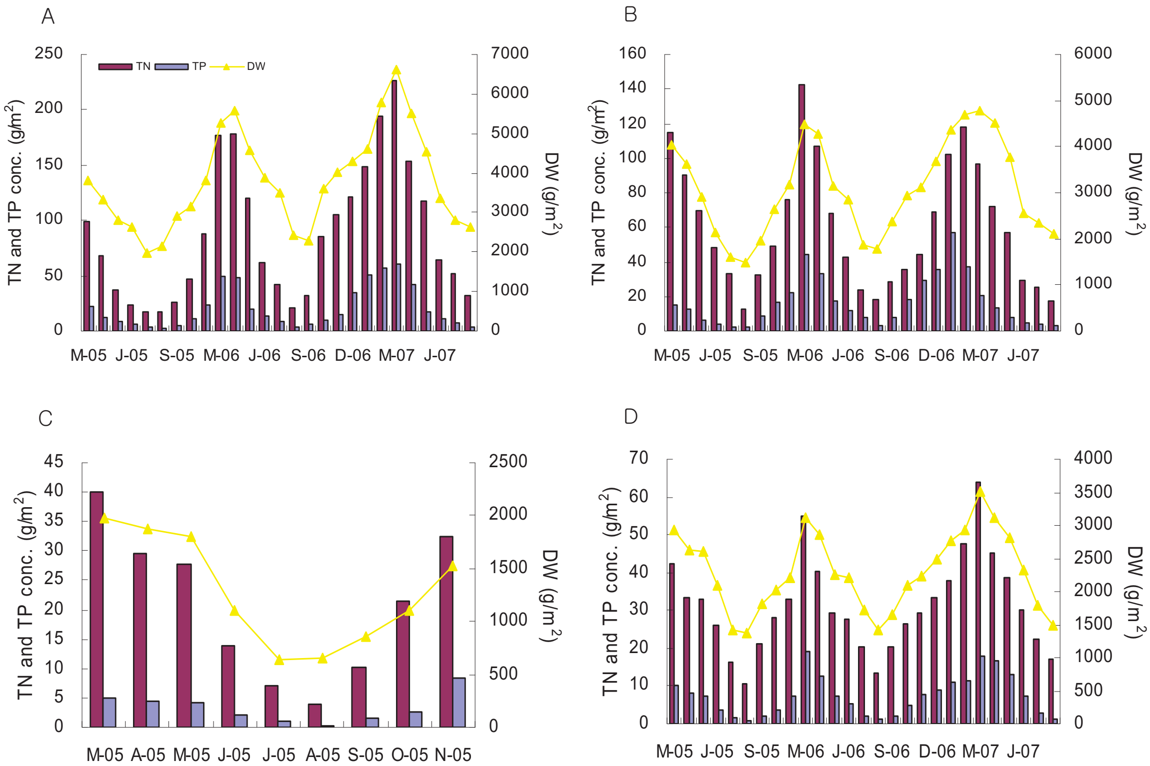


Figure 5-26: Monitoring results of above ground plant dry weight and nutrient concentration. A. Site 1, B. Site 2, C. Site 3 and D. Site 4

5.3.2 Seasonality

The seasonal patterns for plant parameters from the different sites showed a distinguished difference between the non-growing and growing seasons (Figure 5-27). The physical parameter measured in form of dry weight showed a difference in the seasonal pattern.

The dry weight was for most of the vegetated sites the highest in summer with exception of site 3. An explanation for that will be provided later. The same seasonal patterns were observed in the measured chemical parameters in form of total nitrogen and total phosphorus. The levels of concentrations for total nitrogen and total phosphorus were the highest during summer, with exception of site 3 (Figure 5-27).

Site 3 had a different seasonal pattern for the physical and chemical parameters. The reason for the seasonal difference was due to the harvesting of the above-ground biomass at the beginning of the second year (Feb. 2006) of the monitoring program. After the harvesting of the macrophyte population at site 3, the population didn't recover until the end of study period, so that the only data from the first year covering the period from March 2005 to November 2005, which is excludes the summer season (Figure 5-27).

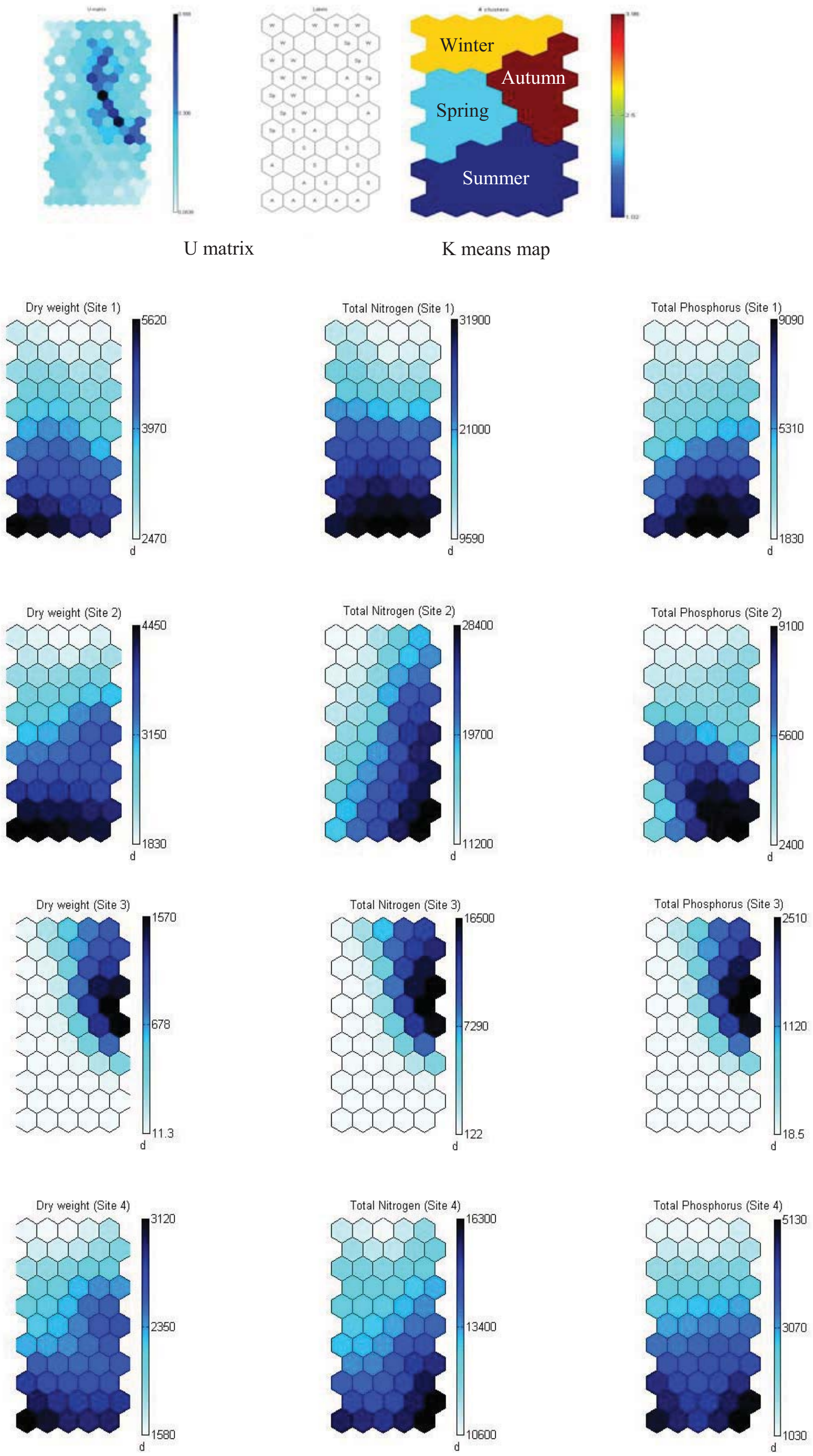


Figure 5-27: Seasonal patterns visualized as a U matrix, K means map and SOM maps for measured plant parameters for the different sites

5.3.3 Annual pattern

The annual pattern for the measured pant parameters showed significant differences for dry weight and total phosphorus (Figure 5-28 A & C). In case of the dry weight the ANOVA and Tukey's test results showed, that the biomass in year 2005 was significantly lower than 2006 and 2007. The biomass in 2007 seems to have higher magnitude than 2006, but the statistical analysis revealed that there was no significant difference between both years (Figure 5-28 A).

For total phosphorus the statistical analysis revealed, that the levels of TP concentration in year 2005 was significantly lower than year 2006 and 2007. No significant difference existed between the annual concentration of the years 2006 and 2007 (Figure 5-28 C). ANOVA and Tukey's results showed no significant difference between the years for the levels of TN concentrations (Figure 5-28 B).

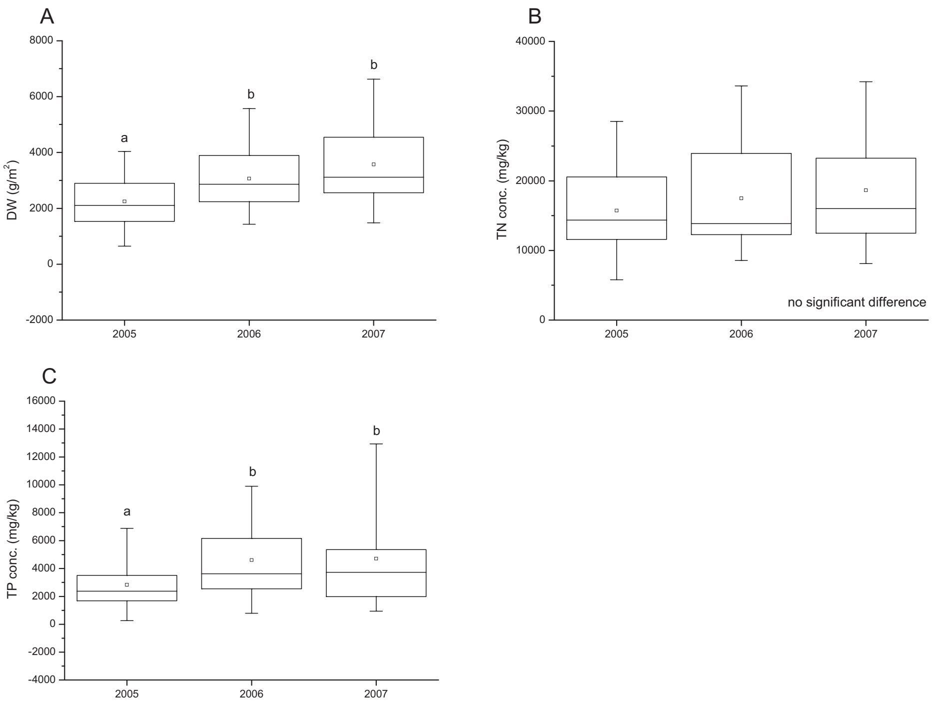


Figure 5-28: Annual comparison of plant parameters. A. Dry Weight, B. Total Nitrogen and C Total Phosphorus. The columns marked with different letters indicate significant difference according to Tukey's test (α = 0.05; 2005 n= 36, 2006 n = 30 and 2007 n = 24)

5.3.4 Plant Nutrient Budgets

The average (Table 5-16) and monthly (Figure 5-29 and 5-30) plant nutrient budgets for the different collection sites. The budgets were determined by using the following

equation: $B_N = DW_T \times C_N$ (5-6)

where B_N is the nutrient budget, DW_T is total dry weight (kg) and C_N is the concentration/mass (mg/kg).

Figure 5-29 is showing the total dry weight for the different site area in kg, for site 1 (Figure 5-29 A), site 2 (Figure 5-29 B), site 4 (Figure 5-29 C) and site 4 (Figure 5-29 D). The average dry weight for site 1 is 17.3t, for site 2 it is 14.2t, for site 3 it is 5.9t and site 4 it is 10.5t. The average dry weight for the different sites showed that the sites near inflow have significantly higher levels of dry weight compared to the sites near the outflow (Figure 5-29). The dry weight of site 3 showed significantly lower levels, due to the harvesting at the beginning of the second year of the monitoring scheme, which doesn't include the summer and autumn season. After the harvesting of the above-ground biomass, the macrophyte population didn't recover, so that no data could be collected regards the dry weight and nutrients until the end of the study period. This is the reason for lower levels in biomass and nutrients.

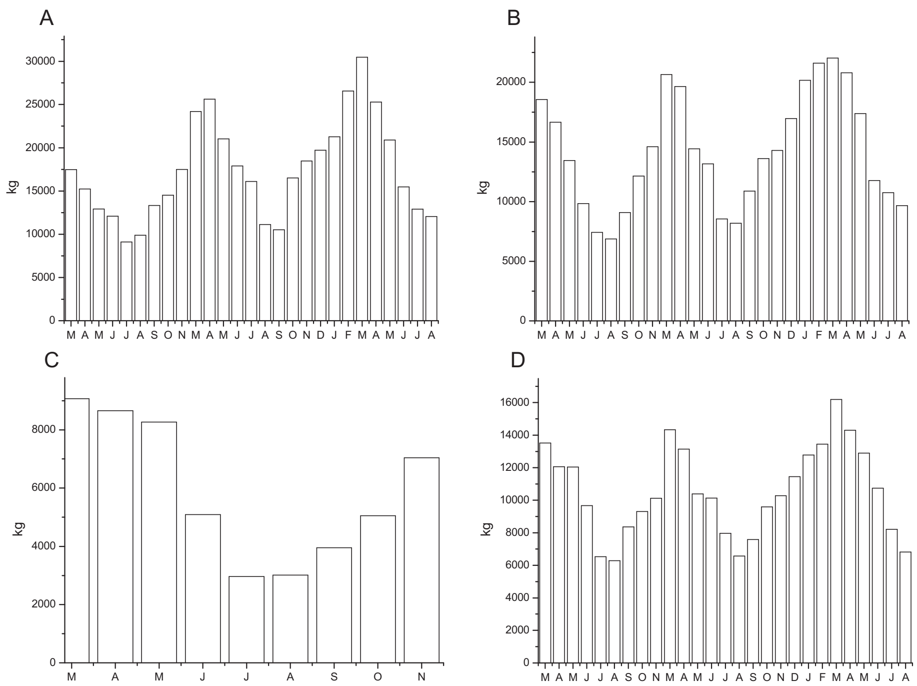


Figure 5-29: Monthly dry weight (DW) for the different sites in kg. A. site 1, B. site 2, C. site 3, and D. site 4

The average nutrient budgets for TN and TP for the above-ground for the different sites showed after performing ANOVA and Tukey's test, that the nutrient budgets of site 1 and 2 were significantly higher than site 3 and 4 (Table 5-16). Site 1 had the highest levels of budgets for TN and TP, but there was no significant difference between the budget levels of site 1 and 2 (P>0.05).

Table 5-16: Average Nutrient Budgets (kg) for the different sites

	TN	TP
site 1	401.3[a]	94.8[a]
site 2	277.3[a]	76.0[a]
site 3	95.0[b]	15.3[b]
site 4	143.4[b]	33.7[b]

Average (n=27, only for site 3 n=9) in the columns followed by different letters indicate significant difference (P<0.05)

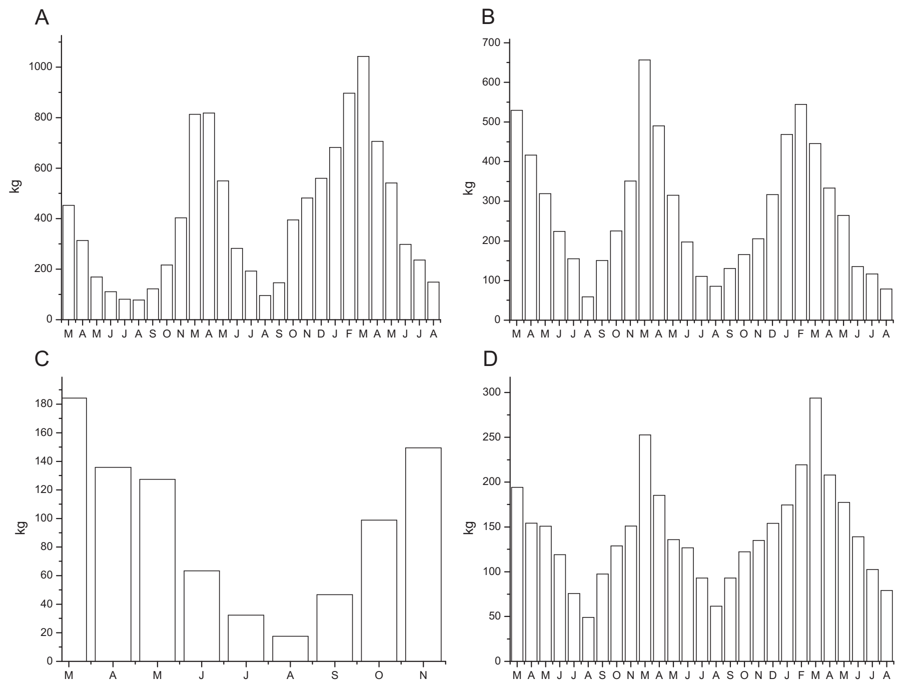


Figure 5-30: Monthly Total Nitrogen (TN) budgets for the different sites in kg. A. site 1, B. site 2, C. site 3 and D. site 4.

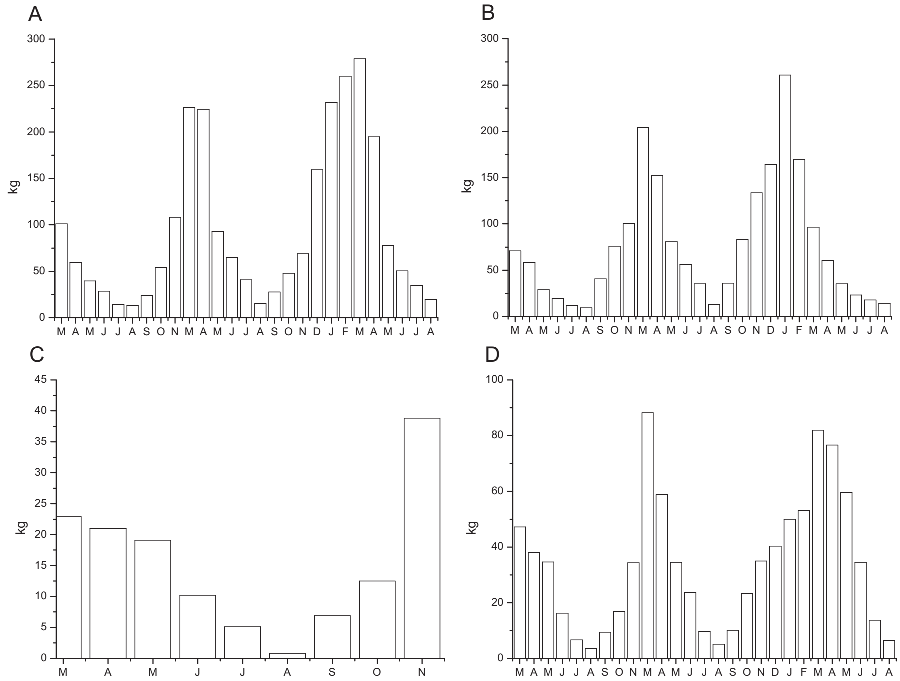


Figure 5-31: Monthly Total Phosphorus (TP) budgets for the different sites in kg. A. site 1, B. site 2, C. site 3 and D. site 4.

The monthly budgets for TN and TP showed for all sites a similar pattern with higher levels in the budgets during the growing seasons, which also corresponds with pattern for the dry weight. The budget levels during the non-growing seasons were much lower in comparison to the budget levels during the growing seasons (Figure 5-30 and 5-31). The total TN budgets for the plant population for the different sites showed distinguishable variations. Site 1 has a total TN budget of 10.8t during the whole study period. Site 2 has a total TN budget of 7.5t, site 3 has a TN budget of 0.9t and site 4 has a TN budget of 3.9t.

The total TP budget showed the same pattern like the TN budget. Site 1 had the highest budget with 2.6t followed by site 2 with 2.1t. Site 3 and site 4 have a total TP budget of 0.1t and 0.9t. In both nutrient forms site 3 had the lowest levels

5.3.5 Plant Nutrient Uptake

Plant nutrient uptake by the macrophyte community was determined by calculating the differences between the months. Figure 5-32 is showing a comparison of the monthly nutrient uptake patterns of the different sites, which showed that there were differences of the uptake rates. Site 1 has overall the highest nutrient uptakes, which is ranging for TN from 1.1g/m^2 to 89.2g/m^2. TP nutrient uptake by plants is ranging from 2.4g/m^2 to 25.7g/m^2. The TN uptake at site 2 is ranging between 7.6g/m^2 and 66.3g/m^2 and the TP uptake is between 5.0g/m^2 and 22.6g/m^2. The nutrient uptake at site 3 is ranging for TN between 6.4g/m^2 and 11.3g/m^2, and for TP between 1.2g/m^2 and 5.7g/m^2.

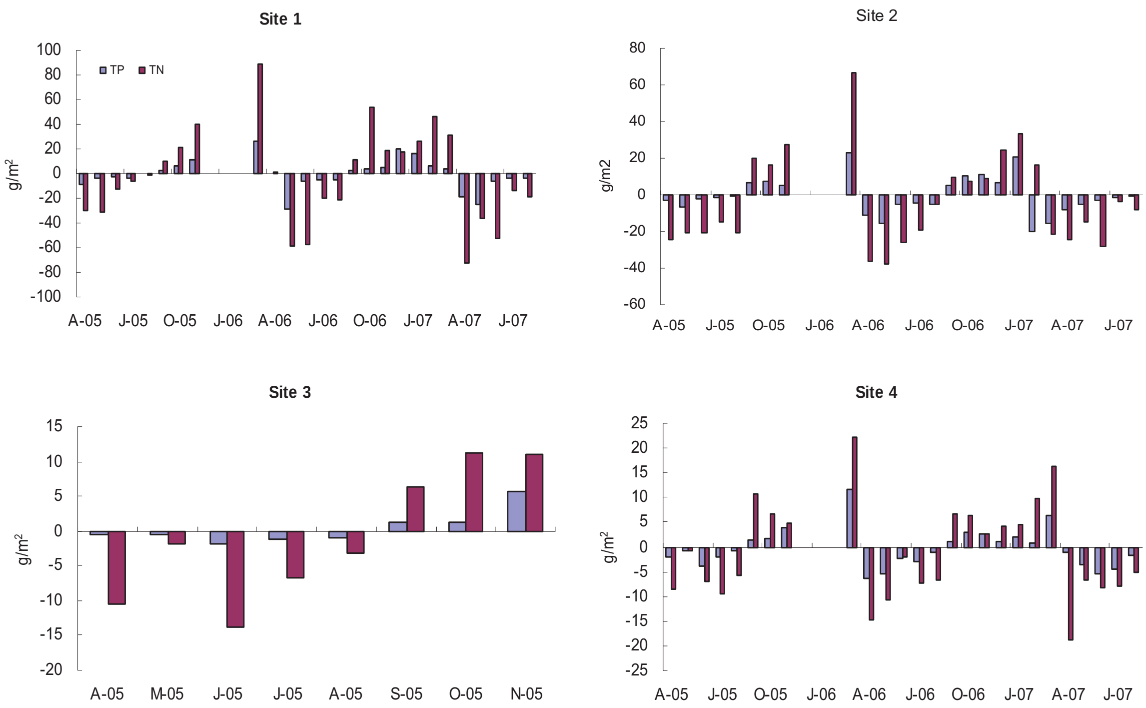


Figure 5-32 Monthly TP and TN uptake rates (g/m^2) for the different sites. Positive bars show the nutrient uptake rate, whereas the negative bars would show a release of nutrients.

At site 4 the nutrient uptake for TN is ranging from 2.8g/m^2 to 22.1g/m^2, and for TP it is between 0.7g/m^2 and 11.7g/m^2. Overall the ranges of nutrient uptakes reveal higher uptake rates at the sites near the inlet, which are site 1 and 2, in comparison to the sites located near the outlet area, which are site 3 and 4. In case of site 3 the values for the nutrient uptake rates were smaller in range due to the harvesting after the first year of monitoring.

To rank the nutrient uptake rates for the different sites, it is like the following: site 1 > site2 > site 4 > site 3. The higher nutrient uptake rates are reflected as primary productivity in form of biomass, which are significantly higher at the sites located near the inflow (Figure 5-32).

5.3.6 Plant Harvesting

At the begin of 2006 (end of January-begin of February) plant harvesting was introduced in the reed bed pond of the Parafield Stormwater Harvesting, which involved the complete removal of the above-ground biomass and organic matter above the sediment level. The harvesting was performed using a harvesting boat and took around 3 days. A total area of 400m^2 was harvested at the location of the sampling site 3. The monitoring of the water column and sediment continued during and after the harvesting period, to compare the differences of the nutrient dynamics in the sediment and water column. The vegetation in form of the macrophyte community didn't recover after the harvesting; therefore no plant data are available for sampling site 3, which is also shown in the previous section. Instead of the re-growth of macrophyte community a green algae from the family of Spirogyra started to grow at the harvested area.

The comparison of the nutrient dynamics and the chl a concentration in the water column before and after the harvesting, as shown in Figure 5-33, showed that there was a change in the nutrient dynamics as well as in the chl a concentration. The chl a concentration at site 3 changed after the harvesting, which showed an increasing trend. The average chl a concentration before the harvesting was 34.41µg/L (min. 7.35µg/L and max. 113.36µg/L), but after the harvesting the average chl a concentration increased to 55.70µg/L (min. 8.80µg/L and max. 139.09µg/L).

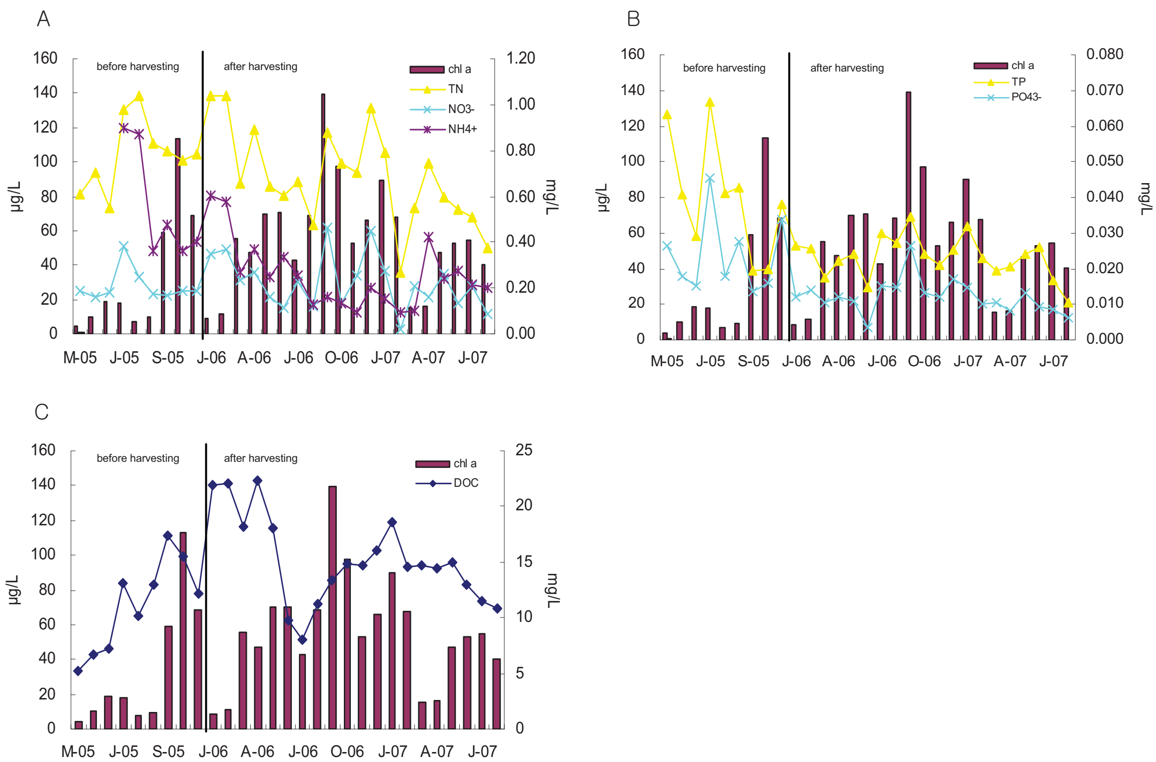


Figure 5-33: Nutrient concentration dynamics and chlorophyll a concentration in the water column before and after the harvesting. A. Nitrogen components, B. Phosphorus components and C. DOC components

The changes for the nitrogen components (Figure 5-33 A) showed that the average concentrations of TN and NH_4^+ decreased after the harvesting. Average TN concentration was 0.78mg/L (min. 0.55mg/L and max. 1.04mg/L) before the harvesting and decreased after the harvesting to 0.69mg/L (min. 0.27mg/L and max. 1.04mg/L). In case of NH_4^+ the average concentration decreased from 0.56mg/L (min. 0.36mg/L and max. 0.90mg/L) to 0.25mg/L (min. 0.09mg/L and max. 0.60mg/L). NO_3^- average concentration showed an opposite trend in comparison to previous nitrogen components. The concentration increased after the harvesting from 0.21mg/L (min. 0.16mg/L and max. 0.38mg/L) to 0.23mg/L (min. 0.02mg/L and max. 0.46mg/L). The comparison of the phosphorus components in form of TP and PO_4^{3-} before and after the harvesting, shown in Figure 5-33 B, showed that the average concentrations decreased. For TP the average concentration decreased from 0.040mg/L (min. 0.019mg/L and max. 0.067mg/L) to 0.023mg/L (min. 0.011mg/L and max. 0.035mg/L). The concentration of PO_4^{3-} changed from 0.024mg/L (min. 0.013mg/L and max. 0.045mg/L) to 0.012mg/L (min. 0.004mg/L and max. 0.026mg/L). The comparison of the DOC concentration before and after the harvesting, shown in Figure 5-33 C, showed that the concentration of DOC was increasing. The average concentration of DOC before the harvesting was 11.17mg/L (min. 5.26mg/L and max. 17.32mg/L) and increased to 15.14mg/L (min. 7.99mg/L and max. 22.34mg/L) after the harvesting.

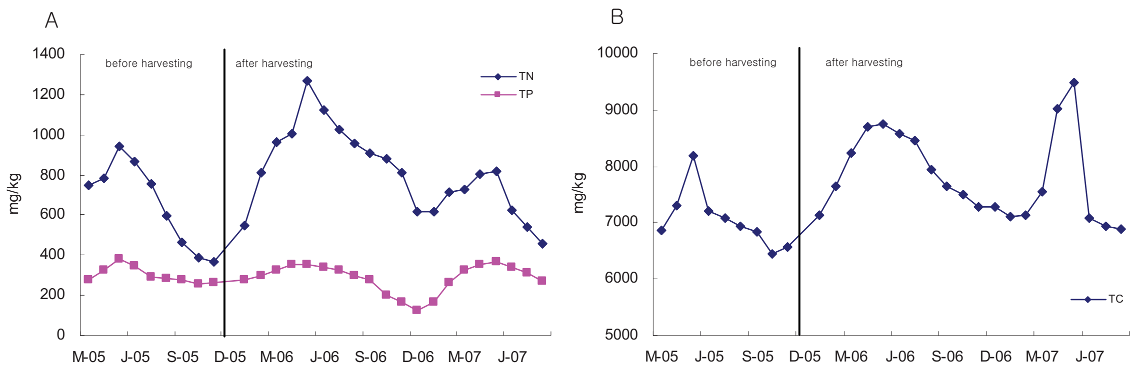


Figure 5-34: Nutrient concentration dynamics in the sediment before and after the harvesting. A. TN and TP and B. TC

The comparisons of the different nutrient concentrations in the sediments before and after the harvesting were shown in Figure 5-34. The dynamics of the different nutrients showed that there were changes in the average concentrations before and after the harvesting. The average concentration of TN before the harvesting was 656.9mg/kg (min. 366.2mg/kg and max. 941.4 mg/kg) and increased after the harvesting to 810.4mg/kg (min. 456.5mg/kg and max. 1264.9mg/kg). The TP concentration dynamics of the sediment showed that the average concentrations before and after the harvesting has decreased from 299.3mg/kg (min. 253.8mg/kg and max. 378.5mg/kg) to 286.9mg/kg (min. 125.6mg/kg and max. 370.4mg/kg). The TC concentration in the sediment showed that the average concentrations before and after the harvesting has increased from 7044.6mg/kg (min. 6450.6mg/kg and max. 8175.6mg/kg) to 7813.6mg/kg (min. 6885.5mg/kg and max. 9477.7mg/kg).

5.4 Sediment

5.4.1 General Physical and Chemical Characteristics

The sediment characteristics in from of the soil textures showed that the sediment types had different compositions of sand, clay and silt (Figure 5-35)

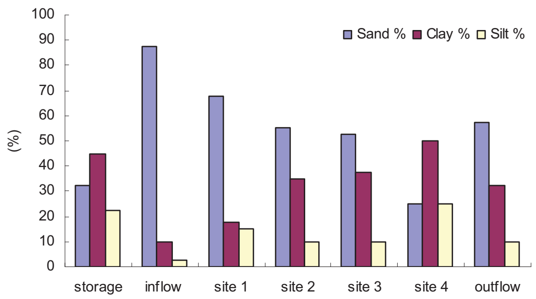


Figure 5-35: Comparison of the sediment characteristics in regards composition for the collection sites

The comparison of the sediment showed different compositions of sand, clay and silt between the collection sites. The composition of the sediment at the storage pond had a higher percentage of clay (45%) followed by sand (32.5%) and silt (22.5%), which falls under the category of clay after the classification of USDA (United States Department of Agriculture). Most of the other collection sites, with exception of site 4, had sand with highest percentage, but by comparing the composition it showed that the sand composition in the sediment seemed to decline towards the outflow. With the decline in sand an increase of the clay particle can be observed. After USDA classification the sediment at the inflow is loamy sand, site 1 is sandy loam, site 2 and site 3 are clay loam, site 4 is silty clay, and outflow is clay loam. Based on the classification we have four different types of sediment within the storage and reed bed pond, with three different sediment types in the reed bed pond (Figure 5-35).

Table 5-17 shows the comparison of the average, standard deviation (S.D.), standard error (S.E.) and the range (minimum and maximum) of measured chemical (TN, TP, and TC), and physical (pH and conductivity) parameters from the different collection sites. ANOVA and Tukey's test were performed on the measured sediment parameters to determine if significant differences exist between sites. The result shows less complexity compared to the differences in water quality parameters (Table 5-17, 5-18; Figure 5-36). The measured physical parameters in form of pH and conductivity (EC) showed in the majority no significant difference between the sites ($P>0.05$), with a few exceptions

Table 5-17: Physical and chemical sediment parameters from the different collection sites over the study period (2005-2007)

Parameters	Site	average	S.D	S.E	Minimum	Maximum
pH	**storage**	8.25	0.31	0.06	7.55	8.77
	inflow	8.31	0.33	0.06	7.34	8.82
	site 1	8.04	0.31	0.06	7.35	8.66
	site 2	8.11	0.22	0.04	7.72	8.54
	site 3	7.91	0.39	0.07	6.63	8.43
	site 4	8.15	0.41	0.08	6.99	8.69
	outflow	8.18	0.25	0.05	7.57	8.52
Conductivity (EC) mS/cm	**storage**	0.107	0.048	0.009	0.042	0.255
	inflow	0.117	0.036	0.007	0.058	0.222
	site 1	0.134	0.035	0.006	0.089	0.225
	site 2	0.156	0.037	0.007	0.089	0.232
	site 3	0.130	0.038	0.007	0.052	0.225
	site 4	0.137	0.037	0.007	0.058	0.226
	outflow	0.132	0.036	0.007	0.100	0.246
Total Carbon (TC) mg/kg	**storage**	4971.63	2654.82	492.99	322.10	9147.80
	inflow	4671.44	2094.36	388.91	1195.30	8465.50
	site 1	7435.78	587.08	109.02	6564.60	9047.60
	site 2	7664.70	667.82	124.01	6061.30	9244.00
	site 3	7574.98	783.37	145.47	6450.60	9477.70
	site 4	7501.80	600.31	111.47	6320.70	8950.80
	outflow	5544.00	830.82	154.28	3550.00	7185.50
Total Nitrogen (TN) mg/kg	**storage**	571.71	252.79	46.94	170.20	1131.00
	inflow	554.36	280.14	52.02	96.10	1125.00
	site 1	762.68	209.22	38.85	280.70	1273.80
	site 2	773.05	191.24	35.51	315.80	1208.30
	site 3	762.73	219.50	40.76	366.20	1264.90
	site 4	738.06	205.99	38.25	332.10	1247.00
	outflow	644.47	208.50	38.72	296.10	1184.50
Total Phosphorus (TP) mg/kg	**storage**	274.67	64.98	12.07	129.00	393.60
	inflow	235.56	72.48	13.46	117.50	378.10
	site 1	287.67	69.27	12.86	145.00	420.50
	site 2	280.44	65.81	12.22	127.90	395.60
	site 3	290.72	62.69	11.64	125.60	378.50
	site 4	309.14	60.04	11.15	162.10	420.70
	outflow	276.37	59.60	11.07	146.10	375.60

S.D. = standard deviation; S.E. = standard error

The physical parameters pH and conductivity (EC) showed in the majority of the sites no significant (P>0.05), with a few exceptions. In case of pH site 3 showed a significantly lower level of pH in comparison to the non-vegetated sites (P<0.001, P<0.01 and P<0.05). The pH of site 1 showed significantly lower levels in comparison to the inflow (P<0.05). In case of conductivity the level at site 2 was significantly higher in comparison to storage and inflow (P<0.001 and P<0.01). The conductivity level at site 4 was significantly higher than the storage (P<0.05) (Table 5-17, 5-18; Figure 5-36).

Table 5-18: ANOVA table for physical and chemical sediment parameters for the different collection sites. In case the ANOVA test confirmed a significance, post-test (Tukey's analysis) was performed to compare the individual datasets

Parameters	ANOVA	Site-Site	pH	EC	TC	TN	TP
pH	***	**sto-in**	N.S.	N.S.	N.S.	N.S.	N.S.
EC	***	**sto-S1**	N.S.	N.S.	***	*	N.S.
TC	***	**sto-S2**	N.S.	***	***	*	N.S.
TN	***	**sto-S3**	**	N.S.	***	*	N.S.
TP	**	**sto-S4**	N.S.	*	***	N.S.	N.S.
		sto-out	N.S.	N.S.	N.S.	N.S.	N.S.
		in-S1	*	N.S.	***	**	*
		in-S2	N.S.	**	***	**	N.S.
		in-S3	***	N.S.	***	**	*
		in-S4	N.S.	N.S.	***	*	**
		in-out	N.S.	N.S.	N.S.	N.S.	N.S.
		S1-S2	N.S.	N.S.	N.S.	N.S.	N.S.
		S1-S3	N.S.	N.S.	N.S.	N.S.	N.S.
		S1-S4	N.S.	N.S.	N.S.	N.S.	N.S.
		S1-out	N.S.	N.S.	***	N.S.	N.S.
		S2-S3	N.S.	N.S.	N.S.	N.S.	N.S.
		S2-S4	N.S.	N.S.	N.S.	N.S.	N.S.
		S2-out	N.S.	N.S.	***	N.S.	N.S.
		S3-S4	N.S.	N.S.	N.S.	N.S.	N.S.
		S3-out	*	N.S.	***	N.S.	N.S.
		S4-out	N.S.	N.S.	***	N.S.	N.S.

*****, Extremely significant difference (P<0.001); **, moderately significant difference (P<0.01); *, significant difference (P<0.05); N.S., no significant (P>0.05); α = 0.05, n=29**
sto = storage pond; in = inflow; S1 = site 1; S2 = site 2; S3 = site 3; S4 = site 4; out = outflow

The ANOVA and Tukey's test of the chemical parameters showed significant differences between the vegetated and non-vegetated sites. The mean of total carbon (TC) concentration showed a significant higher level in concentrations between the vegetated and non-vegetated sites (P<0.001). The mean of total nitrogen (TN) concentration showed similar patterns like TC, but the significance was lower (P<0.01 and P<0.05) (Table 5-17, 5-18; Figure 5-36).

The mean of total phosphorus (TP) concentration showed a totally different pattern in significance between the different sites. In majority the difference between the different collection sites showed no significant differences. The only significant difference was between inflow and site 1, site 2 and site 4. In all three sites the level of TP were significantly higher than the inflow (P<0.01 and P<0.05) (Table 5-17, 5-18; Figure 5-36).

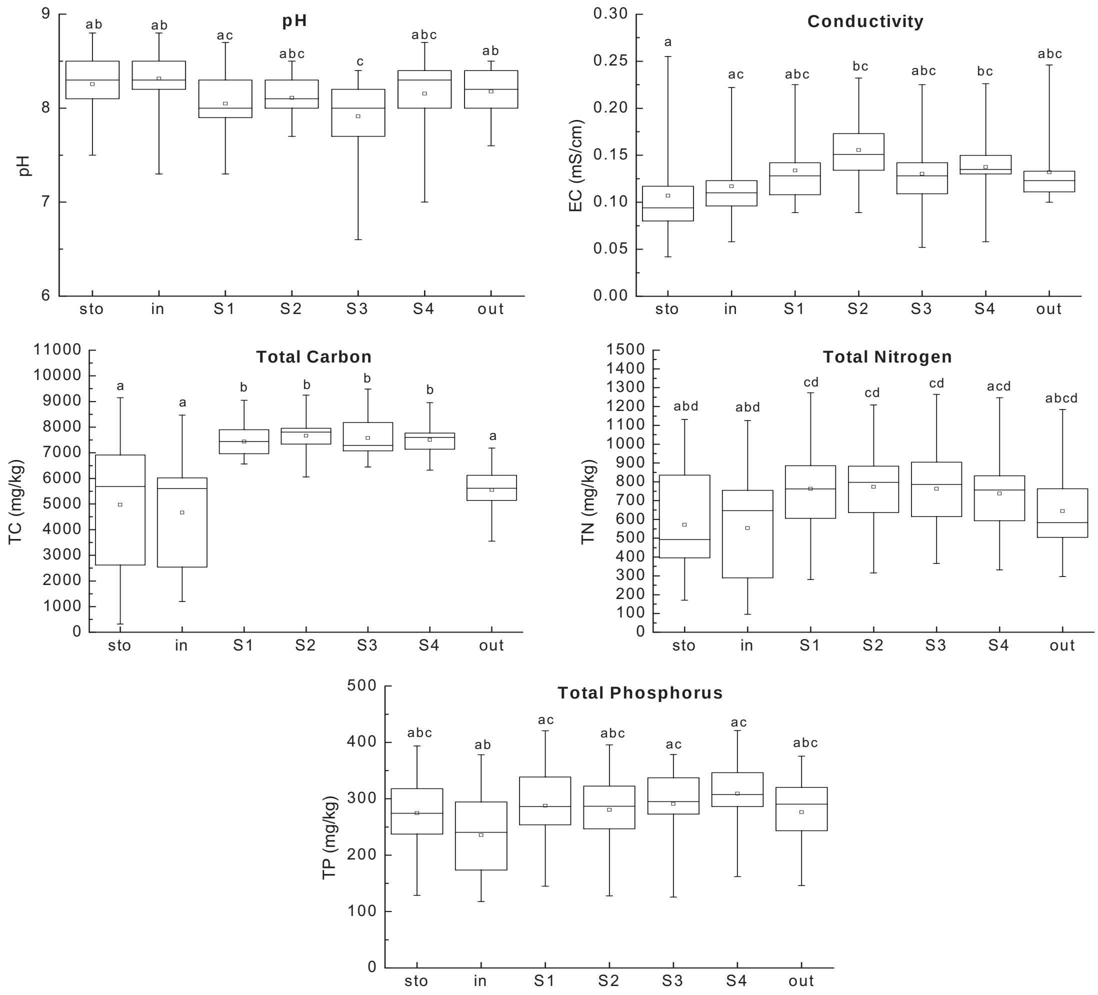


Figure 5-36: Bar and whisker graphs of physical and chemical sediment parameters for the different collection sites. The columns marked with different letters indicate significant difference according to Tukey's test (α = 0.05, n= 29). sto = storage, in = inflow, S1 = site 1, S2 = site 2, S3 = site 3, S4 = site 4, out = outflow

5.4.2 Seasonality

The seasonal patterns of the physical and chemical parameters showed overall a similar trend for all sites (Figure 5-38a, b, c, d).

The physical parameters measured in form of pH and conductivity showed in most the sites a similar seasonal pattern. The pH in most of sites had a distinguished differences between the non-growing (autumn and winter) and growing seasons (spring and summer). The pH levels were higher during the non-growing seasons than the growing seasons. In case of the storage pond the soil pH showed more or less a constant levels during all seasons (Figure 5-38).

Soil conductivity showed a different seasonal pattern than the pH, with having higher levels during the winter and spring seasons in comparison to the lower levels in summer and autumn.

The seasonal pattern of the chemical parameters measured in form of total carbon, total nitrogen and total phosphorus showed for all sites the same variation. The concentrations of different nutrient forms showed a similar pattern like the physical parameters, in which the concentration levels for all three nutrient forms were higher during the non-growing season than the growing seasons.

The general function of the sediment in the reed bed pond of the Parafield Stormwater Harvesting Facility is depending on the changes of the seasons. The results for the different nutrient concentration showed that the nutrient dynamics were changing over the different seasons. In general the nutrient concentrations were lower during the growing seasons (spring and summer season) and were significantly higher during the non-growing seasons (autumn and winter) (Figure 5-37).

For TC, TN and TP of the sediments the concentrations were the highest during the autumn season followed by winter, and spring and autumn. This means that the concentrations during spring and summer were lower, which indicates that during the spring and summer seasons the sediment is acting as source pool of nutrients and as sink during the autumn and winter seasons. During the growing seasons nutrients in the sediment pool will be utilized by the uptake of emergent macrophytes and released to the water column.

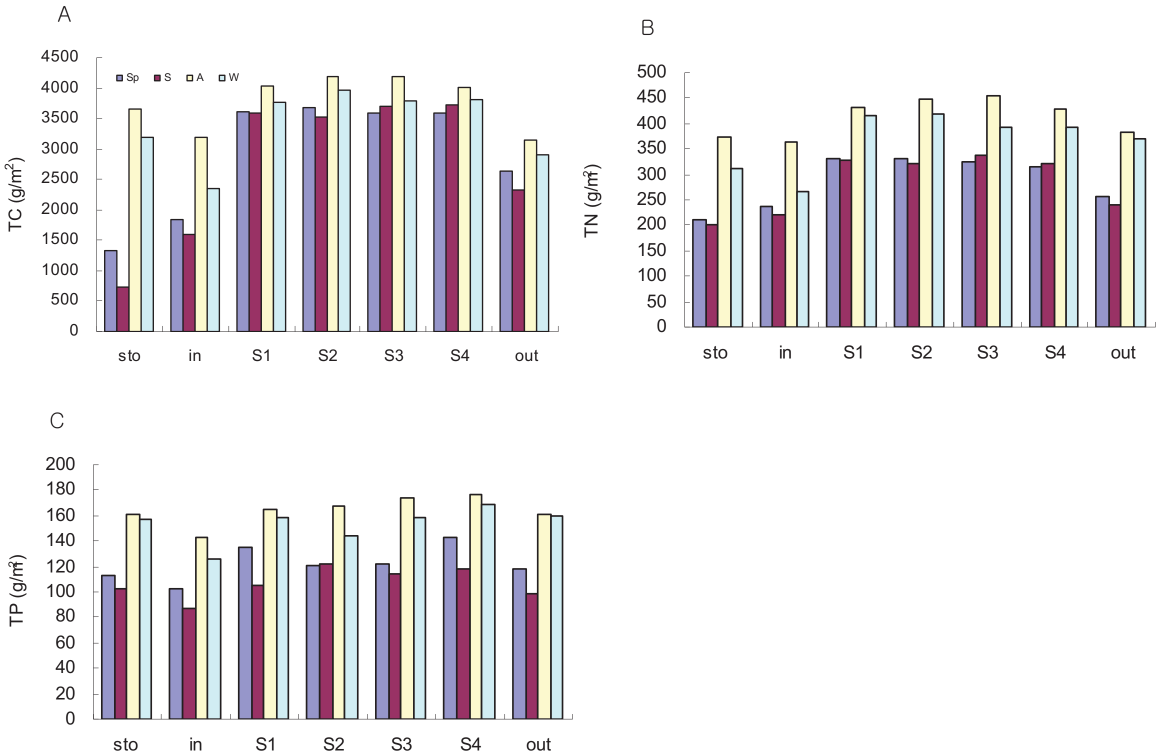


Figure 5-37: Seasonal nutrient concentrations (in g/m^2). A. Total Carbon (TC), B. Total Nitrogen (TN) and C. Total Phosphorus (TP)

5.4.3 Annual pattern

The annual comparison of the measured sediment parameters revealed, that the physical parameter in form of pH and conductivity showed significant differences between the years (Figure 5-39 A&B). In case of the sediment pH, the level in 2007 was significantly lower than the previous two years (Figure 5-39 A). The conductivity levels showed a significant decline during the whole study period (Figure 5-39 B). The level in 2007 was significantly lower than 2006, and the conductivity in 2006 was significantly lower than 2005.

For the chemical parameter the annual comparison showed a different pattern. The chemical parameters measured were total carbon, total nitrogen and total phosphorus. In case of total carbon and total phosphorus the ANOVA test revealed, that no statistical significance existed in the levels of concentration between the years (Figure 5-39 C&E). The concentration levels of total nitrogen showed a significant difference between the years. The level of TN in 2006 was significantly higher than 2005 and 2007. No significant difference was detected between the TN concentration of 2005 and 2007, however looking at the magnitude the level of TN in 2005 seem to be bigger (Figure 5-39 D).

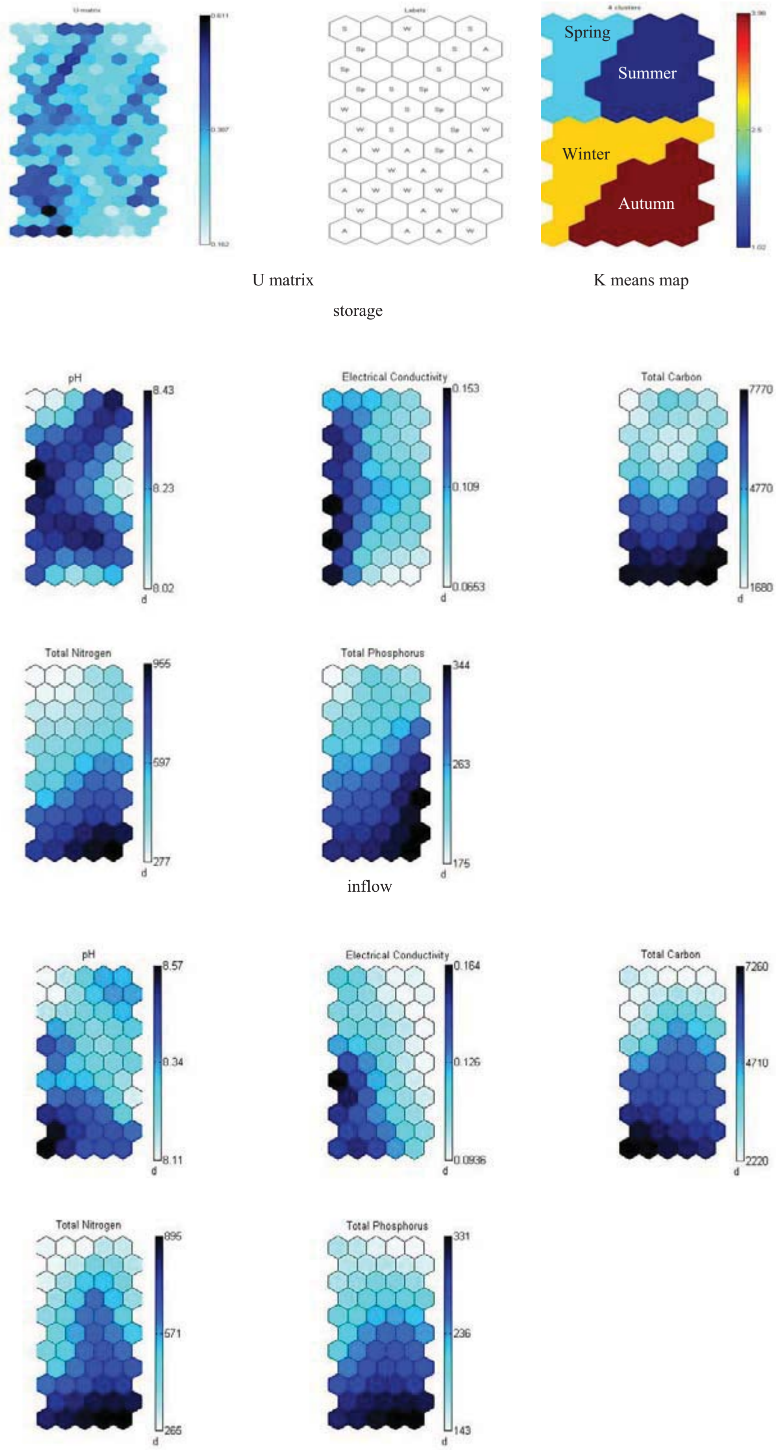


Figure 5-38: Seasonal patterns visualized as a U matrix, K means map and SOM maps for measured sediment parameters for the at storage and inflow area

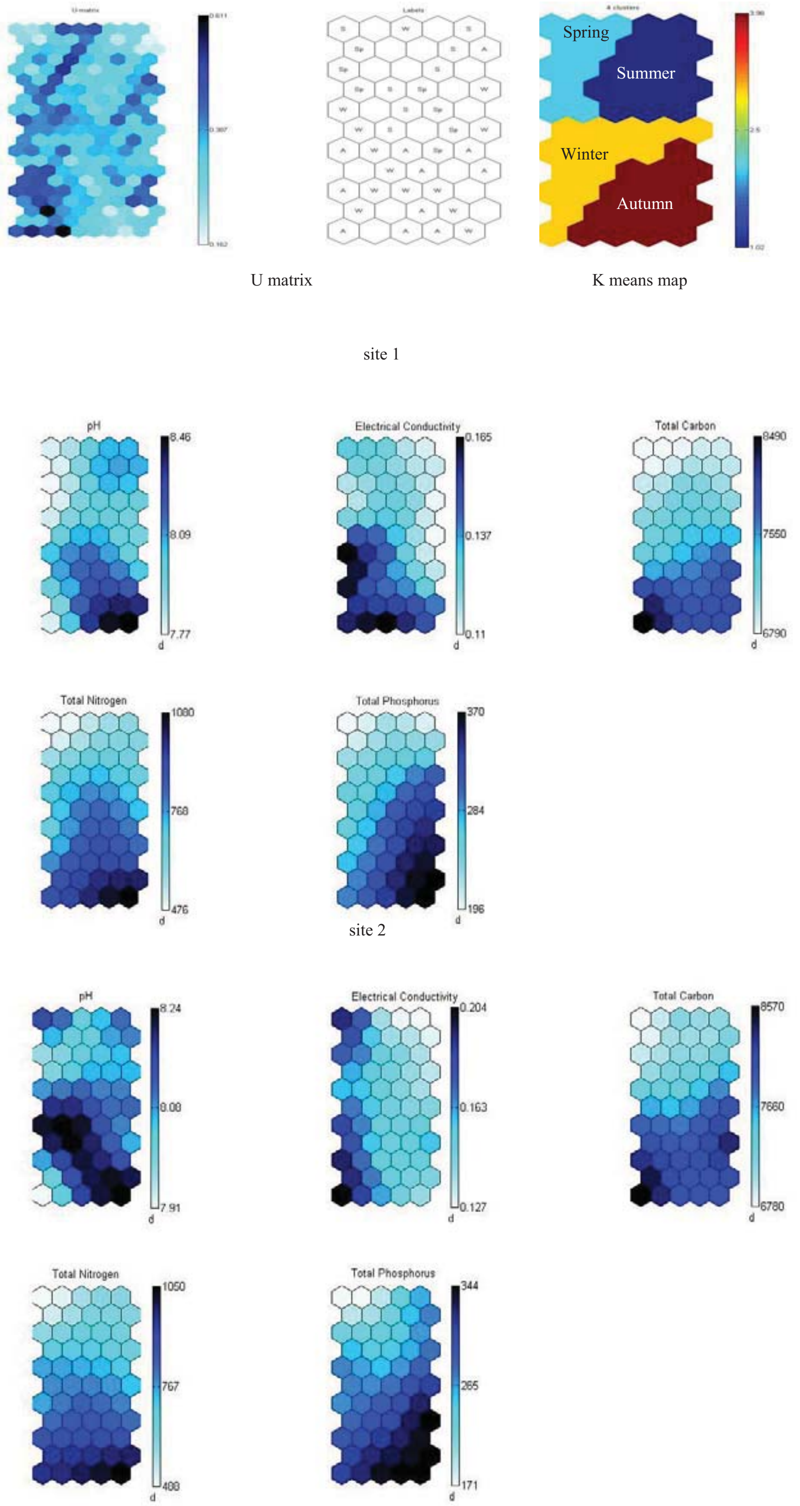


Figure 5-38 a: Seasonal patterns visualized as a U matrix, K means map and SOM maps for measured sediment parameters for the at site 1 and site 2

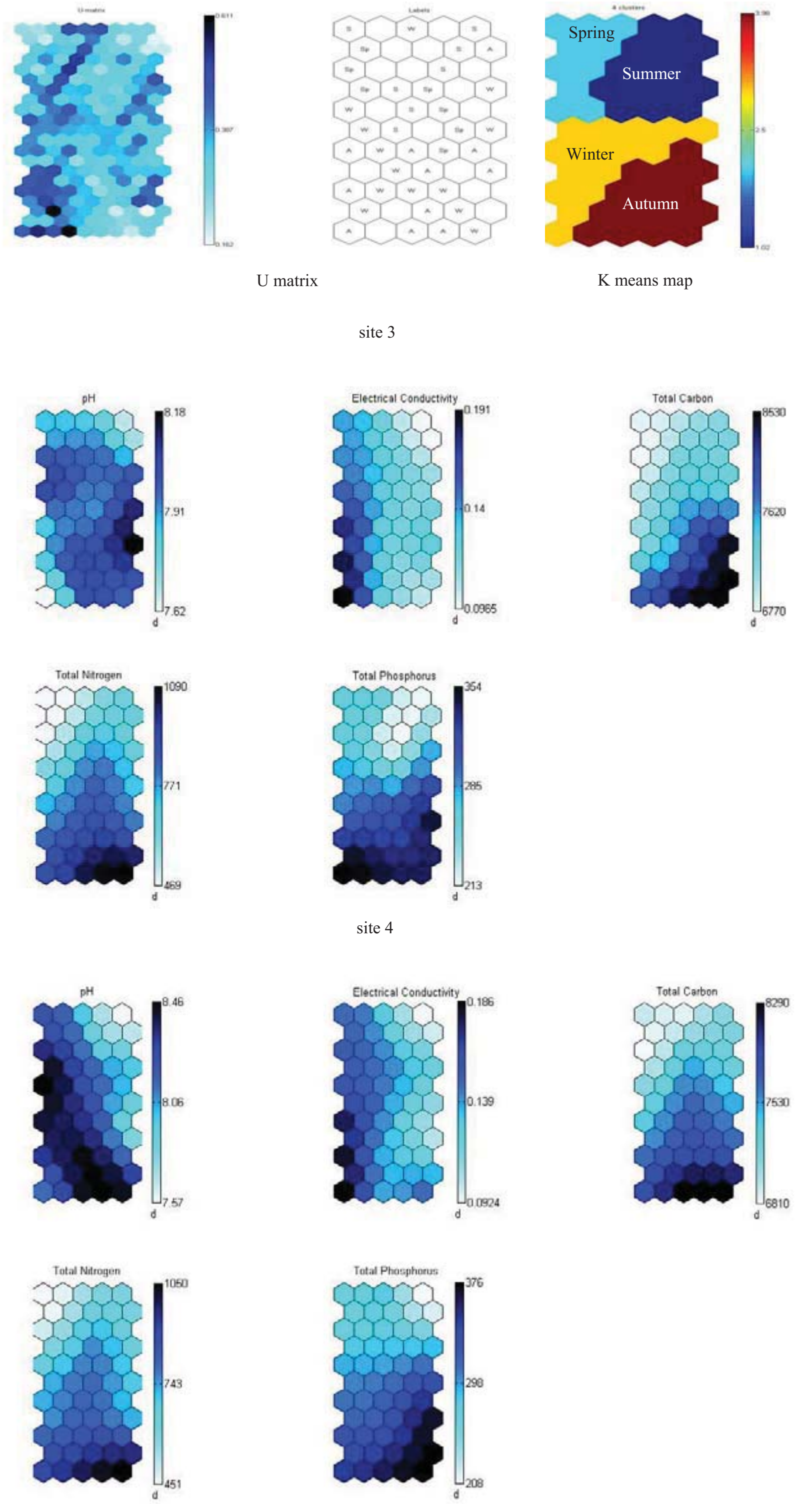


Figure 5-38 b: Seasonal patterns visualized as a U matrix, K means map and SOM maps for measured sediment parameters for the at site 3 and site 4

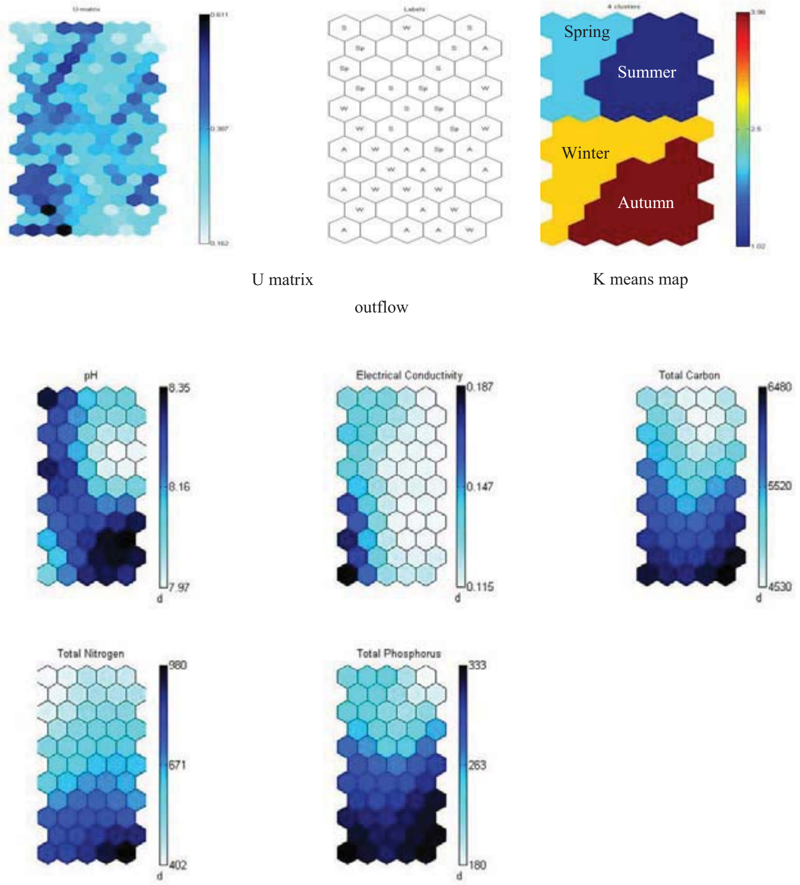


Figure 5-38 c: Seasonal patterns visualized as a U matrix, K means map and SOM maps for measured sediment parameters for the outflow area

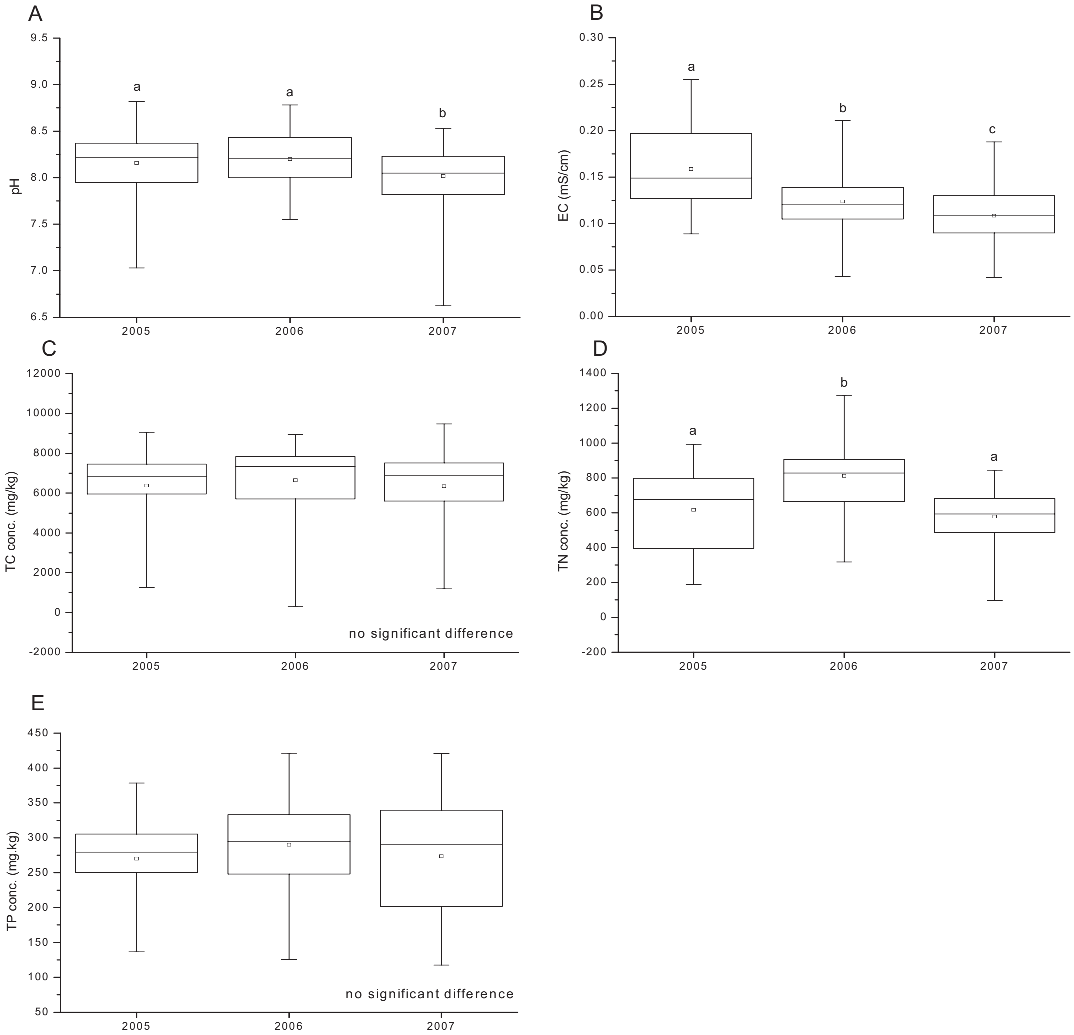


Figure 5-39: Annual comparison of sediment parameters. A. pH, B. EC, C. TC, D. TN and E. TP. The columns marked with different letters indicate significant difference according to Tukey's test ($\alpha = 0.05$; 2005 n= 63, 2006 n = 84 and 2007 n = 56)

5.4.4 Sediment Nutrient Budget

The average (Table 5-19) and monthly (Figure 5-40, 5-41 and 5-42) sediment nutrient budgets for the different collection sites. The budgets were determined by using the following equation: $B_N = A \times C_N$ (5-7)

where B_N is the nutrient budget, A is area (m^2) and C_N is the concentrations/area (kg/m^2).

The sediment budgets calculated for TN, TP and TC showed a similar pattern like the nutrient budgets for water. The nutrient budget levels were significantly higher in the storage pond in comparison to the other collection sites (Table 5-19). On the other hand the nutrient budget levels in the inflow and outflow areas were significantly lower than the budgets in the storage pond and the reed bed sites (Table 5-19). The nutrient budgets within the reed bed sites showed for all three nutrient parameters no significant difference (P>0.05).

Table 5-19: Average Nutrient Budgets (t) for the different sites

	TN	TP	TC
storage	6.00[a]	2.88[a]	52.20[a]
inflow	0.15[b]	0.06[b]	1.23[b]
site 1	1.79[c]	0.67[c]	17.43[c]
site 2	1.81[c]	0.66[c]	17.96[c]
site 3	1.79[c]	0.68[c]	17.75[c]
site 4	1.73[c]	0.72[c]	17.58[c]
outflow	0.12[b]	0.05[b]	1.00[b]

Average (n=29) in the columns followed by different letters indicate significant difference (P<0.05)

The monthly average nutrient budgets revealed that the budgets were higher for all three measured nutrient forms during the non-growing seasons (autumn and winter) compared to the growing seasons (spring and summer). The seasonal trends were the same for all sites with differences in the magnitudes (Figure 5-40, 5-41 and 5-42). The total budget regards TN for whole study period for the different sites were 174.1t for the storage pond, 4.2t for the inflow area, 51.8t for site 1, 52.5 for site 2, 51.8t for site 3, 50.2t for site 4 and 3.4t for the outflow area. The total budget regards TP for the different sites were 83.6t for the storage pond, 1.8t for the inflow area, 19.6t for site 1, 19.1t for site 2, 19.8t for site 3, 21.0t for site 4 and 1.4t for the outflow area. The total sediment budgets for carbon over the whole study period at the different collection sites were 1513.7t for the storage pond, 35.6t for the inflow area, 505.t for site 1, 521.0t for site 2, 514.9t for site 3, 509.9t for site 4 and 28.9t for the outflow area.

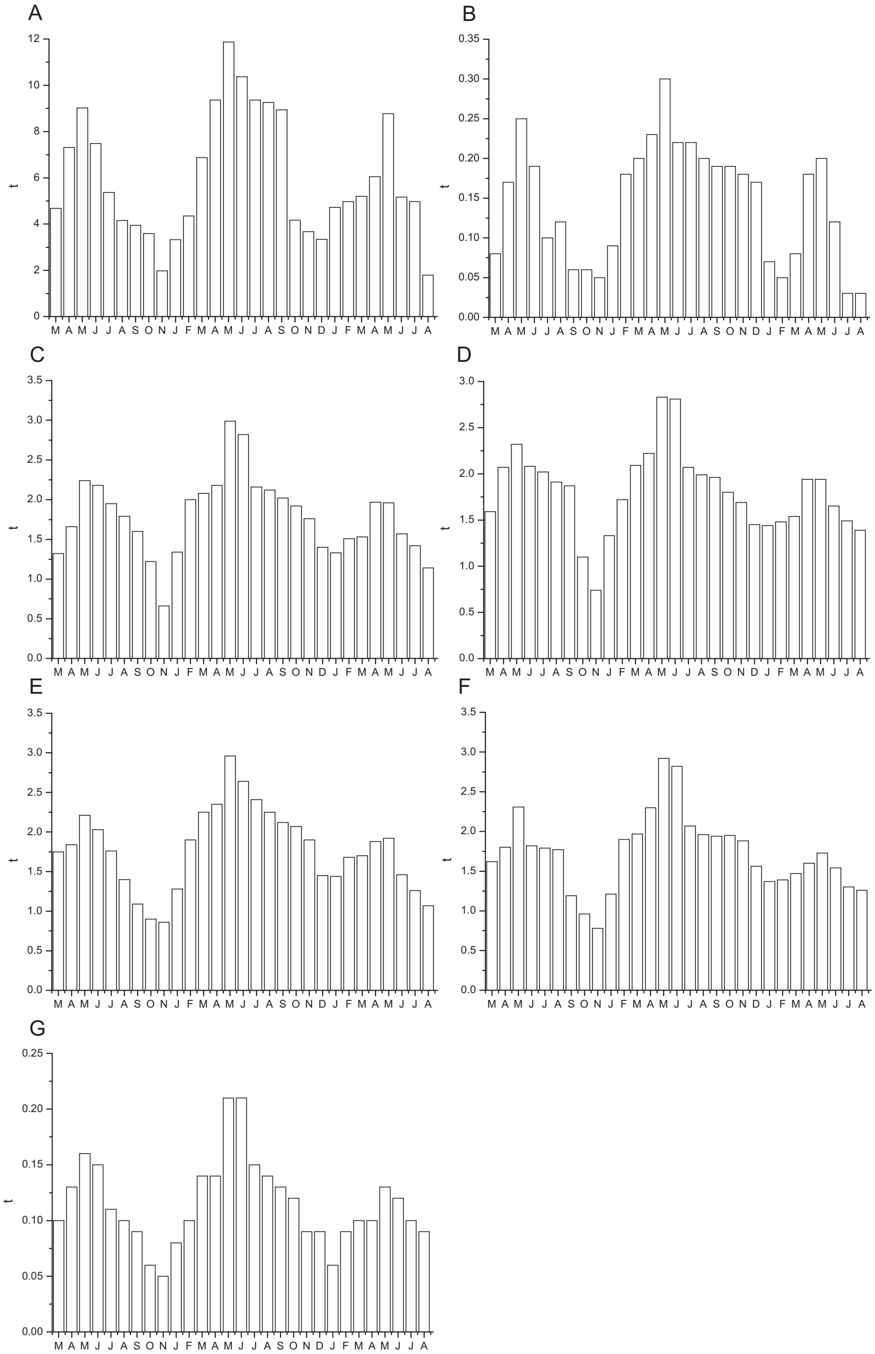


Figure 5-40: Monthly Total Nitrogen (TN) budgets for the different sites in t. A. storage, B. inflow, C. site 1, D. site 2, E. site 3, F. site 4 and G. outflow.

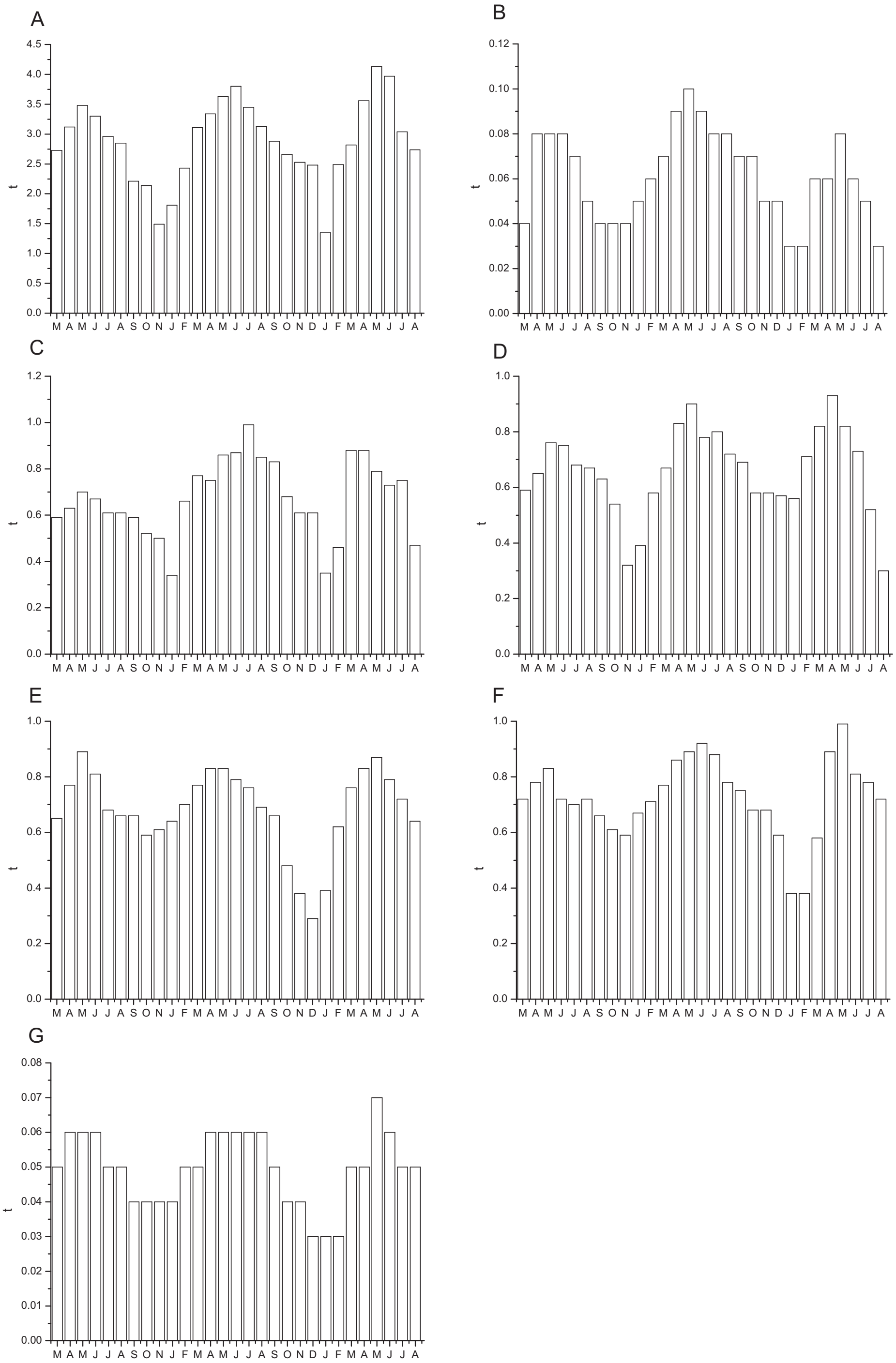


Figure 5-41: Monthly Total Phosphorus (TP) budgets for the different sites in t. A. storage, B. inflow, C. site 1, D. site 2, E. site 3, F. site 4 and G. outflow.

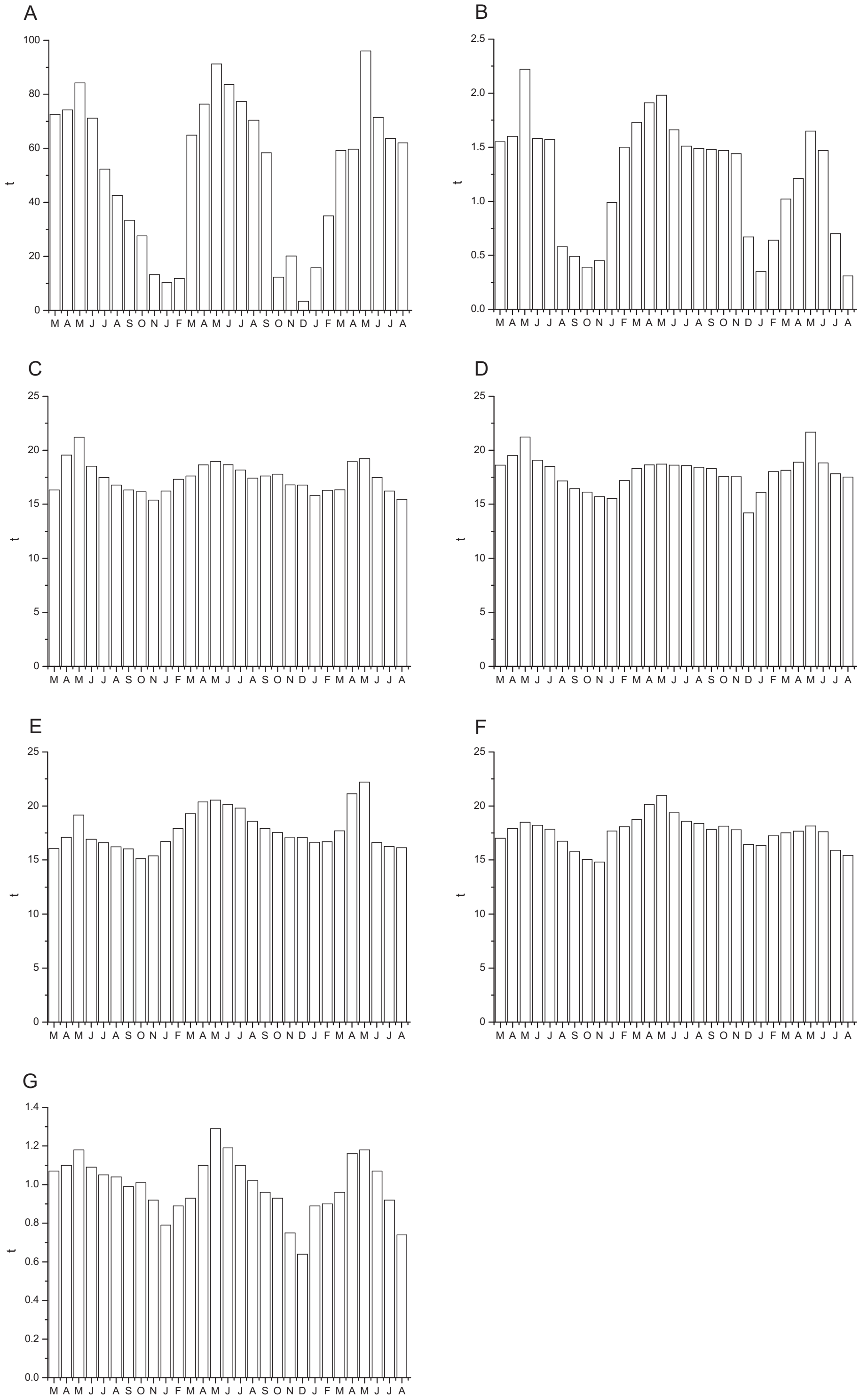


Figure 5-42: Monthly Total Carbon (TC) budgets for the different sites in t. A. storage, B. inflow, C. site 1, D. site 2, E. site 3, F. site 4 and G. outflow.

5.5 Hybrid Evolutionary Algorithm (HEA)

The HEA method was used to determine key parameters to determine or develop rules for the different forms of the nutrients, which was mentioned in method section. Due to the short nature of the data set, a method called "Bootstrap method" was used to develop the rule sets to predict and calculate the nutrient concentrations.

The HEA produces 100 models for each experiment from which the best 50 will visualized. The next step is to use the following criterion for selecting the "best" model based on the highest R^2 value and the lowest total error value.

5.5.1 HEA models predicting the nutrient concentration at the outflow

The first experiment is to test the ability of the "Bootstrap" method of HEA to predict the different nutrient concentrations at the outflow area by having the variables of the inflow area as input.

5.5.1.1 Total Nitrogen

The TN concentrations at the outflow area have been predicted by using the parameters from the inflow site as input. Two models have been developed using different input variables for the prediction of TN conc. at the outflow. Model 1 used as input variables water temperature, conductivity, dissolved oxygen, pH, redox potential, turbidity, TN conc. of inflow and flow rate at inflow (Figure 5-43 A). Model 2 has the same input variables with the exception of TN conc. of inflow (Figure 5-43 B).

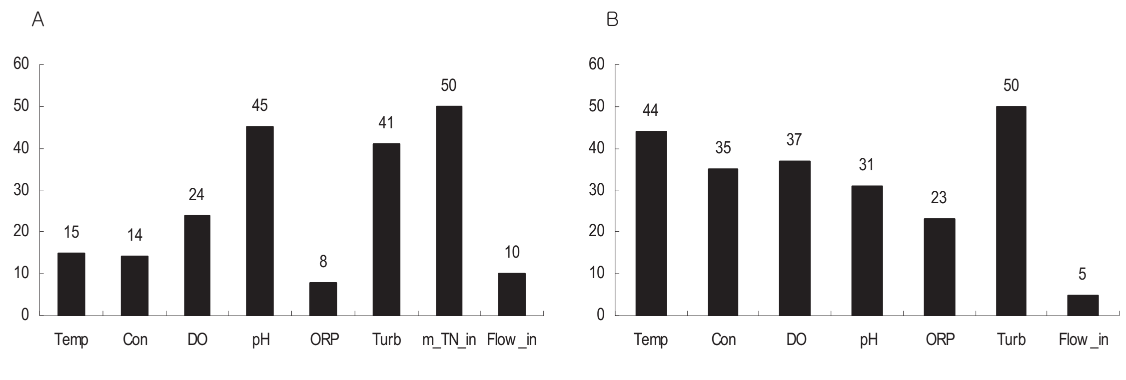


Figure 5-43: Input variables used for the prediction of TN conc. at the outflow area. A. HEA model including TN conc. of inflow (model 1) and B. HEA model without TN conc. of inflow (model 2)

The figure 5-43 is showing the appearance of input variables within the rule set equations developed by the HEA for the prediction of the output. In case of model 1 the parameters, which were linked having an influence on the TN concentration (outflow), were pH, turbidity and TN (inflow). For the model 2 the variable selected mainly in the rule set equations were turbidity, water temperature, DO, conductivity and pH.

The prediction of the TN conc. at the outflow area using the variables from the inflow area as an input showed different prediction accuracies for model 1 (R^2 = 0.77 & total error = 0.1315) and for model 2 (R^2 = 0.41 & total error = 0.2105) (Figure 5-44 A & B). Model 1 was more accurate in predicting the most of peak events in regards timing and magnitude. It mostly underestimated the TN conc. Model 2 was less accurate in predicting the major peak events. The rule set missed the peak events of the period June 2005-September 2005 completely, but was able to predict the following peak events, but the timing and magnitude were less accurate than compared to model 1 (Figure 5-45 A & B).

A

IF (Con<=0.342)

THEN =((((Turb+pH)+pH)+pH)/(((Turb+10.259)/TNin)+(174.195/pH)))

ELSE =(((Turb+Temp)+9.900)/(((Temp+11.980)/Con)+(Turb+pH)))

Total error=0.1315 R^2 value=0.77

B

IF ((14.908/Con)<63.104)

THEN =((Turb+(Temp+(ORP/Temp)))/64.728)

ELSE =(((Con*(-111.682))-225.071)/(-304.138))

Total error=0.2105 R^2 value=0.41

Figure 5-44: Rule sets developed by HEA. A. model 1 and B. model 2

The sensitivity of model 1 for the THEN and ELSE branch (Figure 5-45 C & E) showed, that for the THEN branch all selected key variables had a positive effect on the output variable. In the THEN branch TN conc. of inflow was the variable, which seemed to have greatest impact on the TN concentration compared to the variables turbidity and pH. The ELSE branch is showing the same trends in regards that all chosen parameters had a positive impact on the output parameter. In the ELSE branch the parameter with the greatest influence on output parameter was conductivity. Water temperature and pH showed only small influence on the TN concentration at the outflow site.

For the model 2 the sensitivity analysis for the THEN and ELSE branch (Figure 5-45 D& F) showed the same like for model 1, which is that the selected variables have a positive effect on the output parameter. For the THEN branch turbidity had the greatest positive effect on the output parameter, whereas ORP and water temperature had a smaller impact on the TN conc. For the ELSE branch conductivity was only variable selected as key parameter to predict the output.

The prediction for other nitrogen components in form of nitrate and ammonium can be seen in the Appendix E.

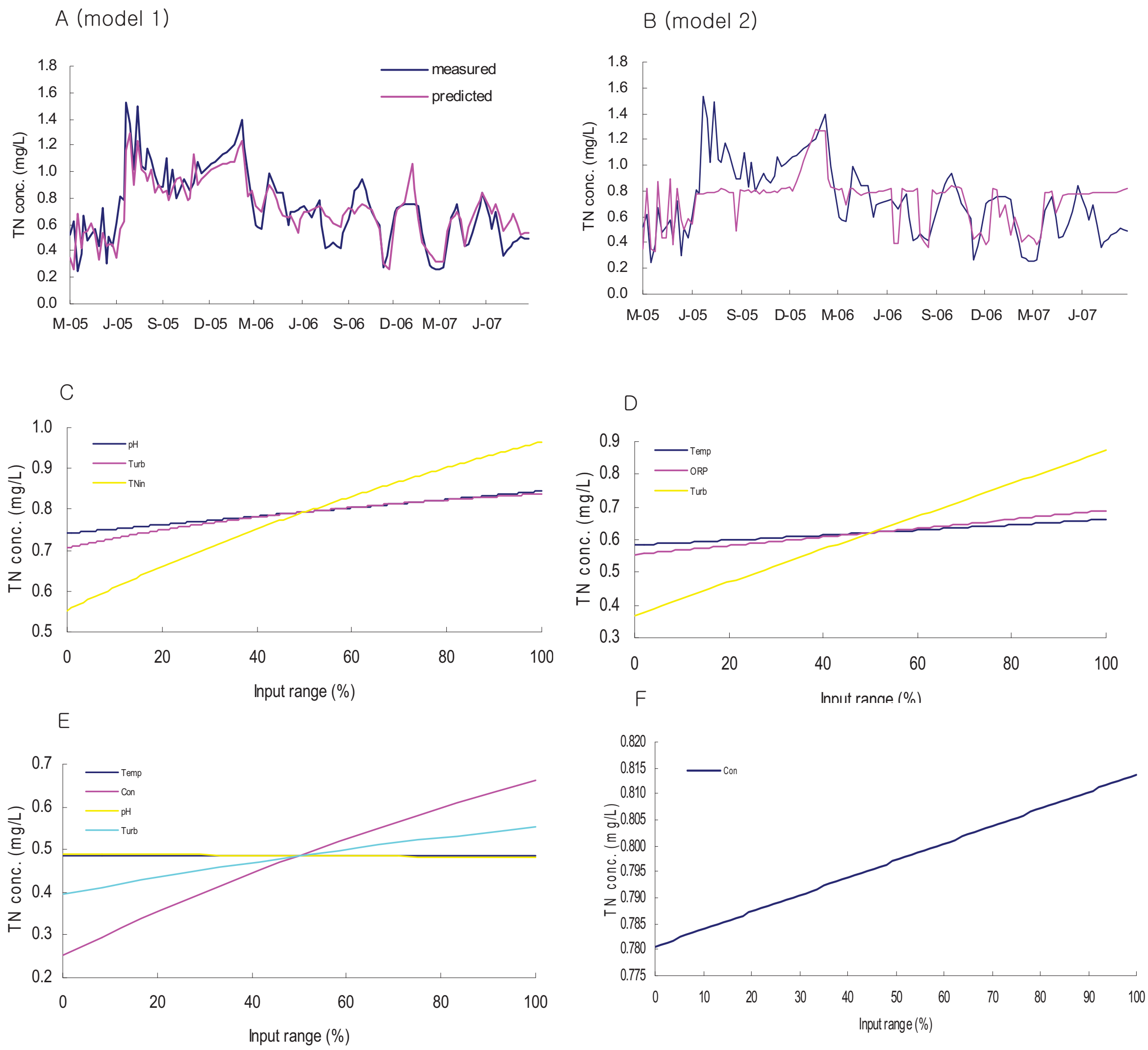


Figure 5-45: HEA results for TN at outflow site. A (model 1) and B (model 2) Comparison between the measured and predicted TN concentrations; C. & D. Sensitivity for THEN branch and E. & F. Sensitivity for ELSE branch

5.5.1.2 Total Phosphorus

The input parameters selected for the prediction of TP concentration at the outflow area, were selected in the same principle like that for TN, which is mentioned in the previous section. Model 1 used all measured physical parameters in addition to the TP concentration of the inflow and the flow rate. Model 2 had the same input parameters with the exception of the nutrient concentration (Figure 5-46 A & B). For model 1 the input parameters selected for the rule set equations were pH, ORP and TP (inflow) and for model 2 the variables were pH, ORP and turbidity.

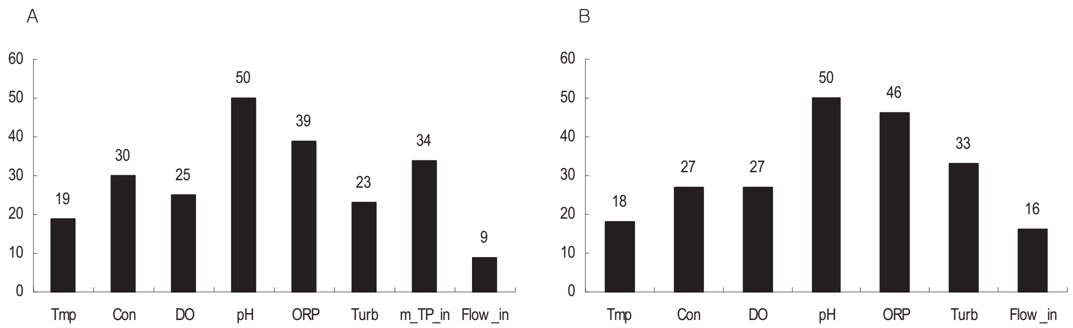


Figure 5-46: Input variables used for the prediction of TP concentration at the outflow area. A. model 1 and B. model 2

The accuracy of TP prediction measured as R^2 showed for model 1 $R^2 = 0.63$ and model 2 $R^2 = 0.44$ (Figure 5-47 A & B). Both models were able to predict the major peak events accurate in timing and magnitude (Figure 5-48 A & B). Model 1 has a more accurate prediction than model 2. Model 2 has failed to predict a peak event in August 2005.

A

IF (((Turb/ORP)*(-164.351))<128.796)

THEN =((66.539/pH)/((122.302/(ORP+(-172.295)))+389.218))

ELSE =(TPin-(TPin/((ORP*Temp)/192.628)))

Total error=0.0070 R^2 value=0.63

B

IF ((((ORP*Flow_in)*322.979)*60.004)<=57.784)

THEN =((92.787/pH)/(pH*(pH*DO)))

ELSE =(45.419/(pH*267.732))

Total error=0.0085 R^2 value=0.44

Figure 5-47: Rule sets developed by HEA. A. model 1 and B. model 2

The sensitivity analysis of model 1 showed that for the THEN branch the parameters pH and ORP were selected, which both showed a negative impact on the output variable. The impact of the parameter pH had the greatest influence on the TP concentration at the outflow area. Lower ORP levels would indicate a reduced state in the sediment, by which iron bounded phosphorus, will be released to the environment (Figure 5-48 C). For the ELSE branch the following variables, in form of water temperature, ORP and TP conc. at the inflow area, were selected to determine the output variable (Figure 5-48 E).

The sensitivity analysis of model 2 showed that for the THEN branch the parameters DO and pH were selected as key variables, which had both a negative impact on the TP concentration (Figure 5-48 D). The parameter selected for the ELSE branch was pH (Figure 5-48 F).

The prediction models for phosphate can be seen in Appendix E.

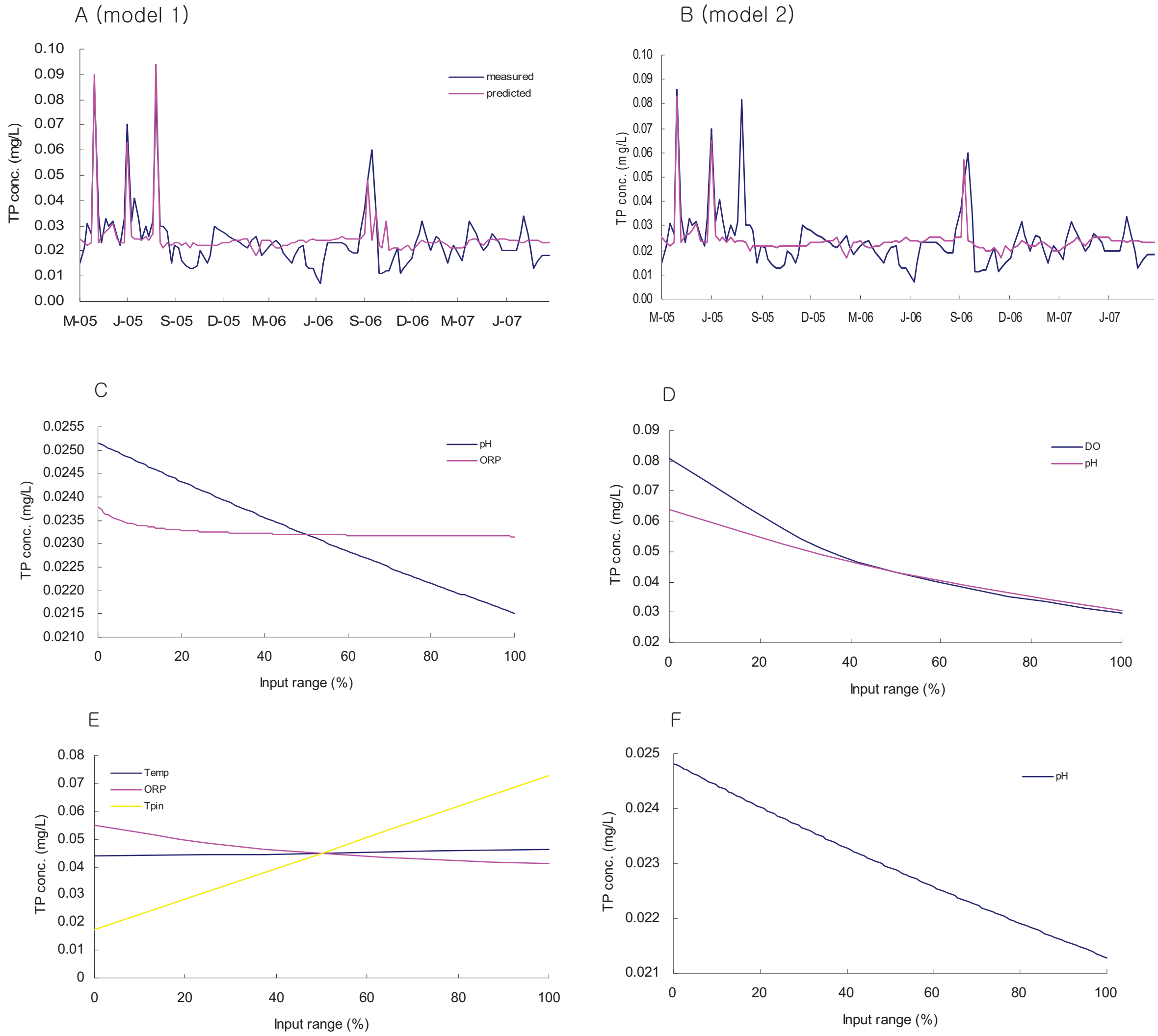


Figure 5-48: HEA results for TP at outflow site. A (model 1) and B (model 2). Comparison between the measured and predicted TP concentrations; C. & D. Sensitivity for THEN branch and E. & F. Sensitivity for ELSE branch

5.5.1.3 Dissolved Organic Carbon

The input parameters for the prediction of DOC at the outflow area were the same as for the other nutrients. Model 1 included all measured physical variables in addition to flow and nutrient data from the inflow; and in case of model 2 all the input parameters were the same like model 1 with the exception of the nutrient data (Figure 5-49 A & B). The input variables selected most for the development of rule set equations for model 1 were water temperature, conductivity and DOC conc. (inflow). For model 2, water temperature, conductivity, pH and ORP were the parameters most selected by the HEA (Figure 5-49).

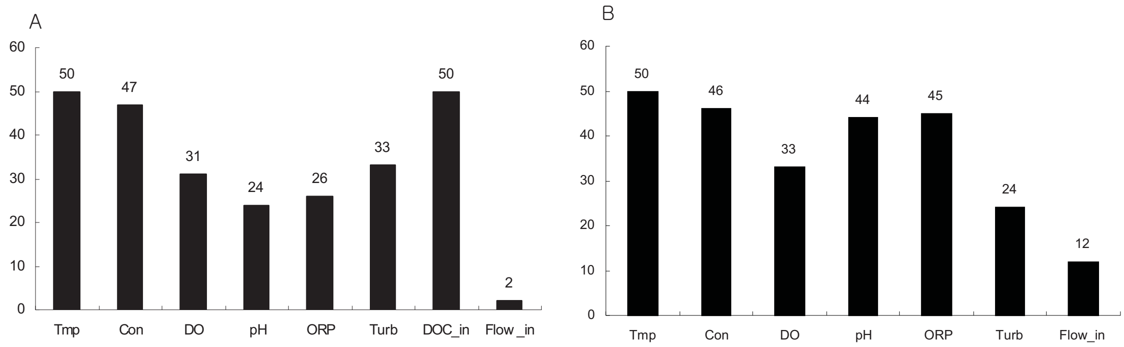


Figure 5-49: Input variables used for the prediction of DOC concentration at the outflow area. A. model 1 and B. model 2

A

IF ((DOC_in<=5.313)OR(DOC_in>15.409))

THEN =((pH+Temp)-(Con*(50.217-(ORP/7.108))))

ELSE =(DOC_in+(((ORP*Turb)/97.559)/70.087))

Total error=2.5706 R^2 value=0.66

B

IF (ORP>=82.971)

THEN =((pH*((Temp*pH)+(DO*pH)))/98.444)

ELSE =(((Temp+(pH/Con))*(Temp+(ORP/32.509)))/76.537)

Total error=3.0692 R^2 value=0.53

Figure 5-50: Rule sets developed by HEA. A. model 1 and B. model 2

The prediction accuracy of the two models were slightly different, with model 1 having a greater accuracy measured in the value of R^2 = 0.66 and total error = 2.5706 than model 2, which had a prediction accuracy of a R^2 = 0.53 and total error = 3.0692 (Figure 5-50). Both models were able to predict major peak events as well as the lower concentration events with sometimes over- and under-prediction (Figure 5-51 A & B). The model 2 failed to predict two peak events around September 2005 (Figure 5-51 B)

The sensitivity analysis of the THEN branch for the model 1 has chosen water temperature, conductivity, pH and ORP as the key parameters having an impact on the prediction of DOC at the outflow (Figure 5-51 C). With the exception of conductivity all the other parameters had a positive impact on the output parameter. Conductivity had a negative impact on the output, which means with the increase of conductivity the output concentration will decrease, which is due to the loading of aquifer water during the summer seasons, where higher conductivity levels were observed, which causes a dilution effect. For the ELSE branch ORP, turbidity and DOC (inflow) were selected as key variables (Figure 5-51 E). The selected parameters had a positive effect on the output parameter; with DOC (inflow) having the greatest influence on the output.

The THEN for model 2 selected water temperature, DO and pH as key variable for the prediction of DOC conc. at the outflow (Figure 5-51 D). All the selected parameters had a positive influence on the output.

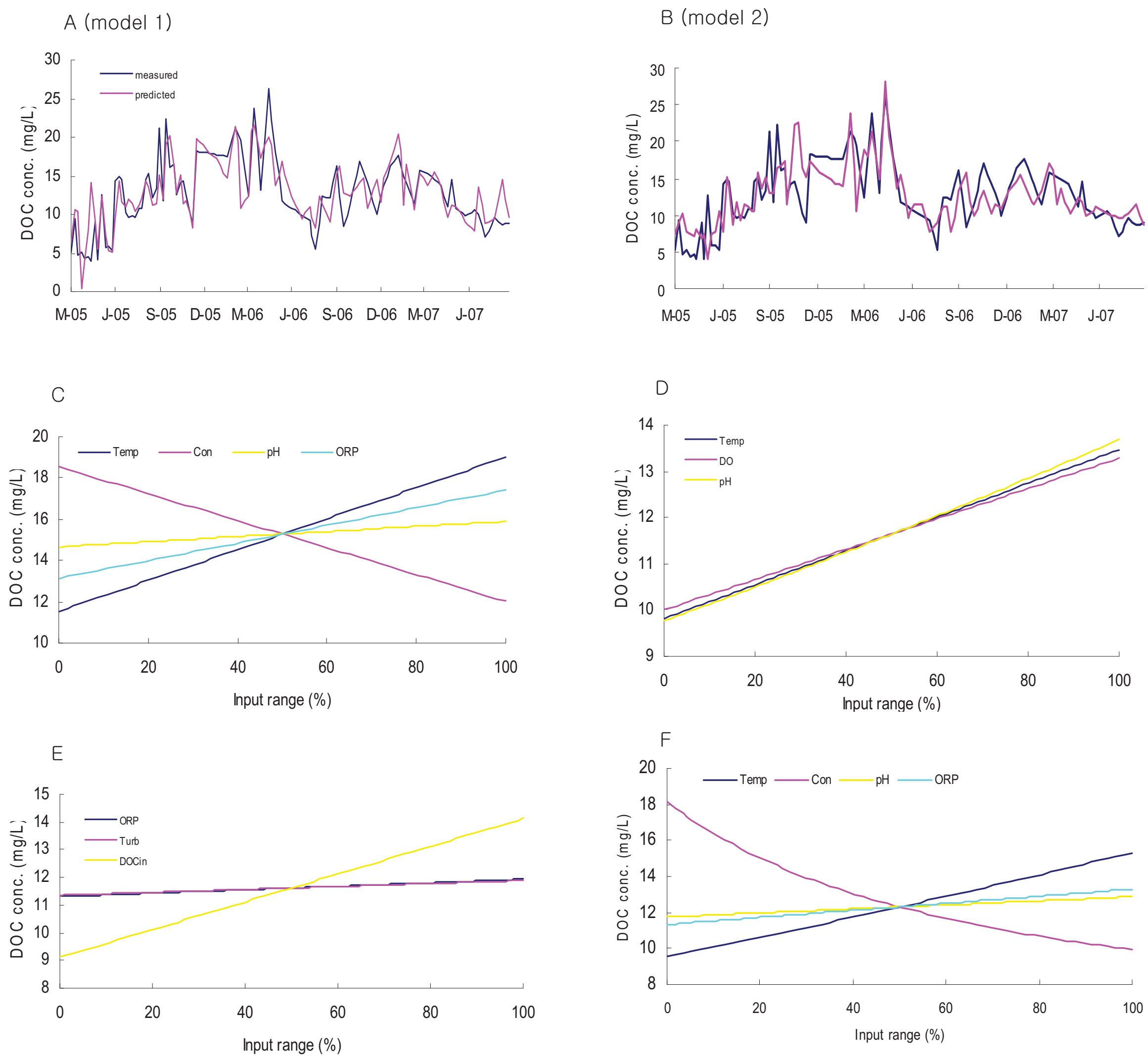


Figure 5-51: HEA results for DOC at outflow site. A (model 1) and B (model 2). Comparison between the measured and predicted DOC concentrations; C. & D. Sensitivity for THEN branch and E. & F. Sensitivity for ELSE branch

The ELSE branch selected the parameters water temperature, conductivity, pH and ORP as the key variables determining the output parameter (Figure 5-51 F). Water temperature, pH and ORP had a positive impact on the output variable; whereas conductivity indicated to have a negative influence on the output parameter.

5.5.2 7 days ahead forecasting of the nutrient concentration at the outflow using HEA

Based on prediction of different forms of nutrients at the outflow using the input parameters from the inflow; a forecasting model has been developed for the different nutrient forms. The forecasting period was set to 7 days ahead forecasting for all the different nutrient forms.

5.5.2.1 Total Nitrogen

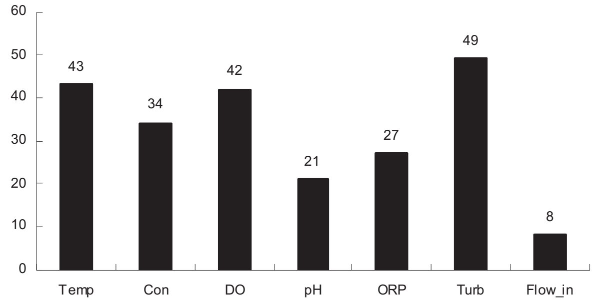


Figure 5-52: Input variables used for the forecasting of TN

The input variables selected for the development of the forecasting model for TN were the same as for the model 2 predicting the TN at the outflow in the previous section (see section 5.5.1.1. Figure 5-43 B). The inputs, which were selected most in the rule set equations, were water temperature, conductivity, DO and turbidity (Figure 5-52).

The prediction results of the developed model had an accuracy of an R^2 = 0.39 and a total error = 0.2132. The model was able to forecast most the of the major peak events. For high peak events (TN conc. > 1.0 mg/L) the model was able to match the timing of the peak events, but wasn't able to match the magnitudes and mostly underestimated the TN concentration at the outflow (Figure 5-53 A).

The sensitivity analysis showed for the THEN branch that water temperature, conductivity and turbidity were selected as the key variables having an influence on TN. The variables water temperature and turbidity have a positive influence on the TN conc., which means that an increase in these parameters is indicated with TN concentrations in the water column. Conductivity has a negative influence on TN, which means that an increase in conductivity will related with lower concentrations (Figure 5-53 B). For the ELSE branch the sensitivity analysis has chosen conductivity, ORP and turbidity has key variables. The variables conductivity and ORP had both a negative effect on the output parameter TN, whereas turbidity had a positive effect on the TN concentration at the outflow (Figure 5-53 C).

The forecasting models for ammonium and nitrate are in the Appendix E.

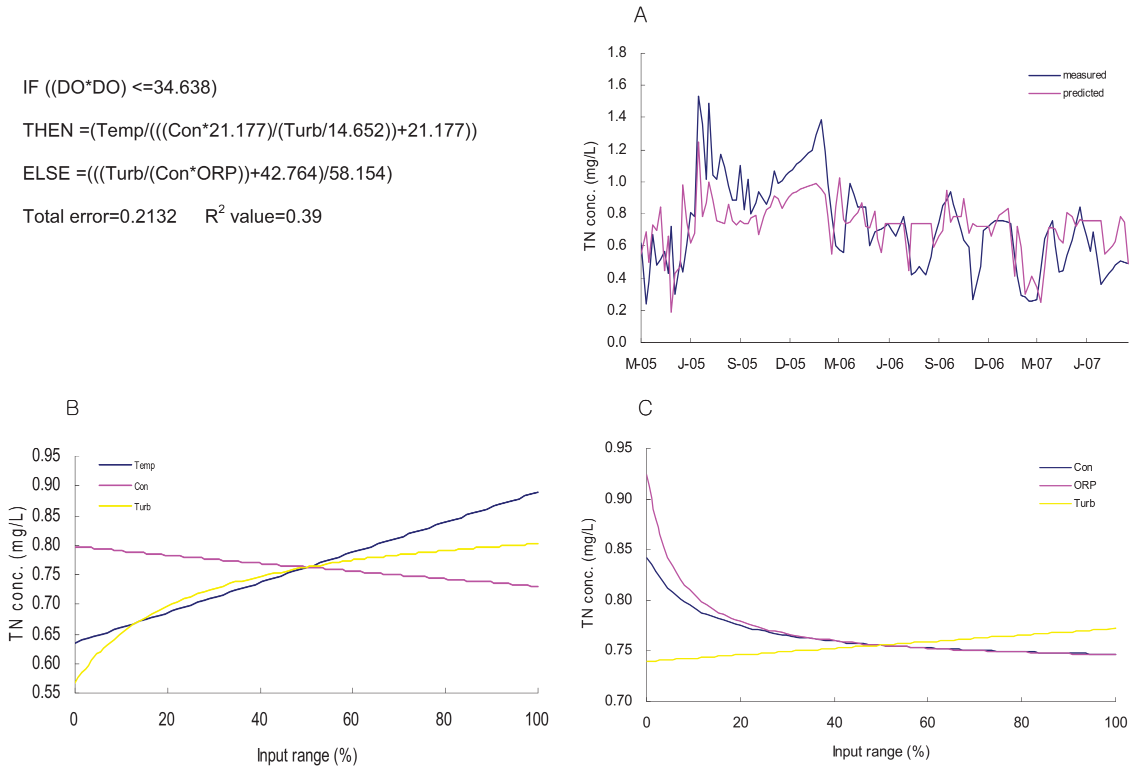


Figure 5-53: 7 days ahead forecasting results for TN at the outflow site. A. Comparison between the measured and predicted TN concentrations; B. Sensitivity for THEN branch and C. Sensitivity for ELSE branch

5.5.2.2 Total Phosphorus

The input variables for the development of the forecasting model for TP were selected like mentioned in the previous section. The variables which appeared the most the rule set equation were pH and ORP (Figure 5-54).

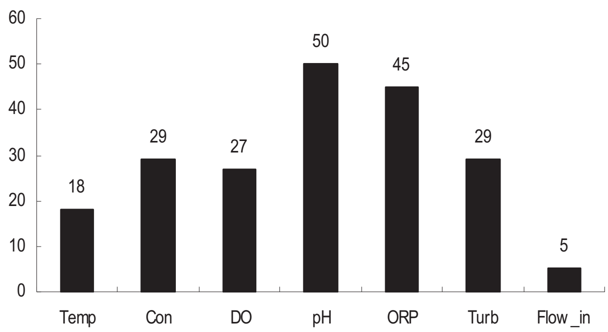


Figure 5-54: Input variables used for the forecasting of TP

The prediction of the forecasting model for TP had an accuracy of a $R^2 = 0.32$ and total error = 0.0094. The forecasting model was able to forecast the lower TP concentrations, ranging between 0.00 - 0.04 mg/L, but wasn't being accurate enough to forecast higher TP concentrations. Most of the major peak events were missed or underestimated. Only one major peak event has been well forecasted in regards timing and magnitude (Figure 5-55 A).

The sensitivity analysis showed for the THEN branch, that the parameters DO, pH and ORP were selected to have a significant influence on the output parameter. All selected parameters showed to have a negative impact on the output variable, with ORP covering a wide range of TP concentration (Figure 5-55 B). For the ELSE branch pH and ORP were selected as key parameter and both had a negative influence on the output parameter (Figure 5-55 C).

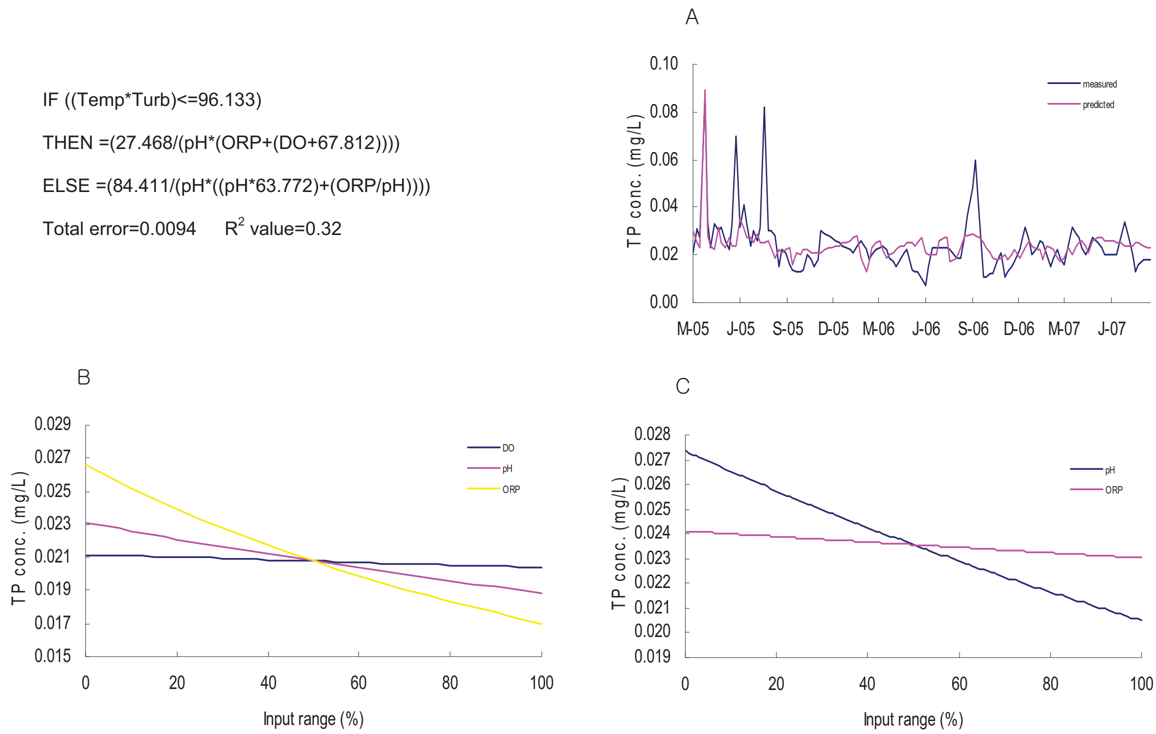


Figure 5-55: 7 days ahead forecasting results for TP at the outflow site. A. Comparison between the measured and predicted TN concentrations; B. Sensitivity for THEN branch and C. Sensitivity for ELSE branch

5.5.2.3 Dissolved Organic Carbon

The input variables, which were selected to development a forecasting model for DOC, were chosen on the same manner as TN and TP (see the previous sections). The inputs, which appeared most in rule set equations were all inputs with the exception of turbidity and flow (Figure 5-56).

The prediction of the forecasting model for DOC had an accuracy of $R^2 = 0.58$ and total error = 2.9164. The forecasting model was able to predict most of the higher peak events, with the exception of two. Middle and lower concentrations were forecasted relatively accurate, sometimes with slight over- and under-estimations (Figure 5-57 A).

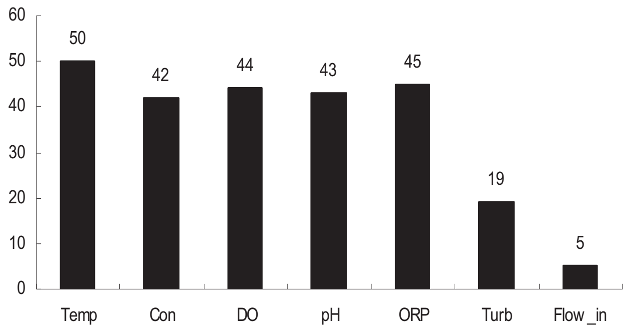


Figure 5-56: Input variables used for the forecasting of DOC

The THEN branch selected the water temperature, conductivity, pH and ORP as key parameter in regards determining the output variable. With the exception of conductivity the rest of the selected variables had a positive influence on the DOC concentration at the outflow (Figure 5-57 B). For the ELSE branch water temperature, conductivity, DO and pH were selected as key variables. All selected parameters had a positive influence on the DOC concentration (Figure 5-57 C).

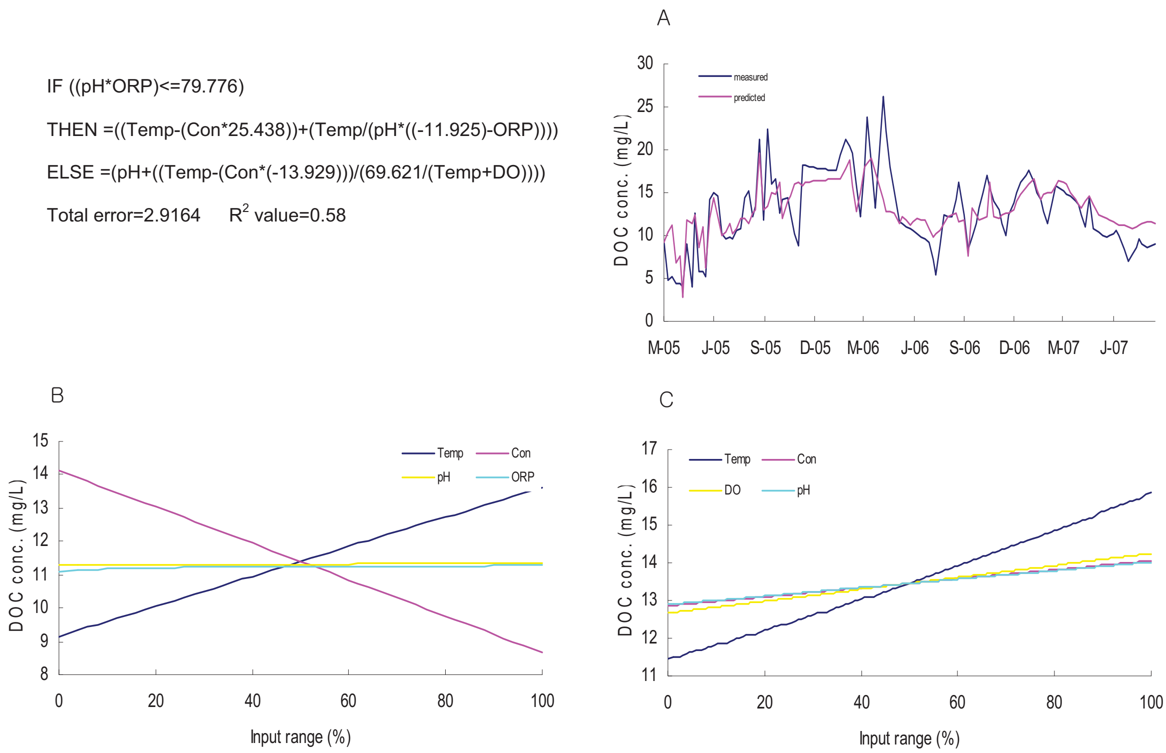


Figure 5-57: 7 days ahead forecasting results for DOC at the outflow site. A. Comparison between the measured and predicted TN concentrations; B. Sensitivity for THEN branch and C. Sensitivity for ELSE branch

5.5.3 Discovery of predictive rules for the different nutrients by HEA

5.5.3.1 Total Nitrogen

The predictions for the TN concentrations in the reed bed were performed for the inflow, site1-site2, site3-site4 and outflow sites. The results produced by HEA are shown in the Figure 5-58, 5-59, 5-60 and 5-61. Figure 5-58 shows the HEA result for TN at the inflow site. The input variables selected for the HEA models of TN produced for the different sites are summarized in Table 5-20.

Table 5-20: Input variables selected for HEA modeling of TN

inflow	site 1	site 2	site 3	site 4	outflow
WT	WT	WT	WT	WT	WT
Con	Con	Con	Con	Con	Con
DO	DO	DO	DO	DO	DO
pH	pH	pH	pH	pH	pH
ORP	ORP	ORP	ORP	ORP	ORP
Turb.	Turb.	Turb.	Turb.	Turb.	Turb.
$TN_{(w)}$ conc. (sto)	$TN_{(w)}$ conc. (in)	$TN_{(w)}$ conc. (S1)	$TN_{(w)}$ conc. (S2)	$TN_{(w)}$ conc. (S3)	$TN_{(w)}$ conc. (S4)
$TN_{(sed)}$ conc. (in)	$TN_{(sed)}$ conc. (S1)	$TN_{(sed)}$ conc. (S2)	$TN_{(sed)}$ conc. (S3)	$TN_{(sed)}$ conc. (S4)	$TN_{(sed)}$ conc. (out)
Evp.	Evp.	Evp.	Evp.	Evp.	Evp.
Rainfall	Rainfall	Rainfall	Rainfall	Rainfall	Rainfall

w = water, sed = sediment,
sto=storage, in = inflow, S1 = Site 1, S2 = Site 2, S3 = Site 3, S4 = Site 4, out = outflow
WT = Water Temperature, Con = Conductivity, DO = Dissolved Oxygen, ORP = Redox Potential, Turb. = Turbidity and Evp. = Evaporation

The HEA produced a rule set, which predicted the TN concentration at the inflow with an accuracy of R^2 = 0.65. The IF-THEN-ELSE rule created by HEA predicted most of the peak events. In most cases the predictions were underestimated. The selection of the equation to calculate the TN concentration, either THEN or ELSE, was chosen by verification of IF condition. If the IF condition was true the THEN branch was selected and if the IF condition was false the ELSE branch was selected to calculate the TN concentration. The driving parameters to predict the TN concentration for the THEN branch were turbidity, evaporation and the TN concentration from the previous site. The parameters turbidity and TN concentration from the previous site showed in the sensitivity analysis a positive influence on the prediction of the TN concentration, which means with an increase of these parameters leads to an increase of TN concentration calculated or predicted at the inflow site. High turbidity readings indicate high amounts of particles in the water column, due to the decomposition of organic matter or resuspension, in both cases nutrients will leach back into the water column. The concentration of TN at the previous site would highly affect the nutrient concentration at the current site, which would be considered as an input and has been detected by HEA. Evaporation had a slight negative influence on the TN concentration at the inflow site. Negative effect means that an increasing value of the input parameter causes decreasing values of the predicted output parameter. The TN concentration at the inflow site would higher at low evaporation, which would be a link to the rainy seasons. The rainfall is a major source for nutrients in particular nitrogen. During the summer periods, where usually the evaporation is relatively high the TN concentration is showing a slightly decreasing trend, but there is no big difference in the concentration at low and high evaporation.

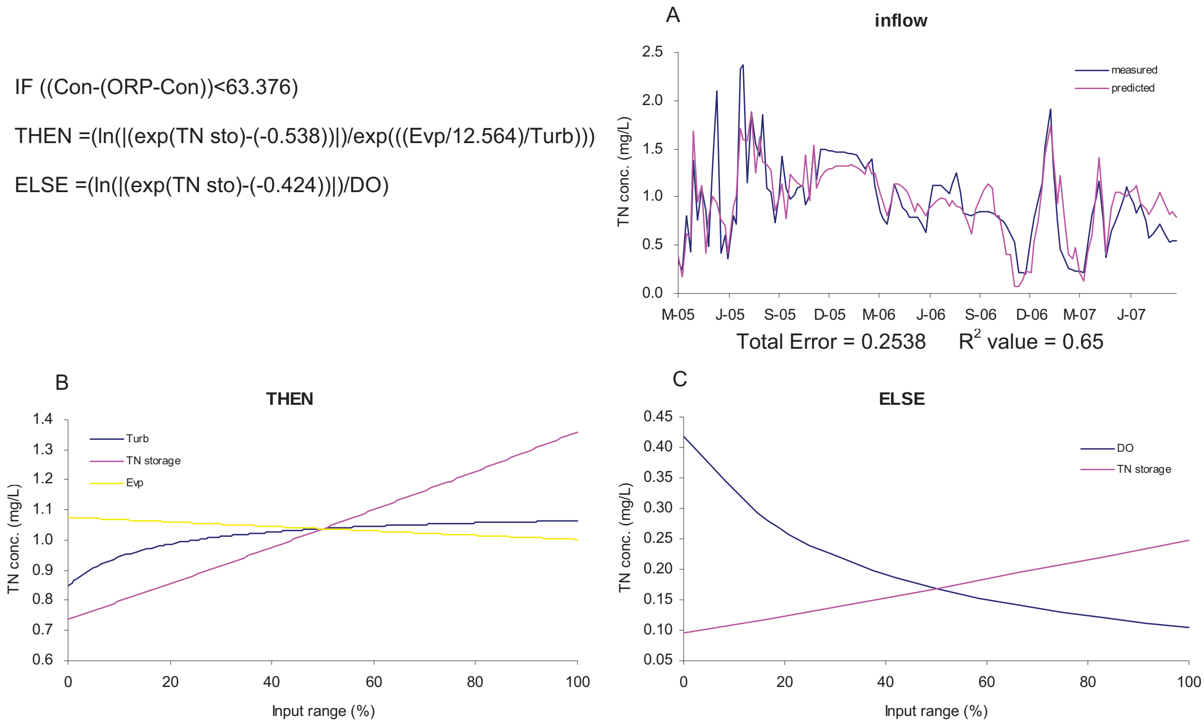


Figure 5-58: HEA results for TN at the inflow site. A. Comparison between the measured and predicted TN concentrations; B. Sensitivity for THEN branch and C. Sensitivity for ELSE branch

The ELSE branch had the parameters DO and TN concentration from the previous site as the most influential. DO have a negative influence, whereas the concentration from the previous site had a positive influence on the prediction of TN concentration at the inflow site. Low DO concentration in the water column will indicate anaerobic conditions, which will lead to a higher TN concentration in water column, whereas high concentrations will allow the system to transform the nitrogen into other components due to the process of nitrification. The nutrient concentration from the previous site has a great influence on the nutrient concentration at the current site.

IF (((Turb/prevS)/prevS) >= 317.767)

THEN =((prevS-((prevS+Con)/((-186.394)-ORP)))+Con)

ELSE =(prevS+((prevS/(-41.180))/(ln(|Con|)/(DO+(-8.846)))))

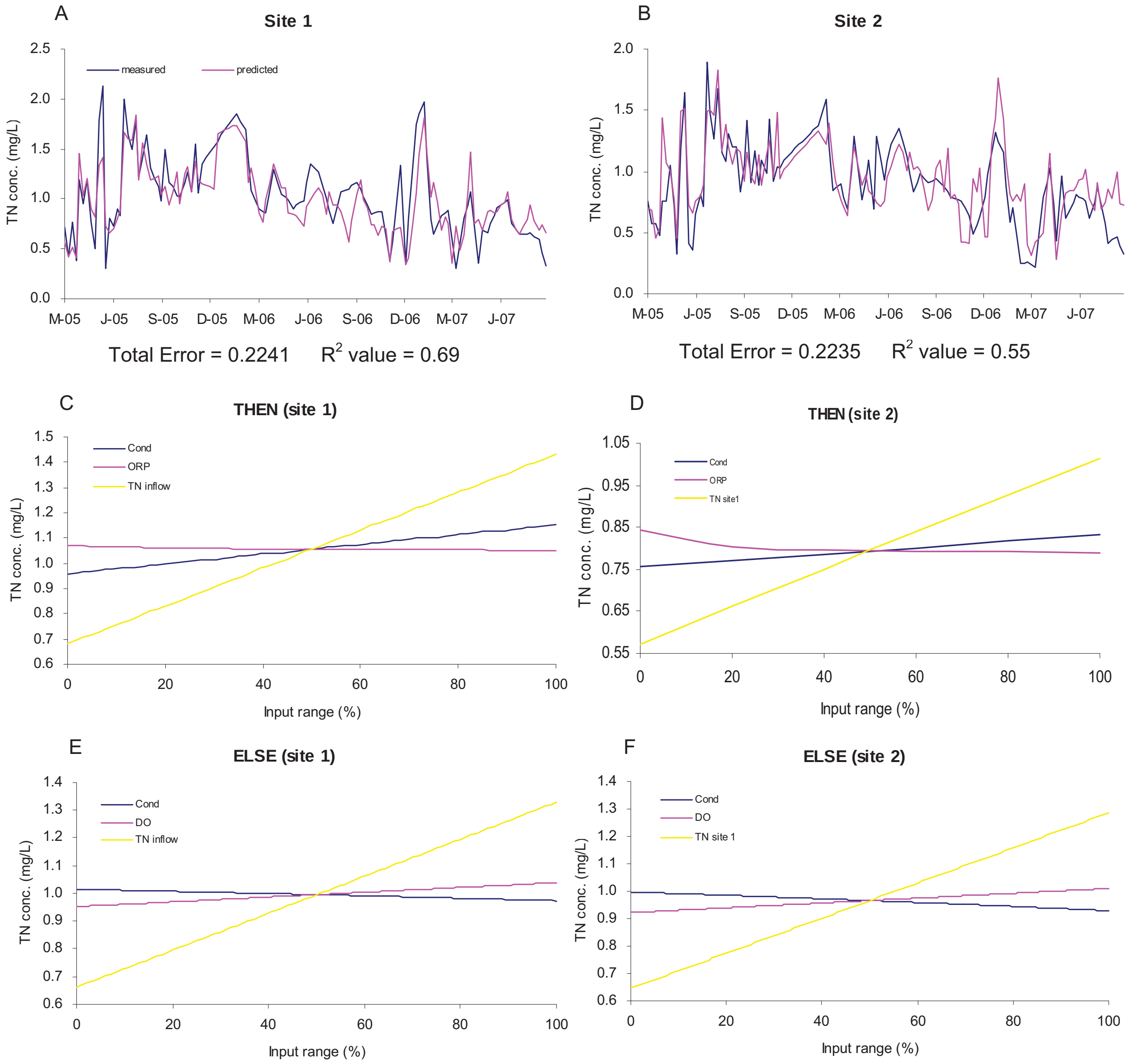


Figure 5-59: HEA results for TN at site 1 and 2. A. Comparison between the measured and predicted TN concentrations at site 1; B. Comparison between the measured and predicted TN concentrations at site 2; C. & D. Sensitivity for THEN branch (site 1 & 2); E. & F. Sensitivity for ELSE branch (site 1 & 2)

Figure 5-59 is showing the predictions of TN concentrations at site 1 and 2 including the sensitivity analysis for the THEN and ELSE branch. HEA created a rule set, which was used to predict the TN concentration at site 1 and 2. The accuracy of the prediction of this rule set had a R^2 value of 0.69 and 0.55 for site 1 and 2. The prediction for both sites showed that most of peak events were detected by the HEA and were mostly underestimated.

IF (Evp>18.862)

THEN =(((ORP+412.639)/(exp(DO)-(-376.685)))*prevS)

ELSE =((((prevS*(-114.926))+375.583)/((11.780-Rain)-(-316.131)))*prevS)

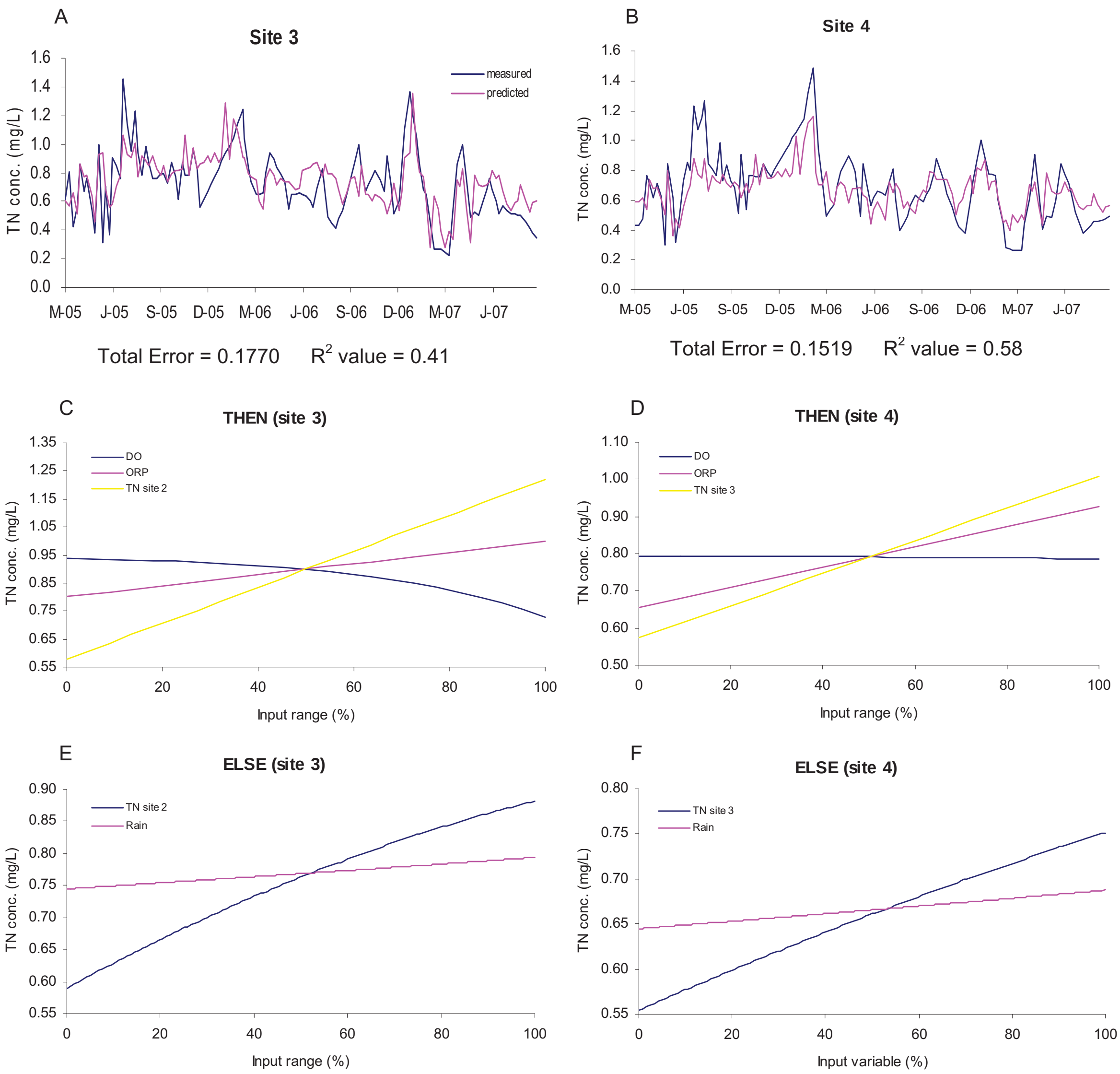


Figure 5-60: HEA results for TN at site 3 and 4. A. Comparison between the measured and predicted TN concentrations at site 3; B. Comparison between the measured and predicted TN concentrations at site 4; C. & D. Sensitivity for THEN branch (site 3 & 4); E. & F. Sensitivity for ELSE branch (site 3 & 4)

The sensitivity analysis for the THEN branch showed that the parameters of the ORP, conductivity and TN concentration from the previous site were chosen as input parameter. With the exception of ORP the other two parameters had a positive impact on the TN concentration at site 1 and 2. For both sites the concentration from the previous site had the biggest influence on the TN concentrations at both sites. The influence of both ORP and conductivity, which had both a negative and positive effects, was minimal. The sensitivity analysis of the ELSE branch showed that DO, conductivity and TN concentration from the previous site were selected as input parameter by the HEA to predict the TN concentration of sites 1 & 2. Similar to the THEN branch the nutrient concentration from the previous site had the biggest impact on output parameter. The sensitivity analysis for both sites showed a strong positive impact of this parameter, whereas the influence of the parameters DO and conductivity showed only small changes in the output variable.

Figure 5-60 is showing the rule set and the predictions made regards the TN concentrations at site 3 and 4. The accuracy of the predictions made had a R^2 value of 0.41 and 0.58 for the site 3 and 4, respectively. Most of the peak events have been detected by the HEA, with the exception of the winter peak in 2005. At both sites this peak event was completely underestimated.

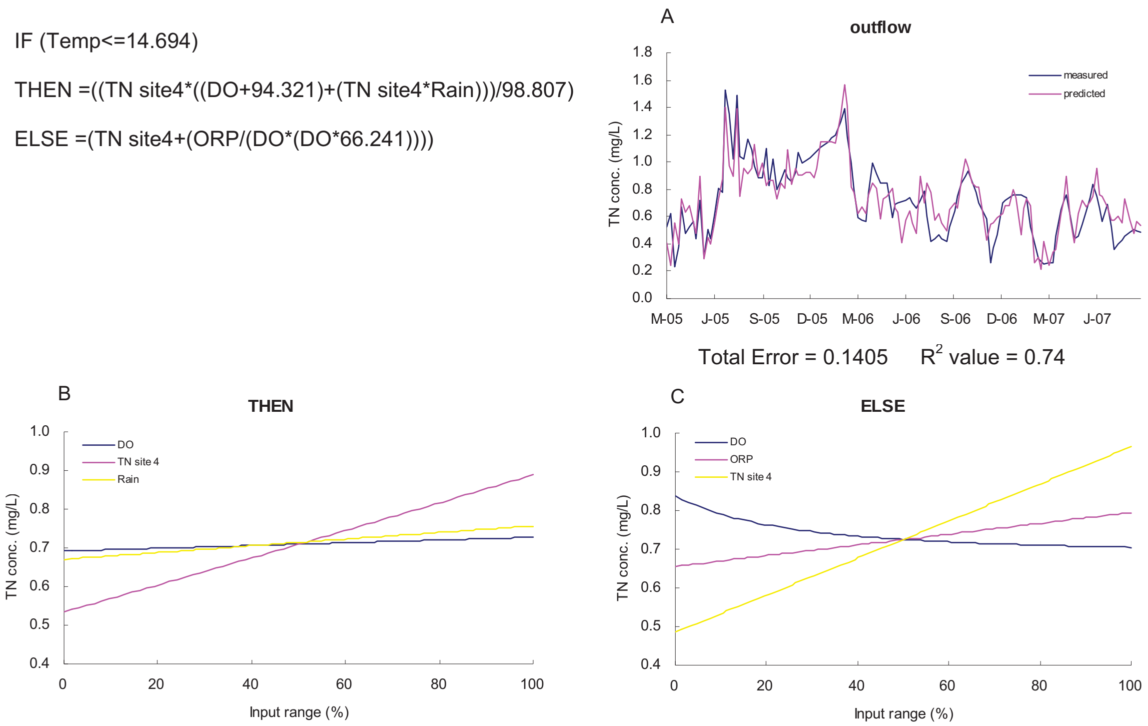


Figure 5-61: HEA results for TN at outflow site. A. Comparison between the measured and predicted TN concentrations; B. Sensitivity for THEN branch and C. Sensitivity for ELSE branch

The sensitivity analysis of the THEN branch showed that the input parameters DO, ORP and TN concentration from the previous site were chosen to predict the output variable. With the exception of the input parameter DO the other two variables had positive influence on the TN concentration at site 3 and 4. Like to the predictions of the previous sites the nutrient concentration from the previous site has an important influence in determining the nutrient concentration at the current site. As input variables for the ELSE branch were the parameters rain and nutrient concentration from the previous site were selected. Both input variables had a positive impact on the TN concentration at site 3 and 4. Again the most influential parameter to determine the output parameter was the nutrient concentration from the previous site.

Figure 5-61 is showing the predicted TN concentration at the outflow site. The accuracy of prediction had a R^2 value of 0.74. The HEA predicted all of the peak events matching well the measured value. The sensitivity analysis for the THEN branch had the input parameters DO, rain and TN concentration from the previous site. All of the input variables had a positive effect on the nutrient concentration at the outflow site. The input variables selected by the HEA for the ELSE branch were DO, ORP and TN concentration from the previous site. With the exception of DO the other input variables had a positive influence on TN concentration at the outflow site. Like to the predictions at the previous sites the parameter of TN concentrations from the previous site had the biggest impact on the nutrient concentration at the current site.

The prediction results for the nitrogen components in form of nitrate and ammonium can be found in Appendix E.

5.5.3.2 Total Phosphorus

The prediction of TP concentration by using the HEA was performed in the same way as the prediction of TN concentration. The input variables selected for the HEA models of TP produced for the different sites are summarized in Table 5-21. Figure 5-62 is showing the prediction results and the sensitivity analysis for the TP concentration at the inflow site. The prediction of concentration at the inflow was fairly accurate. The accuracy had a R^2 value of 0.55. The HEA predicted most of the peak events, with the exception of April 2005 and August 2007.

The sensitivity analysis was performed for the THEN and ELSE branch. For the THEN branch the following input variables were selected to calculate the nutrient concentration, which are conductivity, rain and TP concentration from the previous site. Conductivity had more or less no influence on the nutrient concentration, whereas rain and the nutrient levels from the previous site had a positive effect. The input parameters for the ELSE branch included the same parameters like the THEN branch with an additional variable in form of pH.

Table 5-21: Input variables selected for HEA modeling of TP

inflow	site 1	site 2	site 3	site 4	outflow
WT	WT	WT	WT	WT	WT
Con	Con	Con	Con	Con	Con
DO	DO	DO	DO	DO	DO
pH	pH	pH	pH	pH	pH
ORP	ORP	ORP	ORP	ORP	ORP
Turb.	Turb.	Turb.	Turb.	Turb.	Turb.
$TP_{(w)}$ conc. (sto)	$TP_{(w)}$ conc. (in)	$TP_{(w)}$ conc. (S1)	$TP_{(w)}$ conc. (S2)	$TP_{(w)}$ conc. (S3)	$TP_{(w)}$ conc. (S4)
$TP_{(sed)}$ conc. (in)	$TP_{(sed)}$ conc. (S1)	$TP_{(sed)}$ conc. (S2)	$TP_{(sed)}$ conc. (S3)	$TP_{(sed)}$ conc. (S4)	$TP_{(sed)}$ conc. (out)
Evp.	Evp.	Evp.	Evp.	Evp.	Evp.
Rainfall	Rainfall	Rainfall	Rainfall	Rainfall	Rainfall

w = water, sed = sediment
sto=storage, in = inflow, S1 = Site 1, S2 = Site 2, S3 = Site 3, S4 = Site 4, out = outflow
WT = Water Temperature, Con = Conductivity, DO = Dissolved Oxygen, ORP = Redox Potential, Turb. = Turbidity and Evp. = Evaporation

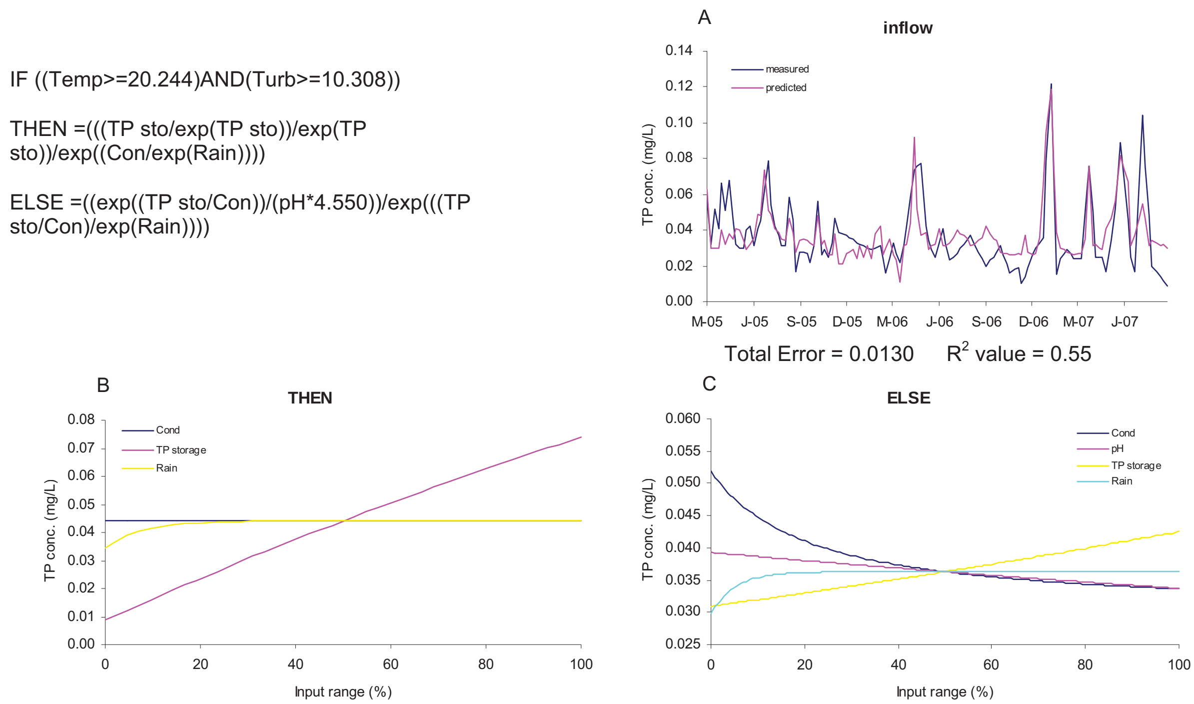


Figure 5-62: HEA results for TP at inflow site. A. Comparison between the measured and predicted TP concentrations; B. Sensitivity for THEN branch and C. Sensitivity for ELSE branch

The influence of the input variables of the ELSE branch showed a slight difference compared with the THEN branch. The input parameters rain and TP concentration from the previous site had a positive effect, which is similar to the THEN branch, but conductivity and pH had a negative impact. The behavior of the output parameter responded differently in the ELSE branch for conductivity parameter than to the THEN branch.

IF (sedTP<375.152)

THEN =((((Con*Evp)/ln(|Con|))-Temp)/((ORP/ln(|Con|))-433.184))

ELSE =((((20.998/DO)/DO)-Temp)/((ORP/(-0.689))-138.233))

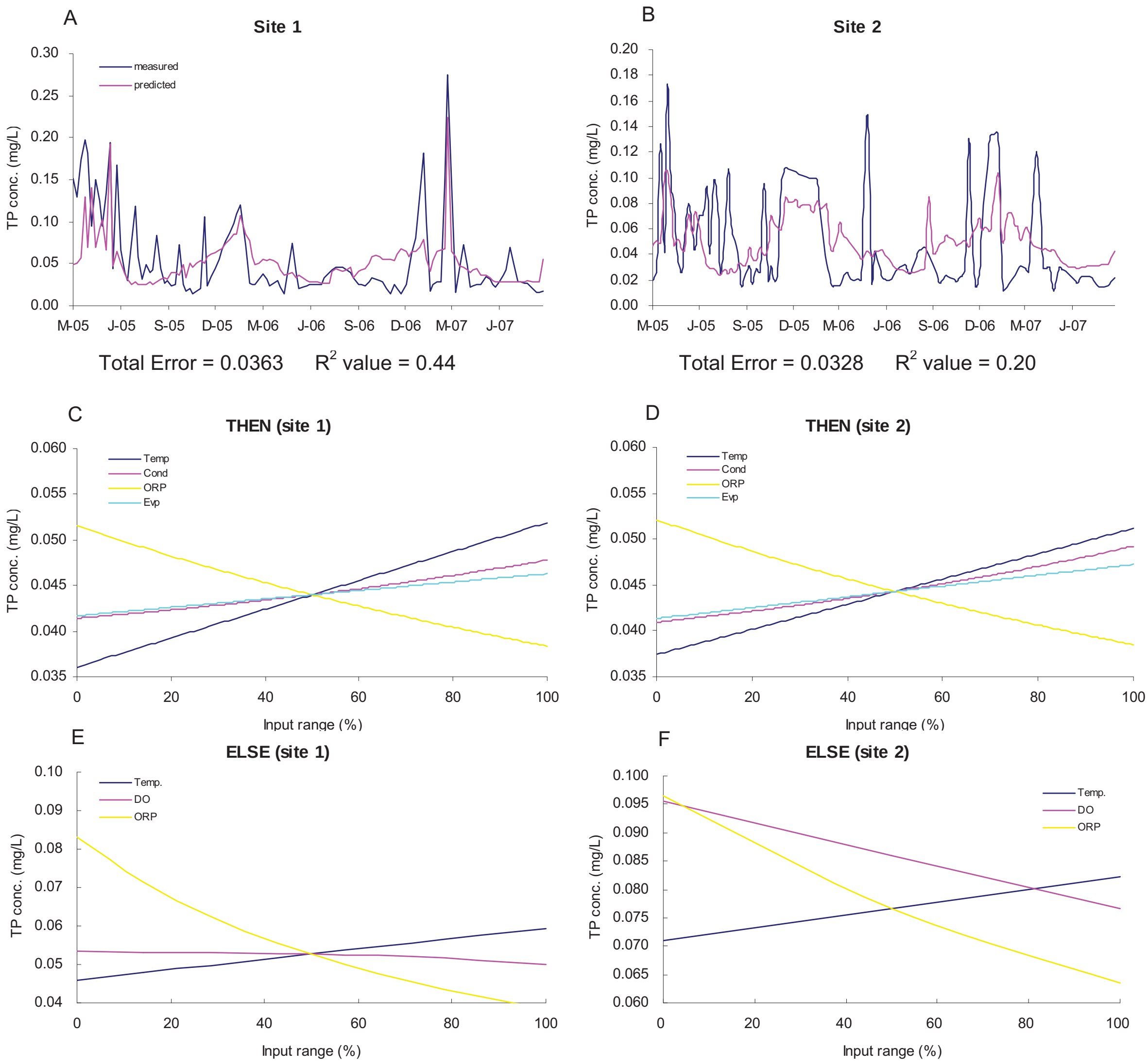


Figure 5-63: HEA results for TP at site 1 and 2. A. Comparison between the measured and predicted TP concentrations at site 1; B. Comparison between the measured and predicted TP concentrations at site 2; C. & D. Sensitivity for THEN branch (site 1 & 2); E. & F. Sensitivity for ELSE branch (site 1 & 2)

Figure 5-63 is showing the TP concentration prediction for site 1 and 2. The rule set developed by the HEA performed differently in the accuracy of prediction. The same rule set was used to calculate the TP concentrations at both sites. The accuracy of prediction had a R^2 value of 0.44 and 0.20 for site 1 and 2; therefore the rule set was predicting the output parameter for site 1 more accurately than for site 2, which can be also seen in the Figure 5-66. The prediction for site 1 showed that not all, but major peak events were detected by the HEA. The same rule set as used for the prediction at site 2 failed to match some of the peak events and more or less underestimated other events. The sensitivity analysis of the THEN branch showed that the following input variables were chosen to calculate the TP for site 1 and 2: water temperature, conductivity, ORP and evaporation. The input parameters with the exception of ORP had all a positive effect on the output parameter. The sensitivity analysis of the ELSE branch chose water temperature, DO and ORP as input variable. The parameters DO and ORP had a negative effect on the TP concentration at site 1 and 2. Figure 5-64 is showing the prediction of TP concentrations at site 3 and 4. The R^2 values of the predictions were 0.43 for site 3 and 0.41 for site 4. The predictions of the TP concentrations at site 3 and 4 were mostly underestimated and some of the smaller peak events were missed by HEA at both sites. At site 4 for the calculation of TP concentration only the equation of ELSE branch was used, therefore no sensitivity analysis of the THEN branch is available for site 4. The sensitivity analysis for the THEN branch for site 3 showed that the input variables ORP, turbidity, sediment TP concentration and TP concentration from the previous site were selected to calculate the output parameter. The parameter of sediment TP showed a negative effect on the output parameter, where the change of output parameter based on the input range was minimal. The rest of the input variables showed a positive effect, especially turbidity and TP concentration from the previous site, which showed great variations between the input ranges. The ELSE branch used pH, ORP, turbidity, TP concentration from the previous site and sediment TP as the input variables to calculate the output parameter. At both sites with the exception of the input variable ORP all other inputs had a positive effect on the output parameter.

Figure 5-65 is showing HEA results of the prediction of TP concentration at the outflow site. The R^2 value of the prediction is 0.53. The rule set predicted most of the peak events, missing one event in September 2006. The THEN branch used only the parameter of TP concentration from the previous site as input to calculate the output parameter. The ELSE branch selected the parameters conductivity and turbidity as inputs. All inputs used in the equations of the THEN and ELSE branch had a positive effect in calculating the TP concentration at the outflow area.

The prediction results of the HEA for phosphate for the different sites can be found in Appendix E.

IF (DO>10.684)

THEN =(Turb/(((Turb*prevS)*sedTP)+(41.077-ORP)))

ELSE =(pH/(((Turb-ORP)/(-3.636))-((prevS*sedTP)+(-243.914))))

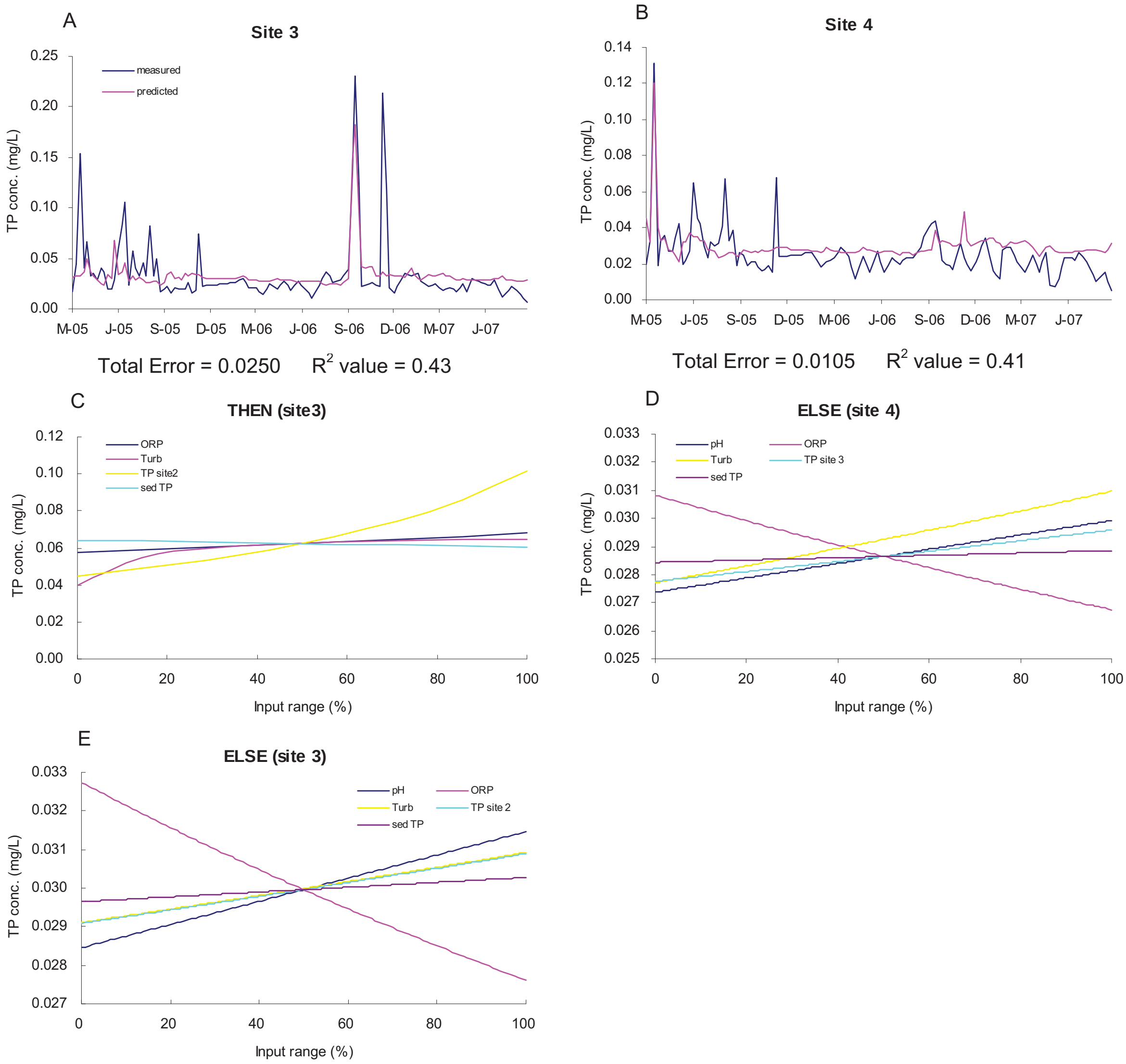


Figure 5-64: HEA results for TP at site 3 and 4. A. Comparison between the measured and predicted TP concentrations at site 3; B. Comparison between the measured and predicted TP concentrations at site 4; C. Sensitivity for THEN branch (site 3); D. & E. Sensitivity for ELSE branch (site 4 & site 3)

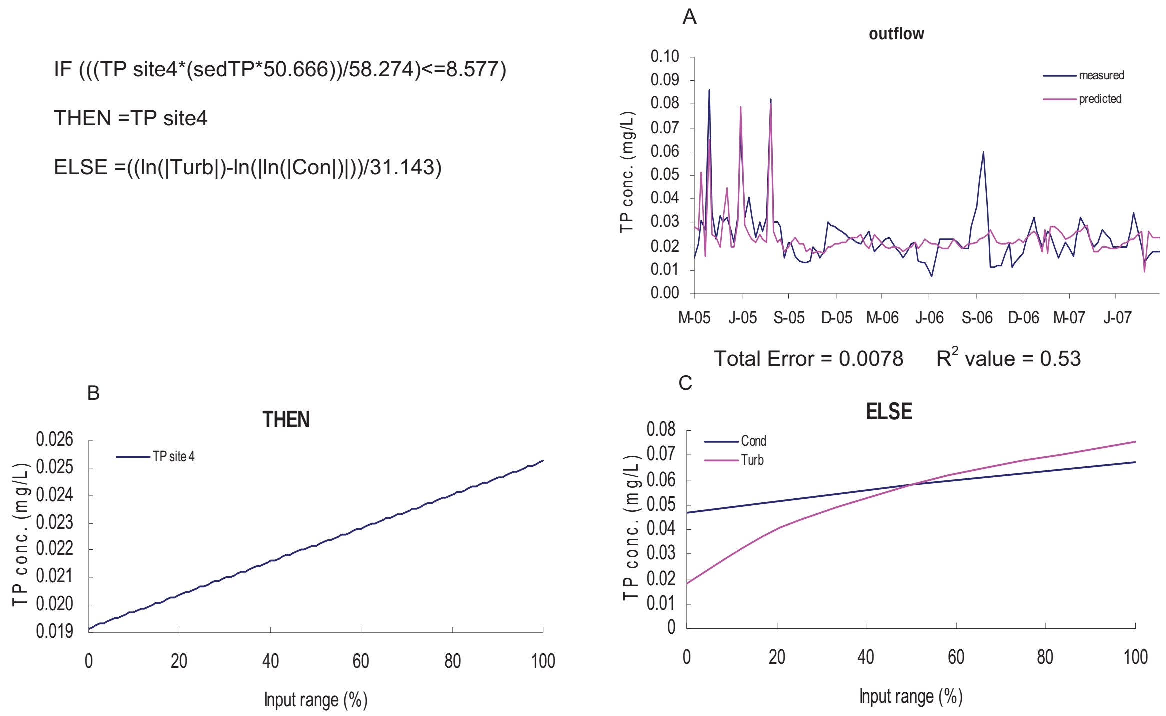


Figure 5-65: HEA results for TP at outflow site. A. Comparison between the measured and predicted TP concentrations; B. Sensitivity for THEN branch and C. Sensitivity for ELSE branch

5.5.3.3 Dissolved Organic Carbon

The predictions of DOC concentrations for the different sites were performed the same way like the other nutrient forms. The input variables selected for the HEA models of DOC produced for the different sites are summarized in Table 5-22.

Figure 5-66 is showing the result of prediction and the sensitivity results for the THEN and ELSE branch. The accuracy of the prediction of the DOC concentration at the inflow area had a R^2 value of 0.40. The comparison between the measured and calculated DOC concentration showed that most of the major peak events had been detected, but the HEA missed or underestimated some of the middle peak events. The inputs selected in the equation of the THEN branch were water temperature and the DOC concentration from the previous site. Both parameters had a positive effect on the output parameter. The inputs selected to calculate the DOC concentration were water temperature, pH and chl-a. The inputs of the ELSE branch also showed a positive influence on the DOC concentration of the inflow area.

Table 5-22: Input variables selected for HEA modeling of DOC

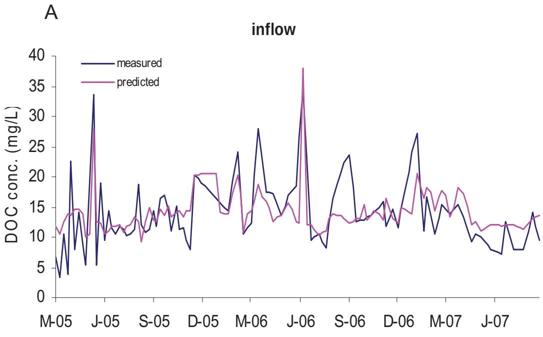
inflow	site 1	site 2	site 3	site 4	outflow
WT	WT	WT	WT	WT	WT
Con	Con	Con	Con	Con	Con
DO	DO	DO	DO	DO	DO
pH	pH	pH	pH	pH	pH
ORP	ORP	ORP	ORP	ORP	ORP
Turb	Turb	Turb	Turb	Turb	Turb
$DOC_{(w)}$ conc. (sto)	$DOC_{(w)}$ conc. (in)	$DOC_{(w)}$ conc. (S1)	$DOC_{(w)}$ conc. (S2)	$DOC_{(w)}$ conc. (S3)	$DOC_{(w)}$ conc. (S4)
chl-a $_{(w)}$ (in)	chl-a $_{(w)}$ (S1)	chl-a $_{(w)}$ (S2)	chl-a $_{(w)}$ (S3)	chl-a $_{(w)}$ (S4)	chl-a $_{(w)}$ (out)
Evp.	pl biomass (S1)	pl biomass (S2)	pl biomass (S3)	pl biomass (S4)	Evp.
Rainfall	Evp.	Evp.	Evp.	Evp.	Rainfall
	Rainfall	Rainfall	Rainfall	Rainfall	

w = water, pl = plant; sto=storage, in = inflow, S1 = Site 1, S2 = Site 2, S3 = Site 3, S4 = Site 4, out = outflow
WT = Water Temperature, Con = Conductivity, DO = Dissolved Oxygen, ORP = Redox Potential, Turb. = Turbidity and Evp. = Evaporation

IF ((exp(DO)<10.289)OR(DO>=10.289))

THEN =(Temp-(4.696/ln(|(stoDOC+(-6.495))|)))

ELSE =(pH+(ln(|(chla/Temp)|)+pH))



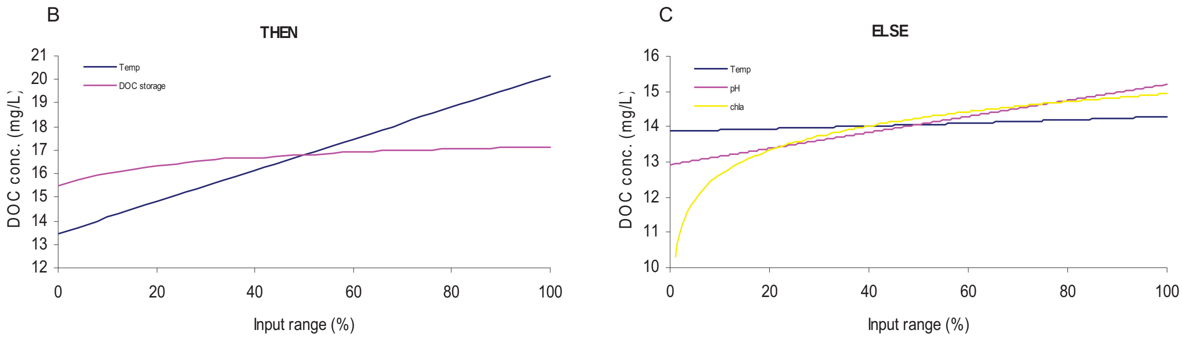


Figure 5-66: HEA results for DOC at inflow site. A. Comparison between the measured and predicted DOC concentrations; B. Sensitivity for THEN branch and C. Sensitivity for ELSE branch

Figure 5-67 is showing the prediction results by HEA and the sensitivity analysis results for the THEN and ELSE for the sites 1 and 2. The prediction accuracy of DOC concentration had a R^2 value of 0.31 for site 1 and 0.36 for site 2. However the rule-set created by HEA couldn't detect all the major peak events. The rule set missed some of the peak events and overestimated the DOC concentration in June 2006. The equation of the THEN branch used the parameters of the DOC concentration from the previous site and rain to calculate the DOC concentration at site 1 and 2.

IF ((DO*(DO*(DO*25.462)))<94.339)

THEN =(ln(|exp(prevS)|)+(2.600/((Rain-118.497)+(prevS*prevS))))

ELSE =(ln(|exp(prevS)|)+(prevS/(-8.283)))

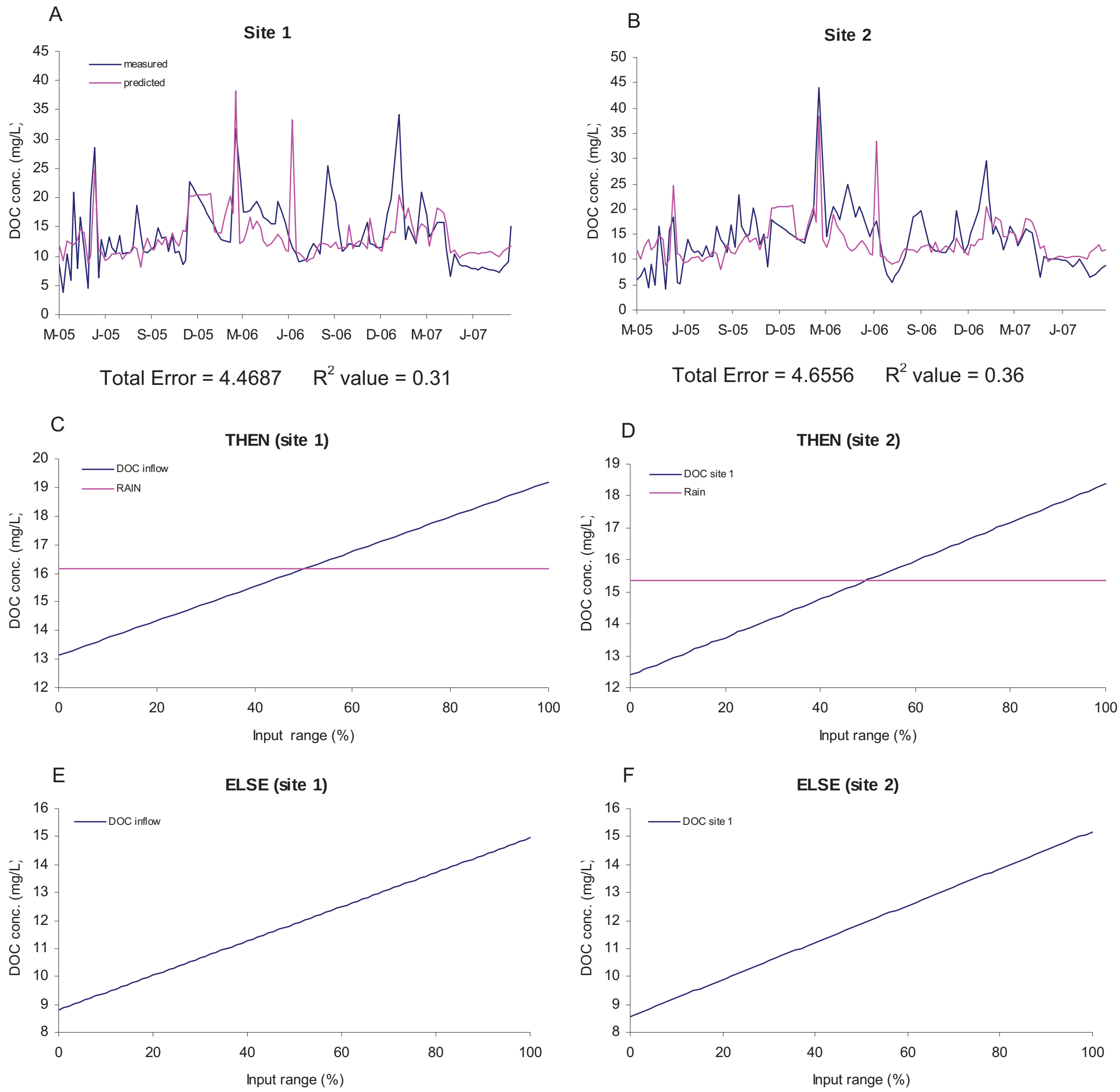


Figure 5-67: HEA results for DOC at site 1 and 2. A. Comparison between the measured and predicted DOC concentrations at site 1; B. Comparison between the measured and predicted DOC concentrations at site 2; C. & D. Sensitivity for THEN branch (site 1 & 2); E. & F. Sensitivity for ELSE branch (site 1 & 2)

The sensitivity analysis showed for both sites, that the input range of rain didn't influence the DOC concentration at all, whereas the nutrient condition from the previous site had a positive influence on the output parameters. The ELSE branch used only the parameter of the DOC concentration from the previous site to predict the DOC concentration at site 1 and 2. Like the parameter input in the THEN branch the input variable in the ELSE branch had a positive influence on the output parameter.

IF (((exp(ORP)*(ORP-prevS))<=52.237)OR((prevS-(Rain-78.773))>96.498))

THEN =((((Temp*Temp)+98.407)+Temp)/((Con+(Con+30.799))-ln(|(ORP+23.961)|)))

ELSE =prevS

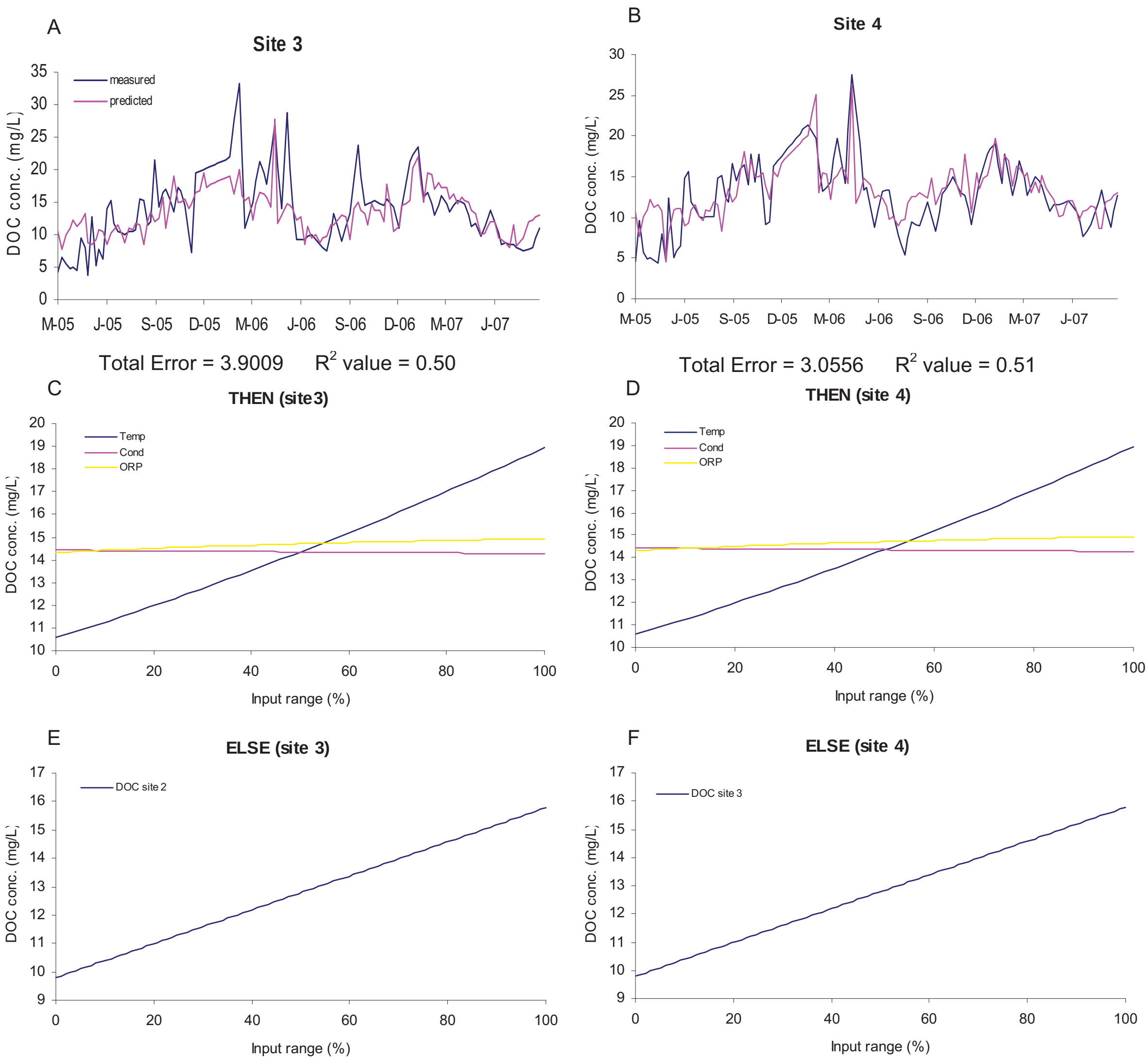


Figure 5-68: HEA results for DOC at site 3 and 4. A. Comparison between the measured and predicted DOC concentrations at site 3; B. Comparison between the measured and predicted DOC concentrations at site 4; C. & D. Sensitivity for THEN branch (site 3 & 4); E. & F. Sensitivity for ELSE branch (site 3 & 4)

Figure 5-68 is showing the prediction results and sensitivity analysis results by the HEA for site 3 and 4. The prediction accuracy had a R^2 value of 0.50 for site 3 and 0.51 for site 4. The rule set created by the HEA predicted most of the major peak events, but at site 3 the rule set missed a peak event around February 2006 and at site 4 the rule set overestimated the DOC concentration. The input variables in the equation of the THEN branch were water temperature, conductivity and ORP. The conductivity had a negative influence, but the variations between the low and high input ranges were almost equal to each other. The other two inputs had a positive influence on the DOC concentrations.

The ELSE branch determined the DOC concentration by using the DOC concentration from the previous site and sensitivity analysis showed that the input parameter had a positive influence on the output parameter.

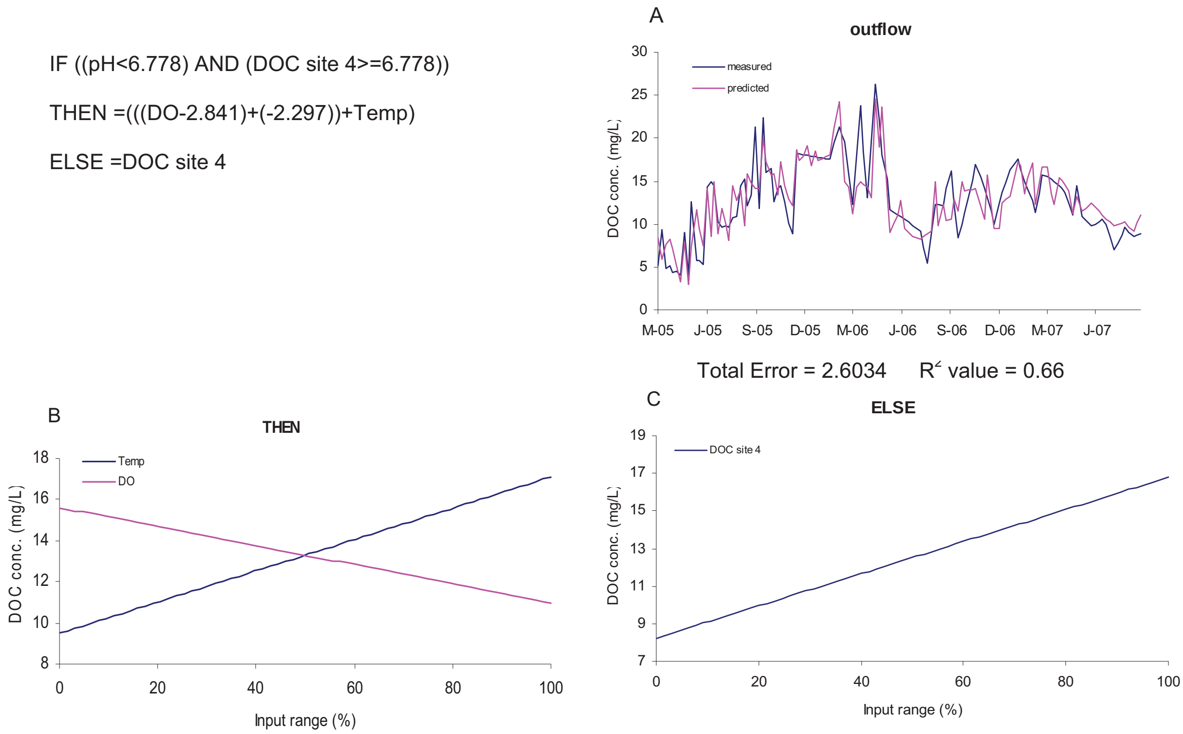


Figure 5-69: HEA results for DOC at outflow site. A. Comparison between the measured and predicted DOC concentrations; B. Sensitivity for THEN branch and C. Sensitivity for ELSE branch

Figure 5-69 is showing the HEA prediction and sensitivity analysis results for the outflow area. The prediction accuracy had a R^2 value 0.66. The rule set created by HEA predicted most of the major peak events accurately, but missed two peak events completely, but the predicted trends and magnitudes are similar to the measured data. The equation of the THEN branch used water temperature and DO as input variables and the ELSE branch selected the DOC concentration from the previous site to calculate the output parameter. The sensitivity analyses of the THEN branch showed that the water temperature had a positive influence and DO a negative effect on the output parameter. The sensitivity analysis results of the ELSE branch showed that the input variable had a positive influence on the DOC concentration for the outflow area.

5.5.4 General Rule Application of HEA

The following approach is to find a general rule set, which can be used to predict the different nutrients for the whole reed bed system. The selection of the general rule set was chosen from the rules developed for the prediction of the different nutrients for the different sites, which are documented in the previous section 5.5.3.

5.5.4.1 General rule for Total Nitrogen prediction using HEA

The general rule set selected for the prediction of TN for the whole reed bed system is the rule set, which was developed to predict TN concentration at site 1 & 2. The rule set is defined as follows:

IF (((Turb/prevS)/prevS)>=317.767)

THEN =((prevS-((prevS+Con)/((-186.394)-ORP)))+Con)

ELSE =(prevS+((prevS/(-41.180))/(ln(|Con|)/(DO+(-8.846)))))

The prediction accuracy of the rule set selected for the TN prediction has summarized in Table 5-23 for the different sites in the reed bed pond.

Table 5-23: R^2 and total error of the rule set for the different sites

	inflow	site 1	site 2	site 3	site 4	outflow
Total Error	0.3328	0.2241	0.2235	0.1966	0.1869	0.1649
R^2	0.40	0.69	0.55	0.28	0.37	0.64

The rule set, which was developed for predicting the TN concentration at site 1 & 2 showed therefore the highest accuracy with an R^2 of 0.69 for site 1 and 0.55 for site 2. But the rule set showed high prediction accuracy for the outflow site, which had an R^2 of 0.64.

The comparison of the predicted and measured TN concentration for the different sites showed that for the majority of the sites the prediction in regards timing and magnitude were relatively accurate, especially for sites 1 & 2 (Figure 5-70 B & C). For the inflow area the comparison of the measured and predicted TN concentration showed slight underestimation of TN concentration for a majority of events ranging from a time period starting from April 2005 until August 2006 (Figure 5-70 A). For site 3 in general there were some overestimations of the lower concentration events, but was able to predict most of the major peak events in regard timing and magnitude (Figure 5-70 D). For site 4 the rule set was able to predict most of the lower and medium concentrations, but the rule wasn't able to predict the major peak events, which were completely missed (Figure 5-70 E). For the outflow the model was able to predict most the events, but missed some of the major peak events from a period starting from July 2005 until February 2006 (Figure 5-70 F).

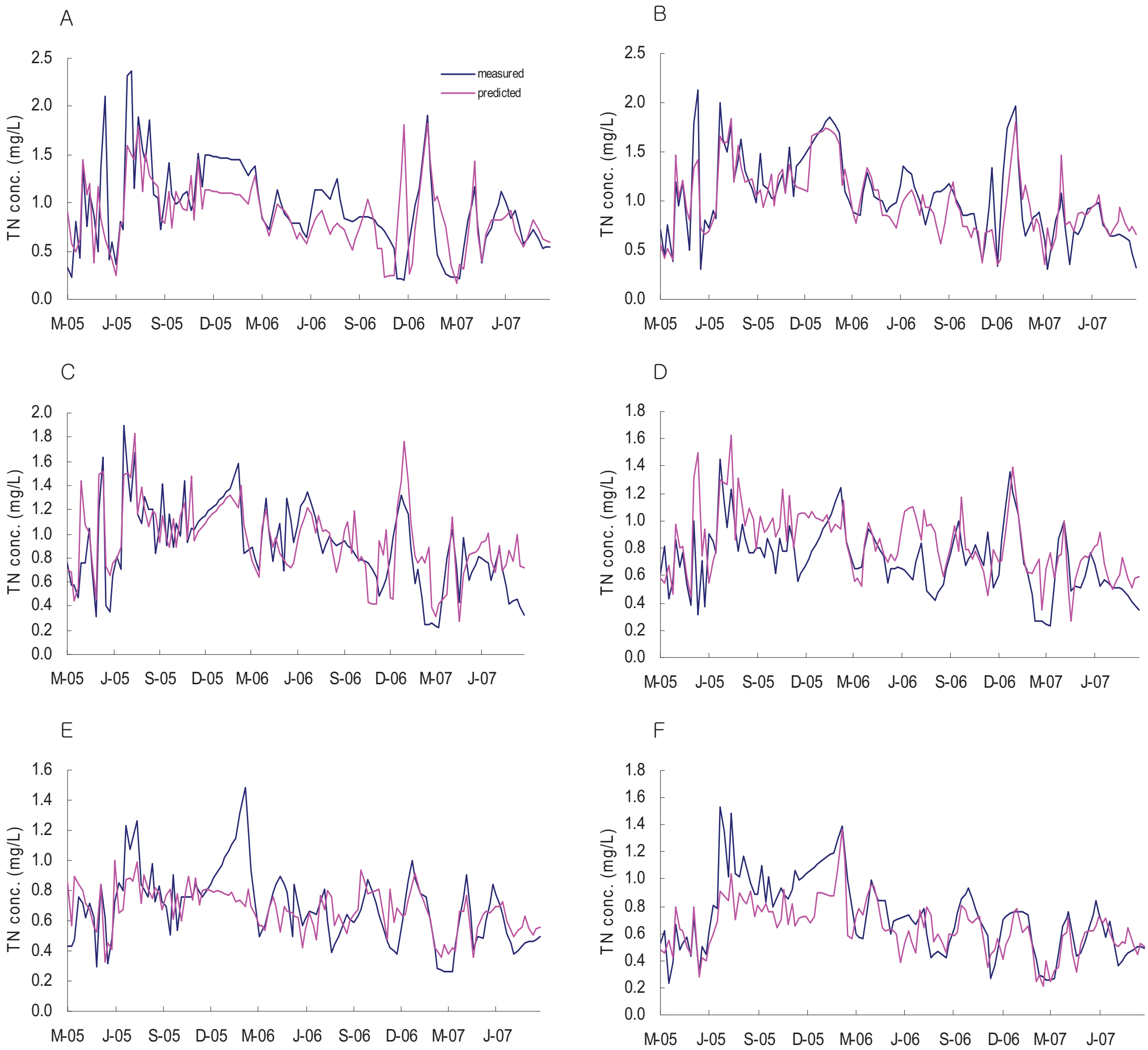


Figure 5-70: General rule application for prediction of TN. A. inflow; B. site 1; C. site 2; D. site 3; E. site 4 and F. outflow

5.5.4.2 General rule for Total Phosphorus prediction using HEA

The general rule set selected for the prediction of TP for the whole reed bed system is the rule set, which was developed to predict TP concentration at site 1 & 2. The rule set is consists of the following equations:

IF (sedTP<375.152)

THEN =((((Con*Evp)/ln(|Con|))-Temp)/((ORP/ln(|Con|))-433.184))

ELSE =((((20.998/DO)/DO)-Temp)/((ORP/(-0.689))-138.233))

The prediction accuracy of the selected rule set for the different sites has been summarized in Table 5-24, but to find a general rule set was didn't to be as accurate as for TN. The prediction accuracies measured in form of R^2 was much lower than compared to TN. For site 1 & 2 the R^2 had been the highest, which is 0.44 and 0.20; the R^2 for the other sites were close 0.00.

Table 5-24: R^2 and total error of the rule set for the different sites

	inflow	site 1	site 2	site 3	site 4	outflow
Total Error	0.0193	0.0364	0.0329	0.0332	0.0137	0.0114
R^2	0.02	0.44	0.20	0.00	0.01	0.01

The graphs showing the comparison between measured and predicted TP concentrations for the different sites showed with the exception of site 1 (Figure 5-71 B), that the predictions of TP were inaccurate in timing and magnitude, which is the reason for the very low R^2 value. The rule set predicted for the inflow site a peak event for September 2005, which didn't match with natural condition monitored and missed peak events towards the end of monitoring period (Figure 5-71 A). For site 1 the rule set was reasonably accurate in predicting some of the major events, with missing some events or underestimating them. The comparison of the predicted and measured TP concentration at site 2 showed that many of the peak events were underestimated. The events which weren't underestimated the prediction were missed by rule set (Figure 5-71 C). For site 3 the rule set was missing almost all major peak events and predicting peak events, which didn't match the natural conditions. Lower concentrations were missed also. Overall the rule set missed the major peak events and overestimated lower concentration events (Figure 5-71 D). For site 4 and outflow the rule set in most cases overestimated the TP concentration. For site 4 the rule set was unable to predict the major peak events in regards timing and magnitude (Figure 5-71 E). For the outflow the rule set was predicted the first peak events relatively well, but then failed to predict the natural trend. In most cases it overestimated and in some cases predicted a peak event, which wasn't confirmed by the natural condition (Figure 5-71 F).

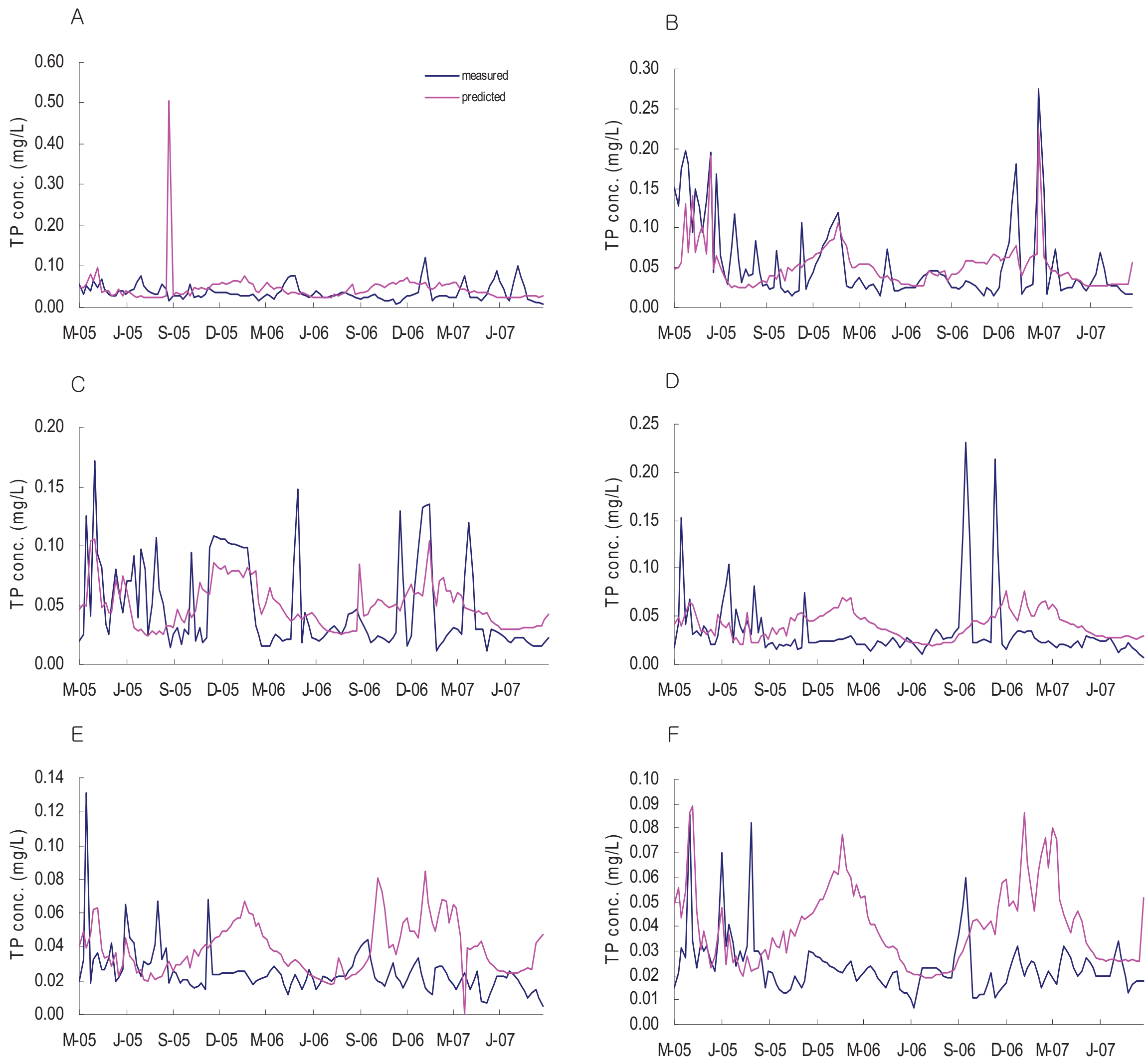


Figure 5-71: General rule application for prediction of TP. A. inflow; B. site 1; C. site 2; D. site 3; E. site 4 and F. outflow

5.5.4.3 General rule for Dissolved Organic Carbon prediction using HEA

The general rule set selected for the prediction of DOC for the whole reed bed system is the rule set, which was developed to predict DOC concentration at site 1 & 2. The rule set is made out of the following equations:

IF ((DO*(DO*(DO*25.462)))<94.339)

THEN =(ln(|exp(prevS)|)+(2.600/((Rain-118.497)+(prevS*prevS))))

ELSE =(ln(|exp(prevS)|)/(prevS/(-8.283)))

The prediction accuracy of the selected rule set for prediction of DOC for the different sites has been summarized in Table 5-25.

Table 5-25: R^2 and total error of the rule set for the different sites

	inflow	site 1	site 2	site 3	site 4	outflow
Total Error	5.2325	4.4686	4.6562	5.0845	3.2212	3.0868
R^2	0.08	0.31	0.36	0.15	0.45	0.52

The prediction accuracy for the different sites was shown as R^2 values and total error. The prediction accuracy was the highest at the outflow area (R^2 = 0.52). The rule set is actually being used to predict DOC concentration at site 1 & 2 with prediction accuracy of 0.31 and 0.36. For site 4 the prediction accuracy is an R^2 = 0.45.

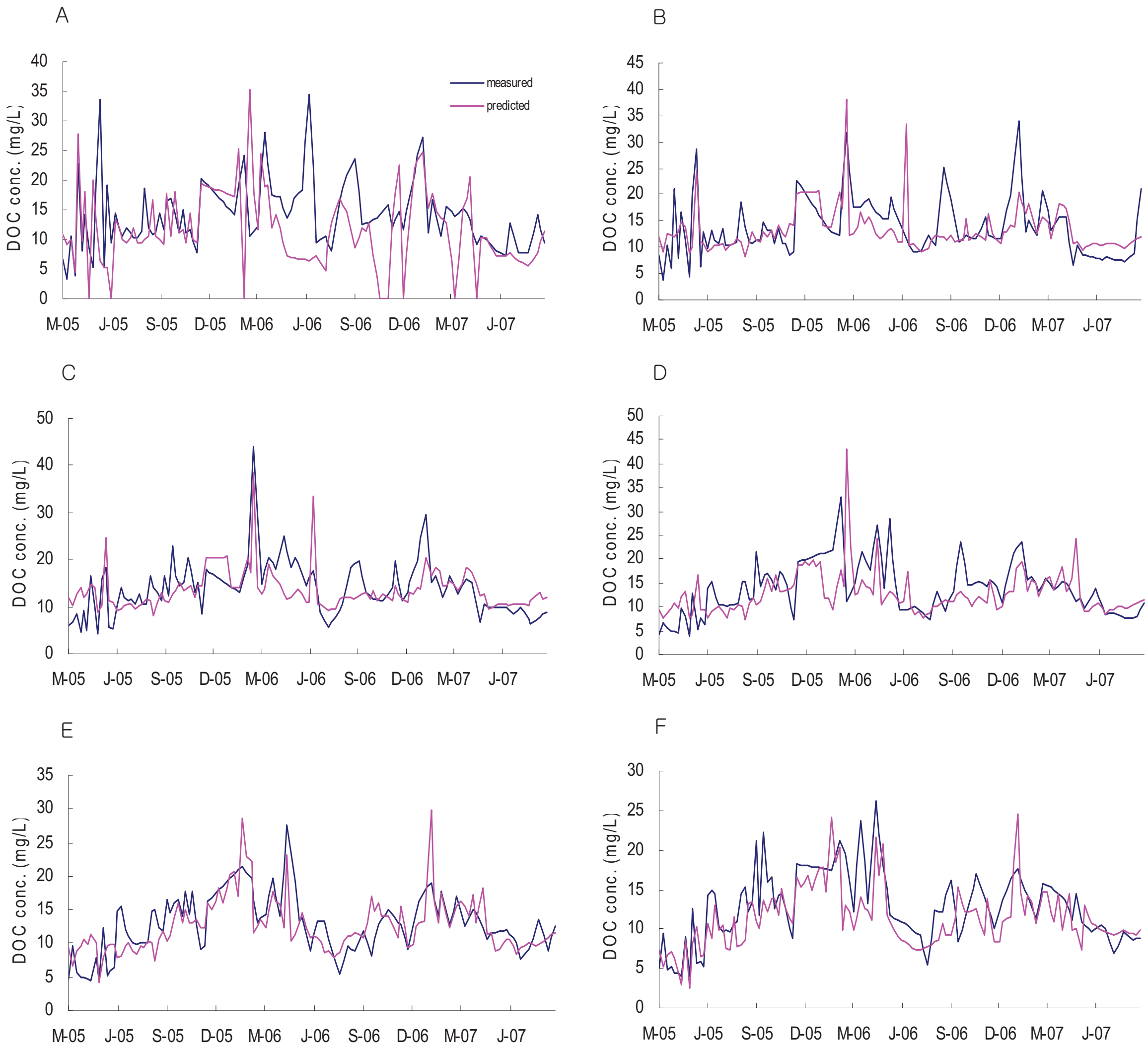


Figure 5-72: General rule application for prediction of DOC. A. inflow; B. site 1; C. site 2; D. site 3; E. site 4 and F. outflow

The comparison of the measured and predicted DOC concentration is visualized in Figure 5-72. The selected rule set predicted for the inflow site most of the major peak events, with the exception of missing two events and underestimating one, but overall the rule set was successfully predicting the DOC concentration at the inflow area (Figure 5-72 A). The rule set was developed to predict the DOC concentration for site 1 & 2. For site 1 the prediction of DOC by the selected rule set showed that most of the peak events were well predicted for both sites 1 & 2, but the rule missed a peak event and underestimated a few events for site 1 (Figure 5-72 B). For site 2 the prediction was similar to that of site 1. Most of the peak events were well predicted in regards magnitude and timing, with a few events underestimated and missed (Figure 5-72 C). For site 3 the rule set prediction was less accurate than the other sites, but still had a higher accuracy than the inflow area. In general the rule underestimated most of the events, but some of the peak events were predicted reasonably in regards timing and magnitude (Figure 5-72 D). The rule showed the highest accuracy in prediction for site 4 and outflow. For site 4 the rule able to predict most peak events in timing and magnitude. In some cases there was slight overestimation of the DOC concentration for site 4, but in general it predicted the trends relatively accurate resulting in a relative reasonable R^2 (Figure 5-72 E). For the outflow the rule was predicting most of the peak events well with a slight underestimation to predict three peak events, but showed good prediction results for the lower and medium concentrations (Figure 5-72 F).

6 Discussion

6.1 Stormwater Water Quality

The stormwater runoff arriving at the Parafield Stormwater Harvesting Facility is coming from a catchment area of 1600ha. The catchment area is mainly residential area with a small number of parkland areas and industrial area, which is located north of the wetland. The catchment area is serviced with a complex stormwater drainage system for the collection of stormwater runoffs, which is mainly street and roof drainage. The stormwater runoff, which flows through the drainage channel, will pick up a variety of different dissolved and suspended substances in form of nutrients, organic compounds, salts, heavy metals and microorganisms. The research was focusing mainly on the pollutants including nutrients in form of N and P components and DOC.

6.1.1 Physical Water Quality Parameter

The physical water quality parameters measured in form of water temperature, conductivity, pH, DO, ORP and turbidity showed variations due to seasonal changes. Beside the seasonal variations of the physical water quality parameters, spatial differences between the sites were observed as well. The spatial differences can be categorized by comparing non-vegetated (open-water area) and vegetated area; and by comparing sites located near the inflow area (site 1 and 2) with sites located near the outflow area (site 3 and 4).

6.1.1.1 Water Temperature

Water temperature is one of most important and most obvious water quality parameters. It is influenced by solar radiation, which is the major driving force for the overall hydrologic cycle, including thermal stratification in lakes or providing energy essential for evaporation leading to atmospheric moisture creating precipitation (Duever 1988). High solar radiation during the summer seasons will lead to higher water temperature and vice versa. Water temperature is one of the simplest regulating factors, which makes it therefore an important water quality parameter, in consideration of the seasonal denitrification potential, microbial activity and diffusion rate in treatment wetlands (Kamp-Nielsen 1975a; Sutton et al. 1975; Phipps and Crumpton 1994). Water temperature, beside atmospheric pressure, is influencing the DO concentration. The solubility of oxygen is greater in colder water than warmer, and therefore the concentration of DO is greater when the water temperature is lower (APHA 2005). The water temperature between the sites in the reed pond showed some differences between inlet (inflow-site1-site2) and outlet sites (site3-site4-outflow). The average water temperature at the inlet sites was ranging between 15.55-16.25ºC, with a maximum range between 21.39-23.30 ºC in summer and a minimum range between 10.35-10.73 ºC in winter. The average water temperature at the outlet sites was ranging between 14.19-14.73 ºC, with a maximum range between 21.14-23.63 ºC in summer and a minimum range between 8.10-9.04 ºC in winter.

6.1.1.2 Dissolved Oxygen (DO)

Dissolved Oxygen is the most fundamental parameter in freshwater ecosystems, which controls several treatment processes, like oxidation, respiration and nitrification. It is essential to the metabolism of all organisms that possess aerobic respiratory biochemistry. The DO concentration is influenced by the water temperature based after HENRY's law, which states that the solubility of oxygen is decreasing with increasing water temperature (IWA 2000; APHA 2005). The DO concentration in the reed bed pond showed significant differences between the vegetated (2.82-4.31 mg/L) and non-vegetated (5.03-5.73 mg/L) areas.

The explanation for the higher DO concentration at the inflow area is due to the hydrological loading creating turbulence causing an aeration effect. Higher concentration at the outflow area is explained due to photosynthetic activity of algae community in form of *Spirogyra*. Algal processes can raise DO to very high levels during bloom conditions due to photosynthetic production, therefore gases easily supersaturate with respect to their equilibrium solubility (Kadlec and Knights 1996). To achieve this kind of phenomenon an open water areas are required, which is the case of outflow area. Within the vegetated area there was a difference between the sites located near the inflow and outflow. The DO concentrations at site 1 and 2 (2.82-3.42 mg/L) were slightly lower than the DO concentration at site 3 and 4 (3.93-4.31 mg/L). The dense vegetation especially at site 1 and 2, which had significantly higher primary productivities had lower DO concentrations due to the accumulation of organic matter and decomposition leading to an increased oxygen demand (Department of Resources and Energy 1983), which is creating an anaerobic environment especially in the summer seasons.

6.1.1.3 Conductivity

Conductivity overall showed minimal spatial variation, which weren't statistically significant, but the temporal variation showed greater variations. The conductivity value was higher during the summer seasons or non-rainy seasons compared to the rainy seasons. During the summer season the reed bed pond was fed with groundwater (aquifer), which keeps the system inundated for the whole year. The Parafield Stormwater Harvesting Facility is closely located to the sea (Gulf of St. Vincent), which means that the water table is high and would explain higher conductivity levels (avg. 0.374 mS/cm, data provided by the Salisbury City Council). Therefore as soon as aquifer water is supplied to reed bed pond the conductivity will increase, which is about two times higher than the conductivity from the stormwater runoff (avg. 0.226 mS/cm). Salinity comparisons of stormwater has been estimated at a level around 100mg/L, whereas the salinity of groundwater aquifers had a level of around 1000 mg/L (Fisher and Clark 1989).

6.1.1.4 pH

pH reflects the hydrogen ion content of the wetland water. The spatial distribution showed a decline of pH from the inflow towards the outflow area (7.37-6.74), which is ranging within the circumneutral range (pH 6 to 8) (Kadlec and Knights 1996). During the growing seasons higher pH levels were observed linked to higher chlorophyll a concentrations, which is an indicator for algal growth. Algae are known to increase the pH of wetland water during periods of high productivity, especially during blooms where the pH can increase up 8 or 9 (Department of Resources and Energy 1983; Shapiro 1990; Kadlec and Knights 1996; IWA 2000). On the other hand pH will drop due to rain events, causing a dilution effect as well as that the average pH of precipitation is ranging under 6, due atmospheric CO2 dissolving in the droplet forming carbonic acid (Department of Resources and Energy 1983).

6.1.1.5 Redox Potential (ORP)

Redox potential is showing the same trend like the DO. ORP is a measure of the oxidation potential in the water and sediments (Kadlec and Knights 1996). The ORP values showed significant difference, which were higher in the non-vegetated open water area (avg. 59.75 mV) than the vegetated area. (avg. -5.90 mV). Within the vegetated sites the ORP showed statistically significant differences with the sites located near inflow area having a lower ORP value than the ORP at the sites near the outflow (2.8 to 30.48 mV).

6.1.1.6 Turbidity

Turbidity showed temporal and spatial differences. High plant density and accumulation organic matter were responsible for the significant higher turbidity levels at site 1 and 2. Study results from previous research suggested that denser macrophyte community will lower the turbidity and that turbidity has a negative influence on vegetation (Scheffer 1998) The turbidity reading between all the other sites showed no significant differences. Turbidity is influenced by residence time and organic matter content in the water column. The Parafield reed bed pond is a relatively shallow system with a maximum water depth reaching up to 60 cm. Turbidity levels were overall highest during the spring and summer seasons for all collection sites in the reed bed pond. Turbidity is regarded as major problem in Australia resulting in significant costs for water users due the treatment which has aesthetic and health reasons (Department of Resources and Energy 1983).

6.1.2 Chemical and Biological Water Quality Parameter

6.1.2.1 Nitrogen

Nitrogen dynamics involved the measurement in form of TN, NO_3^- and NH_4^+. For all nitrogen components the spatial and temporal variations showed the same trend. All concentrations showed a declining trend from the inflow towards the outflow area, due to nutrient removal performance in form of nutrient uptake by macrophytes and algae, and chemical transformation processes, like denitrification, nitrification and ammonification. The N concentrations from the inflow towards site 1 were increasing. The average concentrations for TN, NO_3^- and NH_4^+ were the highest at site 1. Temporal variations showed that the concentrations were higher during the spring and summer seasons and lower during the autumn and winter seasons.

Nitrogen dynamics are controlled or influenced by sequential processes of ammonification, nitrification and denitrification, volatilisation and plant uptake. Higher nitrogen concentrations during the summer periods could be explained, that higher water temperature stimulate microbial community, which are responsible for the process like ammonification and nitrification (Reddy and Patrick 1984; Moshiri 1993; IWA 2000). Research showed that nitrogen component levels were higher during the summer seasons than winter seasons due to the fact that first in the winter seasons the nitrogen inflow concentrations are lower and therefore having lower ammonification and nitrification rate, which means that less ammonia is available for nitrification (Vymazal 1999a). Most of these processes are influenced by temperature, pH and oxygen (Reddy and Patrick 1984).

6.1.2.2 Phosphorus

Phosphorus dynamics included the measurements of TP and PO_4^{3-}. Like the nitrogen components spatial and temporal variations were noted, but compared to the trends of nitrogen, different trends were observed, especially for the temporal variations. The spatial variations showed the same trend like nitrogen, with a decline of concentrations from the inflow towards the outflow. The average P concentrations were the highest at site 1. Temporal variations of the different sites showed different seasonal trends. At the inflow and outflow sites the P concentrations were high during the winter, spring and summer seasons. Site 1 had high concentrations in the summer and autumn seasons and at site 2 the highest concentrations were observed in the summer seasons. Site 3 and 4 had the same seasonal patterns with having high concentrations during the spring and autumn seasons.

Phosphorus is strongly interacting with wetland soils and biota, which is providing both short-term and sustainable long-term storage of phosphorus (IWA 2000). The adsorption and retention of phosphorus is influenced by soil conditions involving the parameters of soil ORP, pH, minerals and native soil P concentration (Faulkner and Richardson 1989; Richardson and Vaithiyanathan 1995). Phosphorus concentrations were significantly higher during growing seasons (spring and summer) than the non-growing seasons (autumn and winter). Unlike the chemical processes, which are responsible for transforming and removing nitrogen, the processes linked to the transformation and removal of phosphorus weren't temperature-dependent. Temperature has little influence on chemical precipitation and physico-chemical sorption (Vymazal 1999b). At Parafield the measurements has shown higher concentrations of P during the growing seasons than the non-growing seasons. The regulations of phosphorus exchange between the sediment and water is based upon the interaction between iron an phosphorus during aerobic and anaerobic conditions (Kamp-Nielsen 1974, 1975b; Hosomi et al. 1981; Boström et al. 1982; Boström et al. 1988; Holtan et al. 1988; Nürnberg 1988; Jensen and Andersen 1992; Khoshmanesh et al. 1999). The conditions at the sediment-water interface for the vegetated area of the reed bed pond in the growing seasons were more or less anaerobic (DO conc.: 1.3-6.3 mg/L and ORP: -75-57mV), which would associate P release due to the reduction of Fe^{3+} to Fe^{2+}.

6.1.2.3 Dissolved Organic Carbon (DOC)

Dissolved organic carbon dynamics showed that concentrations were decreasing from the inflow towards the outflow. The average DOC concentration was the highest at the inflow and site 2. Seasonal trends showed for the majority of sites that the DOC concentrations were highest during the spring and summer seasons. DOC derives from DOM, which is the largest pool of reactive organic carbon (Hedges 1992; Findlay and Sinsabaugh 2003) providing chemical energy and nutrients (Wetzel 1992). The decline of the DOC concentrations, beside degradation by UV radiation (Strome and Miller 1978; Lindell et al. 1995; Wetzel et al. 1995), is due the utilization of DOM by the microbiota involving a process known as microbial loop, which transforms DOM to POM (Azam et al. 1983; Tranvik 1992). The source of autochthonous DOM are algae and macrophytes (Bertilsson and Jones Jr. 2003). Higher DOC concentrations during the summer seasons are explained by higher microbial activity, which will utilize high levels of DOC, but also create an autochthonous loading due to decomposition. The microbial activity will be influenced by temperature increase, leading to an increase of oxygen demand (Kamp-Nielsen 1975a).

6.1.2.4 Chlorophyll a (chl a)

Chlorophyll a dynamics showed that the concentration at the site 3 was significantly higher than the other sites. Like the chemical parameters there is no clear declining trend from the inflow towards the outflow. Temporal or seasonal variation showed that for most of the sites the levels of chl-a was the highest during spring, whereas inflow and site 1 showed high concentrations in summer and autumn, respectively. The chl-a is an indicator of filamentous green algae of the Genus *Spirogyra*. Observation showed significant relationships between Spirogyra and environmental condition, like trophic status and high loadings of TP, SRP and Org. N (Hainz et al. 2009). The chl-a levels at site 1, 2 and 4 were comparably lower than site 3 and outflow, because of high density of macrophyte community, which causes a shading effect inhibiting the productivity. The macrophyte community at site 3 was harvested at the beginning of 2006 (end of January-begin of February). Due to the harvesting site 3 was transformed into an open water area, which provided an optimal environment for Spirogyra to grow. Due to the harvesting the shading effect by macrophytes, which would reduce phytoplankton productivity resulting in a reduction of algal abundance in vegetated area, was removed (Wetzel 1975).

6.2 Nutrient Removal Performance

CWS designed as a treatment system have become widespread worldwide, which extends to treatment of urban and agricultural stormwater runoffs (Carleton et al. 2001). Research results suggest that the performance of a wetland system in treating stormwater is based on the function of inflow or hydraulic loading rate (HLR) and detention time, which are in turn functions of storm intensity, runoff volume and wetland size (area and volume) (Scherger and Davis 1982; Barten 1987; Meiorin 1989).

Therefore to evaluate the removal performance of the reed bed pond at the Parafield Stormwater Harvesting Facility, first the removal efficiency (in %) and removal rate (mg/m^2/day) for the different nutrients was determined, and second the influence of residence time on the removal performance was determined.

6.2.1 Nitrogen Removal Performance

Nitrogen dynamics in CWS involve many processes in a complex biogeochemical cycle The principle removal mechanisms for nitrogen are ammonification followed by microbial nitrification and denitrification, plant uptake and ammonia volatilization (Watson et al. 1991; Brix 1993) The overall removal efficiency was for TN 24%, NO_3^- 53 % and NH_4^+ 17%, which were within the boundaries compared to other removal performances of CWS receiving stormwater runoff (Table 6-1). The average removal rate was for TN 0.85 mg/m^2/day, NO_3^- 0.79 mg/m^2/day and NH_4^+ 0.28 mg/m^2/day.

The removal performances of individual collection sites, which was calculated to determine different spatial removal behaviour within the reed bed pond, which also helped to determine the autochthonous source or the site(s) causing some critical issues to the performance of the wetland. The removal efficiencies for TN was for site 1 -11%, site 2 13%, site 3 20%, site 4 5% and outflow -5%. The removal efficiency for NO_3^- was for site 1 -4%, site 2 23%, site 3 31%, site 4 20% and outflow 1%. The removal efficiency for NH_4^+ was for site 1 -7%, site 2 17%, site 3 15%, site 4 7% and outflow -18%.

The removal performances of the reed bed pond showed seasonal differences, with removal efficiencies higher during the non-growing seasons (autumn and winter) than during the growing seasons (spring and summer) (Figure 5-7).

The seasonal nutrient removal performances showed that during spring the removal performance for NH_4^+ showed a negative performance, whereas the other N components were positively removed by the system. In the summer season the removal of NO_3^- showed negative efficiency, with low removal of NH_4^+. The N removal efficiencies during the autumn and winter seasons showed a positive performance. Comparing the seasonal performances the efficiencies of removal for N components were greater during the non-growing seasons.

The removal mechanisms involving N components are related mostly to microbial supported chemical transformations, which are therefore influenced by the factors of water temperature, pH and oxygen (Reddy and Patrick 1984; Vymazal 1995). The lower removal performances regards NH_4^+ during the spring period could have been caused due to greater ammonification rate than nitrification rate. Increasing water temperature lead to an increase of the microbial activity, which will increase the ammonification rate, which is the conversion of organic N to NH_4^+, which is the preferred chemical composition to be utilized by the macrophyte and algae community (Reddy and Patrick 1984; IWA 2000). During spring and summer seasons at the Parafield reed pond the DO concentrations were low, ranging between 1.3 – 6.3 mg/L, which therefore would inhibit the ammonification process and as the result of this NH_4^+ is not microbially converted to nitrate (Kadlec and Knights 1996). Ammonification can occur under aerobic and anaerobic conditions, but the decomposition under anaerobic conditions is slower and therefore a higher concentration of N in for of NH_4^+ will be available, because nitrification is prevented. Due to the anaerobic condition the N loss through the nitrification-denitrification pathway will enhance by increasing denitrification rates, which will explain the higher removal performances of NO_3^- (Patrick and Reddy 1976; Patrick 1982; Reddy et al. 1989; Busnardo et al. 1992; Thompson et al. 1995; Bachand and Horne 2000a, b)..

6.2.2 Phosphorus Removal Performance

Phosphorus removal in wetlands occurs from the following processes, which is based on soil sorption (adsorption-precipitation reactions with aluminium, iron, calcium and clay minerals in the soil) and plant uptake. Another removal process is the production of phosphine (Watson et al. 1991; Brix 1993). The overall removal efficiency was for TP 34% and for PO_4^{3-} 47%, which comparably lower than the removal performances of other stormwater treatment systems (Table 6-1). The average removal rate for TP was 0.05 mg/m^2/day and for PO_4^{3-} it was 0.03 mg/m^2/day.

Figure 5-10 is showing the removal performance regards P components for the different seasons. Unlike to removal performance of N components the removal performance of P seemed to be relatively constant, with the removal during the summer and autumn seasons to be more efficient than during the spring and winter seasons.

The removal efficiencies for the individual collection sites were for TP for site 1 -54%, site 2 19%, site 3 24%, site 4 26% and outflow 6%. The removal efficiency for PO_4^{3-} was for site 1 -70%, site 2 27%, site 3 29%, site 4 34% and outflow 8%. Similar to N the removal performance for P the removal efficiency was negative at site 1, which would be caused by high plant biomass being an autochthonous source for P due to decomposition and P release from the sediment caused by the anaerobic condition, which is indicated by low DO concentration and negative ORP values. Lower DO concentrations will cause a lower ORP value, which would indicate anaerobic conditions during the summer seasons. Phosphorus typically forms insoluble complexes with oxidized iron, calcium and aluminium under aerobic condition, but under anaerobic conditions, much of the bound phosphorus becomes soluble and therefore more available for plant uptake and diffusion towards the water column (Hosomi et al. 1981; Boström et al. 1982; Boström et al. 1988; Kadlec and Knights 1996; Ann et al. 2000). Looking at the seasonal removal performance of PO_4^{3-} it can be seen, that the during the winter seasons the removal efficiency is the lowest, with higher removal performances during spring, summer and autumn, which would relate to the growing period of the macrophyte and phytoplankton communities, therefore higher uptake rates would result in a higher removal performance (Watson et al. 1991; Brix 1994a; Ennabili et al. 1998). The removal process during the winter process would mostly rely on the adsorption and precipitation reactions with the sediment The relative constant removal performances of reed bed pond is also replying to the fact, that the removal mechanisms in form of soil sorption, involving adsorption-precipitation reaction, and plant uptake are not temperature-dependent (Kadlec and Knights 1996; Brix 1998; Vymazal 1999b).

Table 6-1: Nutrient removal performances of constructed wetlands receiving stormwater runoff

Location	Type[a]	Drainage[b]	Nutrient Removal Efficiency (%)					Reference
			TN	NO_3^-	NH_3	TP	PO_4^{3-}	
Bellevue 3I (B3I), USA	C	U				**8**		(Reinelt and Horner 1995)
Crestwood, USA	C	U	**22**	**40**	**55**	**46**	**36**	(Carleton et al. 2000)
Crookes, Australia	C	A	**11**			**17**		(Raisin et al. 1997)
Des Plains EWA 3, USA	C	U, A		**81**		**66**		(Hey et al. 1994; Mitsch et al. 1995)
Des Plains EWA 4, USA	C	U, A		**50**		**88**		(Hey et al. 1994; Mitsch et al. 1995)
Des Plains EWA 5,USA	C	U, A		**82**		**82**		(Hey et al. 1994; Mitsch et al. 1995)
Des Plains EWA 6,USA	C	U, A		**99**		**89**		(Hey et al. 1994; Mitsch et al. 1995)
Emerald Square Mall, USA	C	U	**40-70**			**60-85**		(Daukas et al. 1991)
Greenwood, USA	C	U	**-11**	**-13**	**10**	**62**	**77**	(McCann and Olson 1994)
Hidden River, USA	N	U	**46**	**94**	**79**	**70**	**67**	(Carr 1995; Carr and Rushton 1995; Rushton 1996)
Lake Putrajaya, Malaysia	C	U, A	**82**	**70**			**84**	(Sim et al. 2008)
Patterson Creek (PC12), USA	C	U				**82**		(Reinelt and Horner 1995)
Queen Anne, USA	C	U	**23**	**55**	**56**	**39**	**69**	(Athanas and Stevenson 1991)
Swift Run, USA	N	U				**49**		(Scherger and Davis 1982)
CWS,USA	C	U	**25**			**45**		(Schueler 1987)
Parafield, Australia	C	U	**24**	**53**	**17***	**34**	**47**	This study

[a]C = constructed, N = natural [b]U = urban, A = agricultural *NH_4^+ removal efficiency

6.2.3 DOC Removal Performance

DOC removal mechanisms would involve utilization by microbiota (Pinney et al. 2000) degradation by UV radiation which is called photochemical degradation (Strome and Miller 1978). This degradation is often accompanied by changes in the optical properties of DOC, called photo bleaching, which is known to reduce molecular weight of the humic substances (Strome and Miller 1978; Lindell et al. 1995; Wetzel et al. 1995) and produce inorganic carbon (Salonen and Vähätalo 1994). The overall removal efficiency for DOC was 11% and the average removal rate was 5.6 mg/m^2/day. The removal performance was slightly higher than the minimal removal efficiency (Table 6-2).

Table 6-2: DOC removal performance for constructed wetlands receiving stormwater runoff

Location	Type[a]	Drainage[b]	RE (%)	Reference
Kingman, USA	C	M	**9-47**	(Pinney et al. 2000)
Hidden River, USA	N	U	**9**	(Carr 1995; Carr and Rushton 1995; Rushton 1996)
Parafield, Australia	C	U	**11**	This study

[a]C = constructed, N = natural [b]U = urban, M = municipal RE = Removal Efficiency

The removal efficiencies for the individual collection sites were for site 1 3%, site -2%, site 3 3%, site 4 7% and outflow 0.3%. The comparison of the seasonal DOC removal efficiencies (Figure 5-13) showed that the removal performances were highest during non-growing seasons, whereas the removal performances during the growing seasons were minimal or close to non-existent.

DOC is produced due to the incomplete mineralization of particulate organic carbon caused by bacterial decomposition of plant detritus (Moran and Hodson 1994). During the autumn and winter seasons the mineralization process is slower, which therefore favours degradation, photolytic decomposition or adsorption processes (Koelmans and Prevo 2003), resulting in an increased removal performance. The autumn and winter seasons in the Adelaide region is also the start of the rainy season, resulting in many rain events and therefore an increased amount of precipitation, which is considered to major allochthonous source of DOC. Therefore the volume of through fall, which is caused by stormwater runoff, will increase. As precipitation moves through vegetation, it is enriched with DOC and DOM, but considering that the catchment area supplying the Parafield Stormwater Harvesting Facility with stormwater runoff is dominated by domestic use, high concentrations of DOC is not expected. But precipitation and through fall are the major allochthonous source for DOC and DOM (Aitkenhead-Peterson et al. 2003). The low removal performance during the spring and summer seasons is due to high amount of autochthonous loadings caused by decomposition of plant material. DOC also can leach into the water flowing through wetlands as plants, algae and bacteria are growing, dying and decaying (Wetzel and Manny 1972a; Wetzel and Penhale 1979; Breen 1990; O'Connell et al. 2000). Sporadic rain events during the spring and summer seasons cause therefore lower loadings due to precipitation and through fall. Removal processes like the utilization of DOC by the microbial community and photochemical degradation by UV radiation will increase, but these removal mechanisms will be counterbalanced by the increase of turbulences like resuspension, which will increase the turbidity resulting in an increased rate of mineralization. This is the cause of higher DOC concentration (Koelmans and Prevo 2003)

6.2.4 Relationship between Removal Performance and Residence Time

Residence time (RT) is an important factor which is influencing the removal performances of treatment wetlands. Beside residence time other study results showed that the removal performance was influenced by inflow or hydraulic loading rate (HLR), type of wastewater, climate (Scherger and Davis 1982; Barten 1987; Meiorin 1989; USEPA 2000; Carleton et al. 2001). The Parafield Stormwater Harvesting Facility showed seasonal differences in residence time, which weren't statistically different (Figure 5-15), but the average residence time during the spring season (RT = 8.7 days) was the lower than the other seasons. For the removal of different nutrient a different flow regime is required. In general most the existing wetlands are designed to have a wide range of RT ranging from 2 to 20 days (Kadlec and Knights 1996; Pinney et al. 2000). Longer RT is generally applied to allow aeration via diffusion from the atmosphere for BOD removal and nitrification, whereas shorter RT is applied when the wetland is receiving higher quality treated wastewater (e.g. denitrification) and wetlands are used for other design objective (e.g. habitat enhancement) (Pinney et al. 2000). The average removal performances of the different nutrients at different flow regimes can be seen in Figure 6-1.

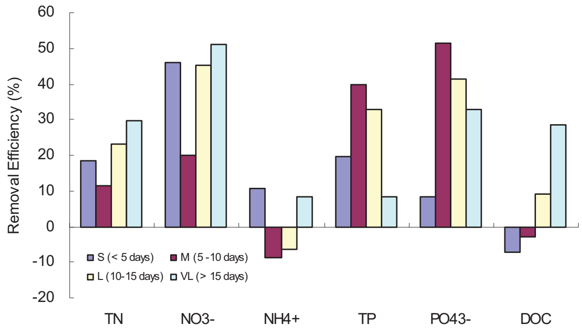


Figure 6-1: Overall nutrient removal performances of the reed bed system

The removal performances for the different nutrient components showed that the removal performances for TN were positive for all flow regimes, but the removal performance showed the highest efficiency at ERT conditions, which gives the system enough to retain the different N components. The removal performance for NO_3^- showed overall positive performances for all residence time categories, with the highest performance at ERT condition, but the removal performance was showing high performances under SRT and LRT conditions as well. The performance for NH_4^+ showed that the removal efficiency was most efficient under SRT and ERT conditions. Other study results suggested that the removal performances for NO_3^- was most effective under short residence time conditions, whereas NH_4^+ was most efficient under longer residence time conditions (Pinney et al. 2000). The removal performance for P components showed the highest efficiency at MRT conditions, whereas the removal efficiencies were decreasing towards shorter and longer residence time, which is also supported by other research results, which mentioned that the P removal efficiency was highest with a residence time of 14 days and reduced performance at shorter residence time (Reinelt and Horner 1995; Rushton et al. 1995). The removal performance for DOC concentration results showed to be most efficient at ERT conditions or in general are more effective with increase of residence time. Research has shown that in systems with long residence time the concentration of DOC is greater than in systems with shorter residence time, because wetlands with shorter RT will experience less DOC leaching from plant material, compared to longer RT designs. Many wetlands are currently designed for nitrogen removal. Short RT can be employed for nitrate removal, while longer RT are required for removing ammonium, since the later requires oxidation of ammonia to nitrate prior to removal as nitrogen gas. Therefore , the greatest impact of DOC leaching will likely be experienced in constructed wetland systems with longer RTs, such as those used when ammonia removal is the key design criteria (Pinney et al. 2000). The results at Parafield showed to be opposite to the previous statement.

Since the relationship between the removal performance and residence time didn't show a clear pattern the assumption was made that the removal performance could have been influenced by the nutrient concentrations at the inflow area, which were visualized in Figure 5-17, 5-18 and 5-19. The relationship between the nutrient concentrations at the inflow removal performance of N components seemed to have a positive effect on the removal efficiency, which means in other words that higher nutrient concentrations at the inflow corresponded with an increased removal performance and vice versa. This pattern was also observed for the P components, whereas for DOC the concentrations at the inflow didn't seemed to correlate with the removal performance. Loading rates are considered to be an important parameter for empirical design and operation. A higher loading rate is desirable for increasing microbial production and respiration, while severe or prolonged overloading will inhibit bacterial activity (Tao et al. 2006), which can give a positive impact to removal efficiencies, but on the other hand it can also lead to degradation of water quality (Turner et al. 1977).

6.2.5 Allochthonous and Autochthonous Sources

The individual removal performance of the different collection sites made it possible to identify the sites, which showed negative removal performances and rates (Figure 6-2). Based on these data the autochthonous source within the reed bed system was determined. The nature of the system functioning as a treatment wetland means, that the majority of loadings to the system is allochthonous, but additionally to the external loading a certain amount autochthonous loadings were expected. The results showed that for N and P components, that site 1 was the only site to have negative efficiencies and rates, which is an indicator for internal loadings. The internal loadings for whole study period was for TN 0.39 mg/m^2/day, NO_3^- 0.07 mg/m^2/day and NH_4^+ 0.12 mg/m^2/day. For TP the internal loadings were 0.08 mg/m^2/day and for PO_4^{3-} it was 0.06 mg/m^2/day. In case of N site 1 wasn't the only source for autochthonous loadings. The outflow site provided internal loadings for TN (0.13 mg/m^2/day) and NH_4^+ (0.21 mg/m^2/day) as well.

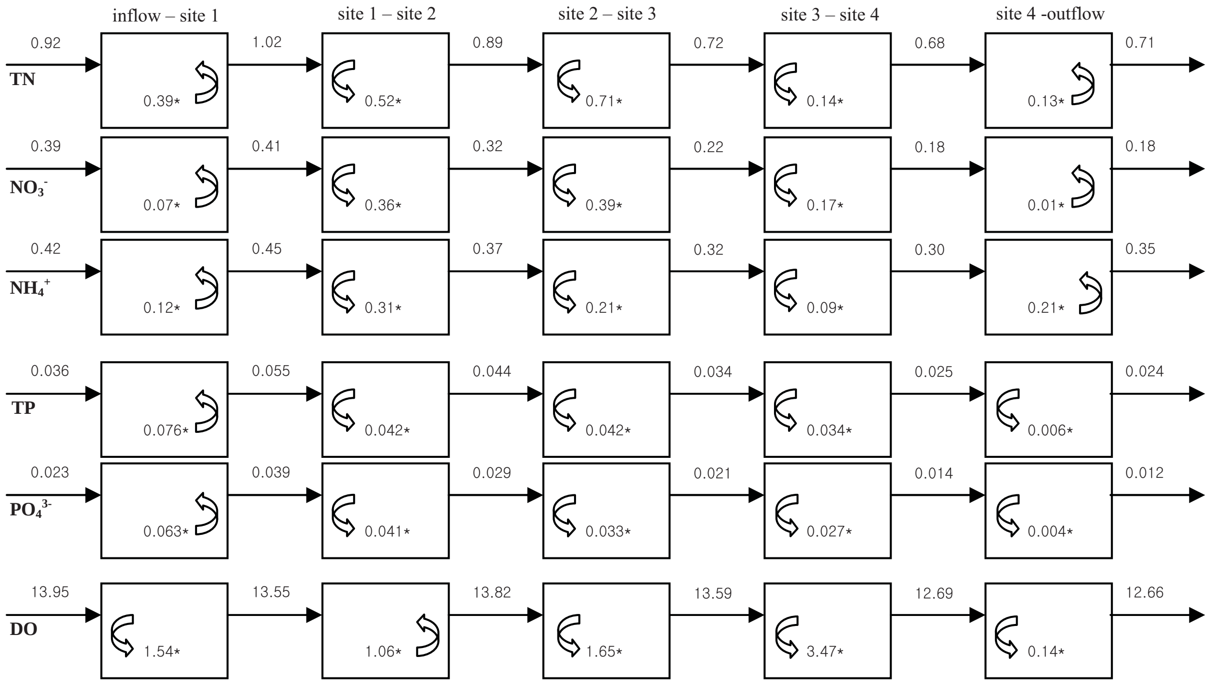


Figure 6-2: Nutrient retention capacity for the different forms of nutrients at the collection sites (mg/L). *mg/m^2/day

The DOC the autochthonous source was site 1, where over the whole study period the amount of internal loading was 1.06 mg/m^2/day.

Overall the autochthonous source was for the majority of the measured nutrients site 1, which is the site 1 next to the inflow area. The plant productivity in form of biomass was the highest at site 1 than the rest of the sites. The conditions in the water column showed high turbidity reading and low DO concentrations. In case of N components the major internal source is due the decomposition of plant material, where org. N is transformed into inorg. N due to the process called ammonification (mineralization), which is driven by microorganisms. Higher turbidity and low DO concentration, especially the low DO concentrations will create anaerobic environment near sediment and water interface, which leads to release of phosphorus form the sediment. All these processes occur simultaneously and counterbalance each other, but site showed that the counterbalance of the internal production is greater than the removal of the external loadings.

6.2.6 Overall Removal Performance

The overall removal performance for the different nutrients showed that the reed bed pond is performing at a medium-low efficiency compared to previous studies (Table 6-1, 6-2). Seasonal changes of the residence time showed no statistical significant differences, but differences of RT between the seasons existed and showed influences on the removal performances. Removal performances for N components showed higher efficiencies during the non-growing season than the growing seasons. The same trend was observed for removal performance of DOC. The removal performance for the P components showed higher efficiencies during summer and autumn, with lower removal efficiencies during winter.

6.3 Macrophytes

Macrophytes is one of the distinguishable characteristics of wetland ecosystems in comparison to other ecosystems beside the existence of water, which makes it an indispensable component (USEPA 1987; Mitsch and Gosselink 2000). Many research publications concluded that the function or contribution of macrophytes in constructed as well as natural wetlands is essential and their management, which also contributes to the performance of the system and is therefore sustainable.

Like previous mentioned the composition of macrophytes in the reed bed pond was the following: Site 1 was consisting of the plant species *E. sphacelata*, *S. validus* and *P. australis*, with *P. australis* being the dominant species. *T. orientalis* was present as well. The results of the macrophyte survey on a monthly bases showed that the average dry weight per m^2 was highest at site 1 (avg.: 3768.2g, min.: 1979.1g and max.: 6622.4g) followed by site 2 (avg.: 3083.9g, min.: 1493.8g and max.: 4789.0g) and site 4 (avg.: 2291.8g, min.: 1366.4g and max.: 3518.2g). Site 3 (avg.: 1282.2g, min.: 644.4g and max.: 1971.0) had the lowest dry weight. For the measurement for all biological and chemical parameters from the different collection sites only the data of the dominant species were used. At site 3 *T. orientalis* appeared to be the dominant species based on the biomass measurement, but *P. australis* was chosen, because *T. orientalis* initially wasn't planted. The biomass production in form of dry weight (g/m^2) compared to other studies showed (Table 6-3), that the primary productivity of the reed bed pond was relatively high, which also correspond with the nutrient concentrations and storage. The N:P ratios within the macrophyte community indicated that the dominant species at the different collection sites were P limited, which had an average N:P ratio of 5.0 for site 1, 5.2 for site 2, 8.4 for site 3 and 6.2 for site 4. At N:P ratios >16, community biomass production is P limited. At N:P ratios <14, community biomass production is N limited, and N:P ratios are between 14 and 16, community biomass production is co-limited by N and P (Koerselman and Meuleman 1996).

Table 6-3: Comparison of above ground biomass (g/m^2), nutrient concentration (mg/g) and storage (g/m^2)

Species	DW g/m^2	N mg/g	P mg/g	N:P	N g/m^2	P g/m^2	Reference
P. australis	**4992**	**4.7-21.6**	**0.4-3.1**	**7:1-12:1**	**5.6-61.8**	**0.5-6.2**	
T. domingensis	**3906**	**9.8-22.1**	**1.4-3.7**	**6:1-7:1**	**0.5-53.0**	**0.5-7.5**	(Bald 2001)
P. australis	**788**	**23**	**1.8**	**13:1**	**18.1**	**1.4**	
B. articulata	**2139**	**11.8**	**1.5**	**8:1**	**25.2**	**3.2**	(Adcock and Ganf 1994)
P. australis	**2296**	**19.4**	**1.7**	**11.5:1**	**42**	**37**	
T. angustifolia	**2158**	**10.4**	**1.5**	**7:1**	**25.4**	**2.8**	(Ennabili et al. 1998)
P. australis		**20.4**	**2.0**	**10:1**			
T. domingensis	**1120**	**15.8**	**2.0**	**8:1**	**12**	**2.24**	(Greenway and Woolley 1999)
P. australis	**1600**	**32**	**3.0**	**10.5:1**	**55**	**4.5**	(Tanner 1996)
T. orientalis		**6.1-12.2**	**1.6-2.1**	**4:1-6:1**			(Breen 1990)
P. australis		**9.0-29**	**1.2-3.3**	**6.5:1-9:1**			(Hocking 1989)
P. australis (Site 1)	**1979-6622**	**7.9-34.2**	**1.3-10.9**	**2.9:1-8:1**	**16.9-226.5**	**2.9-60.6**	
P. australis (Site 2)	**1494-4789**	**8.1-31.8**	**1.4-12.9**	**1.5:1-13.2:1**	**12.8-142.7**	**2.1-56.7**	
P. australis (Site 3)	**644-1971**	**5.8-21.2**	**0.3-5.5**	**3.9:1-23:1**	**3.8-40**	**0.2-8.4**	
B. articulata (Site 4)	**1366-3518**	**7.8-18.1**	**0.6-6.2**	**2.7:1-13.7:1**	**10.6-63.8**	**0.8-19.2**	This study

6.3.1 Macrophyte contribution and influence on nutrient removal

Macrophytes growing in constructed treatment wetlands have a variety of functions contributing to the treatment processes and therefore they are an essential component to be considered for the design. The major role of macrophytes in constructed treatment wetlands is summarized in Table 6-4. The most important effects of the macrophytes in relation to the wastewater treatment processes are the physical effects, such as erosion control, filtration effect and the provision of surface area for attached microorganisms. The presence of vegetation distributes and decreases the flow velocity of water (Pettecrew and Kalff 1992), which is creating a better environment for sedimentation of SS, decreases the risk of erosion and resuspension and therefore increases the contact time between water and the plant surface. The plant surface in form of stems and leaves is providing a huge area for microbial communities or biofilms (Gumbricht 1993b, a; Chappell and Goulder 1994).

The metabolism of the macrophytes, such as plant uptake and oxygen release affects the treatment processes to different extends depending on design. Rooted aquatic macrophytes are creating small aerobic rhizosphere and the importance is, that oxygen is released into the surrounding sediment influencing the chemical composition of interstitial water and therefore affect substrate utilization rates by bacteria, metabolic pathways and nutrient availability (Mann and Wetzel 2000). Macrophyte community also require oxygen, which is normally translocated through the aerenchyma from the stems to the roots. Surplus of oxygen is released from the small roots into the surroundings, which is quickly consumed in the decrease of local oxygen demand (Brix 1994a).

Biological functioning is very important, which is particularly represented by the macrophyte community in form of nutrient sink capacity in form of the processes of nutrient storage and removal within the wetland system (Bald 2001). The nutrient uptake by macrophytes occurs mainly during the growing season, in which the nutrient retention capacity of emergent macrophytes and the amount that can be removed if the biomass is harvested is ranging between 30-150kgPha^{-1}yr^{-1} and 200-2500 kg ha^{-1}yr^{-1} (Brix and Schierup 1989a; Gumbricht 1993b, a; Ennabili et al. 1998). Therefore nutrient uptake will have a positive impact on the removal performance of a treatment wetland, but this can be significantly reduced or suspended during periods of dormancy. Nutrients stored within the plant tissue can be an autochthonous source re-entering the wetland system upon senescence, which is prior to winter. Many rooted macrophytes translocate nutrients from the above ground tissue to below ground storage organs prior to senescence (Davis 1991). Large quantities of labile nutrients are lost from leaves via translocation of nutrients prior to shoot death and leaching from senesced leaves (Davis and van der Valk 1978), with N and P concentrations to be higher in the leaf tissues than those in the stem (Suzuki et al. 1989). However, the amount of nutrients that could be removed due to harvesting is generally insignificant in comparison with the loading into the CWS (Brix 1994b), but if the macrophyte community within wetlands are not harvested, an enormous majority of the nutrients, which have been incorporated into the plant tissues will be returning to the water column due to decomposition processes. The re-entering of nutrient stored in plant tissue would have a negative influence on the removal performance of a treatment wetland and would over a long term influence the storage capacity of the treatment wetland (Kadlec and Knights 1996).

Table 6-4: Summary of the major roles of macrophytes in constructed treatment wetlands (Brix 1997)

Macrophyte property	Role in treatment process
Plant tissue (air)	• Light attenuation → reduced growth of phytoplankton • Influence on microclimate → insulation during winter • Reduced wind velocity → reduced risk of resuspension • Aesthetic pleasing appearance of system • Storage of nutrients
Plant tissue (water)	• Filtering effect → filter out large debris • Reduce current velocity → increase rate of sedimentation, reduces risk of resuspension • Provide surface area for attached biofilms • Excretion of photosynthetic oxygen → increase aerobic degradation
Roots and rhizomes (sediment)	• Prevents the medium from clogging in vertical flow systems • Release of oxygen in crease degradation (and nitrification) • Uptake of nutrients • Release of antibiotics

6.3.2 Macrophyte Harvesting

Harvesting is considered to be one of many management tools. Nutrient loadings are retained by a treatment wetland due to accumulation in the sediment, release to the atmosphere (nitrogen), or taken up by aquatic macrophytes. Macrophytes have a relatively high absorption rate of nutrients (Whigham et al. 1978; Lieffers 1983; Kim 1989; Ennabili et al. 1998). The harvesting of plants generally does not result in removal of a large quantity of chemicals, for examples nutrients; from the system unless plants like *P. australis* are harvested several times during a growing season.

The plant harvesting in the reed bed pond at the Parafield Stormwater Harvesting Facility, which was performed at the beginning of 2006 (end of January to begin of February), which was towards the end of the growing period with the macrophyte community have the greatest biomass and therefore high plant tissue concentrations and nutrient storage. The location where the harvesting was performed included a total area of 400 m^2 including collection site 3 and all biomass and organic materials above the sediment level was removed by using a harvesting boat.

Using the dry weight and nutrient data from the previously monitored date, which was November 2005, the estimated removal of biomass was around 610 kg, with a nutrient storage for TN of 13.0 kg and for TP of 3.4 kg. This is just an estimate, which would be higher at the harvesting period. The problem was, that at the following spring the macrophyte community didn't recover or regrow, instead an increase of chl-a, indicating the algae community in form of *Spirogyra sp.*, was observed at the harvested area. The chl-a were significantly higher after harvesting, which would explain that the growth of Spirogyra was inhibited by the dense macrophyte community due to shading (Mitchell 1989). Factors having a possible impact on the regrowth of the macrophyte community can be the following negative impact due to high turbidity, low oxygen condition in the root zone, flood stress in regards water level too high and competition in particular case it is *Spirogyra* (Scheffer 1998; IWA 2000).

The storage capacity of plants is determined by the total biomass and tissue nutrient concentration of that biomass. The interaction between these parameters determines the timing for optimum harvest potential. Haberl and Perfler found that a harvest in summer would result in three to five times greater removal of nitrogen and phosphorus than a harvest during the autumn and winter after the commencement of senescence (Suzuki et al. 1989; Haberl and Perfler 1990). Research results suggested that plant harvesting doesn't results in a great removal of nutrients, unless plants are harvested several times during the growing season, which is achieving the maximum removal performance of nutrients at the peak and at the end of plant growth (Suzuki et al. 1989). Effective harvesting will stimulate plant growth and therefore a greater amount of nutrients can be removed

Other studies supported a higher removal of nutrients by using a multiple harvesting regime .Above ground material may be replaced up to one to two times each year

(McComb et al. 1989), therefore the frequency of harvests within a season may also be important for overall nutrient removal. Suzuki conducted harvest regimes which removed above ground material multiple times throughout a single growing season, and therefore maintained tissue in a younger, more nutrient concentrated state. The success rate of harvesting is not only depending at the time of maximum nutrient storage, but also at times of maximum allocation to harvestable portion (Asaeda et al. 2000).

Study at the Willunga wetland system regards harvesting showed that the multiple harvesting of different macrophyte species showed a significant increase of nutrient removal for the macrophyte species of *Bolboschoenus* and *Typha* at different harvesting intervals, with the efficient harvesting regime performing a harvest in late summer (February) and in late spring (November), but the results showed also that the harvesting regime for *Phragmites* failed to achieve the anticipated removal (Bald 2001).

The timing and frequency to use harvesting as a management tool is essential, so the harvests should be performed before nutrients are transformed to a soluble form due to plant decomposition or translocation to below ground storage (Gearheart 1992). High growth rates of macrophyte community could be useful in nutrient removal, however with high decomposition rates, harvesting were essential in order to prevent the re-entering of the plant tissue nutrients to the system.

Based on the monitoring results of the dry weight and retention of the different nutrients for the reed bed pond at the Parafield Stormwater Harvesting Facility (Figure 5-26) the best timing to perform plant harvesting, based on the maximum nutrient concentration in the plant tissue, was for the majority of the collection sites between February and March. The maximum amount of the different nutrient concentrations, which would have been harvested during that period, was for site 1 174.1g/m^2 for TN and 47.11g/m^2 for TP. For site 2 it is 118.21g/m^2 for TN and 29.41g/m^2 and for site 4 it is 52.11g/m^2 for TN and 14.71g/m^2 for TP. For site 3 the amount of data was inefficient to make a decision, before after the harvesting the macrophyte community didn't recover, so that the impact of the harvesting regards the nutrient removal wasn't be able to be determined.

6.4 Sediment

The sediment in the reed of the Parafield Stormwater Harvesting Facility is considered to be mineral soil (pH ranging 6.6 – 8.8) (Mitsch and Gosselink 2000). The focus of the monitoring was to determine the functions of the sediment inside the reed bed.

The general function of the sediment in the reed bed pond of the Parafield Stormwater Harvesting Facility is depending on the changes of the seasons. The results for the different nutrient concentration showed that the nutrient dynamics were changing over the different seasons. In general the nutrient concentrations were lower during the growing seasons (spring and summer season) and were significantly higher during the non-growing seasons (autumn and winter) (Figure 6-10). For TC, TN and TP of the sediments the concentrations were the highest during the autumn season followed by winter, and spring and autumn. This means that the concentrations during spring and summer were lower, which indicates that during the spring and summer seasons the sediment is acting as source pool of nutrients and as sink during the autumn and winter seasons. During the growing seasons nutrients in the sediment pool will be utilized by the uptake of emergent macrophytes and released to the water column.

Phosphorus concentration was measured in form of TP showed that the concentrations of sediment of the P component were significantly lower during spring and summer than autumn and winter. During the growing seasons the sediment will function as a source for P, due to the uptake of PO_4^{3-} by the macrophyte community and release from the sediment into the water column caused by anaerobic condition. Phosphorus mobilization as well as phosphorus fixation is affected by different environmental parameters of which temperature, pH and redox potential are the most important (Boström et al. 1988). Increase of water temperature is linked to a higher microbial activity affecting the physical and chemical mobilization processes as well as transport mechanisms. The increase of microbial activity leads to an increase in oxygen consumption and decrease of redox potential. At redox potential levels below 200 mV, part of the iron (III) pool in the surface sediments is reduced to iron (II) and therefore P is released from the sediment into the water column (Hosomi et al. 1981; Boström et al. 1982; Boström et al. 1988; Kairesalo and Matilainen 1994). During the growing seasons the water temperature is increasing, and the comparison of DO and ORP levels between the seasons showed significantly lower levels during the growing seasons, and the P concentrations in the sediment were significantly lower than the concentrations during the non-growing seasons. Additionally to P release from the sediment is the uptake of P by the macrophyte community. The growth rate of the macrophyte community, which is measured in form of biomass, showed an increasing trend towards summer, which is linked to an increased uptake rate of nutrients. Therefore the sediment is acting as source during growing seasons due to physical and chemical mobilization and desorption processes. TP concentrations were higher during the non-growing seasons for all sites in the reed bed pond, caused due to the decomposition or senescence of the macrophyte community leaching P back into the water column (Barko and Smart 1979); but lower water temperature leading to a decrease in microbial activity, meaning decreased oxygen consumption resulting in higher DO concentrations and higher OPR levels. Under this conditions adsorption or precipitation of P with inorganic compounds and P co-precipitation with iron is favoured and therefore the sediment is functioning as a sink of P (Boström et al. 1988).

Like the trend for sediment P, the same trends were observed for sediment N and C. For sediment N measured in form of TN, the concentrations were significantly lower during the growing seasons. Unlike to P, where the sediment is playing an important role in releasing iron-bound P to the water column, N forms are generally not released in great amounts from the sediment. Upwards diffusion is transporting NH_4-N from the sediment surface to the overlying water column. N forms are changing their chemical properties due to processes like nitrification, denitrification and ammonification. The reed bed pond is kept inundated over whole year, in which the sediment characteristic is defined by the absence of oxygen. During the growing seasons the sediment is acting as a N source providing the nutrients for the macrophyte community. Additionally to the uptake by the macrophyte community, there is another process causing a loss of N from the sediment. During summer anaerobic conditions will favour the microbial-driven denitrification process, which will transform NO_3^- to N_2 and N_2O and therefore N is lost by the system (Patrick and Tusneem 1972; Patrick 1982; Reddy and Patrick 1984; Reddy et al. 1989). During the non-growing seasons N concentration in the sediment is higher due to leaching of nutrients by senescence of macrophytes, which is the internal source, whereas the external source is in form of stormwater runoff.

The trend for sediment C was the same like for N and P. C concentration in the sediments were lower during the growing seasons than during the non-growing seasons. Aquatic or wetland macrophytes are the most productive plant communities (Wetzel

1983) and comprise a significant fraction of the total organic inputs to wetland and other aquatic ecosystems. The most common source of C is produced by deposition from the atmosphere in form of precipitation (Aitkenhead-Peterson et al. 2003). The climate condition in Adelaide has majority of the annual precipitation during the non-growing seasons. Therefore the C concentration in the sediment is higher during the autumn and winter seasons. Additionally to the external loadings there is an internal loading due to senescence of the macrophyte community. During the summer seasons C sources are utilized by microfauna, which is very important in the wetland carbon cycle, because most of the organic carbon fixed in wetland systems by both phytoplankton and macrophytes is processed and utilized by bacteria without involving the food web of higher animals (Wetzel 1984). Overall the annual comparison of the function of sediment regards the nutrient retention (Figure 6-3) showed that the sediment had the function as source of nutrients. Only in year 2006 the sediment showed an accumulation of nutrients, indicating that sediment was functioning as a sink. But the overall function of the sediment was to be an internal source of nutrients for the reed bed system, which didn’t showed an accumulation of nutrients.

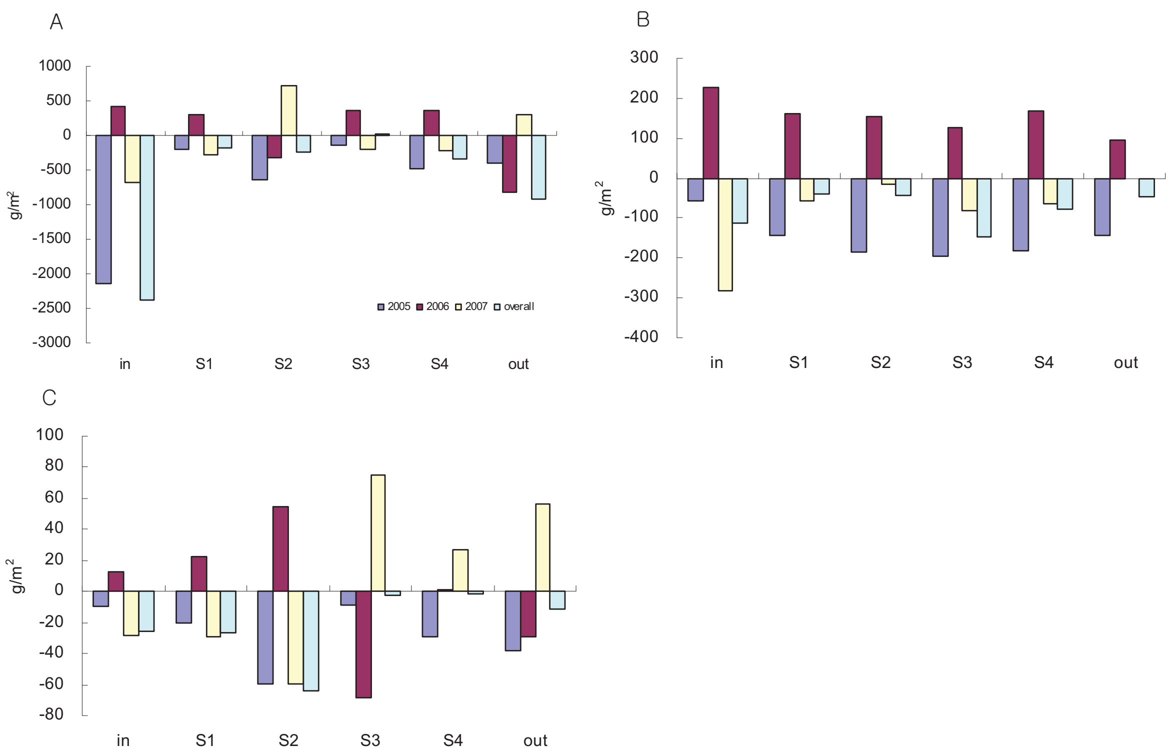


Figure 6-3: Nutrients retention capacity (g/m^2) of the sediment in the reed bed pond. A. TC, B. TN and C. TP (+ bars indicate retention = sink and - bars indicate release of nutrients = source)

6.5 HEA

The HEA was used for to determine the driving parameter, which are influencing the output parameter in form of the different nutrients and determine a rule-set to predict and calculate the different forms of nutrients. By finding the relationship of the driving parameter and output parameter and their influences could provide implications for water quality control by controlling the driving parameters.

To determine these relationships of the different nutrients, a data set ranging from March 2005 – August 2007 was used. The monitoring data for the different components had different time intervals, therefore the sediment and plant data were interpolated to a weekly interval matching the data interval of water data.

6.5.1 HEA for N components

6.5.1.1 Prediction and forecasting of outflow N concentration

The predictions of the N concentrations at the outflow by using the selected measured input variables showed that the HEA was able to develop prediction models with certain accuracy. The input variables chosen for model 1 were the measured physical parameter including N component conc. at the inflow. For model 2 only the physical parameters were selected. The prediction accuracy for model 1 was for TN an R^2 = 0.77, for NO_3^- R^2 = 0.48 and NH_4^+ R^2 = 0.70. For model 2 the prediction accuracy was lower for TN R^2 = .041, NO_3^- R^2 = 0.48 and NH_4^+ R^2 = 0.39. Model 1 showed the possibility of more accurate predictions, but results of model 2 would more realistic in regards the prediction of a chemical output by using only the physical variables. This is also the reason why the input variable setup was chosen to forecast the N components. The forecasting models for the different N components developed by the HEA showed that the models were able to predict most of the major peak events. The prediction accuracies for TN was R^2 = 0.39, for NO_3^- R^2 = 0.38 and NH_4^+ R^2 = 0.36. The lower prediction accuracy was due to fact of missing peak events and underestimating most of the lower and medium range concentrations, but considering the nature of the dataset the models produced by HEA for the forecasting of N component concentration was surprisingly accurate.

6.5.1.2 Knowledge discovery and generic rules for the prediction of N for the different sites

The predictions for the N components as output variables were performed for TN, NO_3^- and NH_4^+. The predictions were performed for different sections within the reed bed pond of the Parafield Stormwater Harvesting Facility, which inflow, site1-site2, site3-site4 and outflow. The prediction accuracy for the different sections and N forms can be seen in Table 6-5.

Table 6-5: Prediction accuracy (R^2) for the different nitrogen components

	inflow	site 1	site 2	site 3	site 4	outflow
TN	0.65	0.69	0.55	0.41	0.58	0.74
NO_3^-	0.58	0.63	0.54	0.42	0.47	0.60
NH_4^+	0.79	0.89	0.70	0.75	0.83	0.86

The input variables for the prediction of the output variables TN, NO_3^- and NH_4^+ were: water temperature, conductivity, dissolved oxygen, redox potential, pH, turbidity, sediment TN, $TN_{(prevS)}$, evaporation and rainfall.

The prediction accuracy measured in form of the R^2 value showed, that prediction of output variables was considerably accurate, especially for NH_4^+. The comparison between the determined and measured N concentrations showed, that HEA was able to determine major peak events, but showed slight overestimation for TN concentration for the peak events and overestimation for NO_3^- concentrations ranging between 0.2-0.4 mg/L.

For TN the input variables with biggest influence in determining the TN concentrations were $TN_{(prevS)}$, DO and ORP additionally parameters selected for determining TN concentrations were turbidity, evaporation, conductivity and rainfall. The sensitivity analysis for the selected parameters showed that $TN_{(prevS)}$ and ORP had a positive effect on the TN concentration, whereas DO had a negative effect. Turbidity and rainfall had a positive influence, which was corresponding with the measurement, which showed higher concentration levels of TN during the summer seasons with lower DO levels and higher turbidity levels.

For NO_3^- the following input variables showed to have the greatest impact in determining NO_3^- concentrations were $NO_3^-{}_{(prevS)}$; other parameters selected were water temperature, DO, ORP, conductivity, pH, evaporation and rainfall. Looking at the seasonal concentration levels, which are showing that the average concentrations were high throughout the year with the exception of autumn other parameter showed a variation which were either positive or negative, for example DO and ORP. The DO and therefore the ORP conditions were relatively low levels during the growing seasons with low variations, therefore at certain conditions DO and ORP could have either a positive or negative influence in determining NO_3^-.

For NH_4^+ the input variables with the greatest impact for determining NH_4^+ showed like for the other N forms that the concentration from the previous site, based on the sensitivity analysis showed a positive effect, and which is appearing in all branch functions of the rule set. DO and ORP showed to have either positive or negative influences on NH_4^+, but low concentrations and low variations seemed to have minimal positive or negative influences, which can be explained that ammonification processes are occurring under aerobic and anaerobic conditions, with lower efficiencies under anaerobic condition. Other input variables selected for determining NH_4^+ were turbidity, conductivity, water temperature, ORP and sediment TN.

Overall it seems that the concentration from the previous site was the parameter with the greatest influence on the concentration levels for N forms, which has to be controlled. Therefore the parameters in form of DO and turbidity must be controlled to maintain a lower N concentration level based on the rule-set developed by HEA. In particular DO is a parameter, which also an important parameter in influencing P forms.

The next approach to find a generic rule set to predict N component concentrations for the different sites was performed by applying already developed rule sets used for knowledge discovery to the data sets of the different sites. The result of this approach showed that a generic rule set has been found for predicting the different N components. The rule set was able for the different N components to predict most of the peak events, especially for NH4+, which showed a very accurate prediction (Appendix E-2). For NO_3^- the generic rule showed lower prediction accuracy for inflow and site 1, but was showed higher prediction accuracies for the rest of the sites, which can be explained due to the fact that the generic rule was selected from the rule predicting the NO_3^- conc. for site 3 & 4, which have significantly lower concentration ranges than compared to inflow and site 1, which explains that the prediction of NO_3^- conc. at these sites were mostly underestimated (Appendix E-1). For TN the generic rule was able to predict most of the major peak events, but showed a higher accuracy in predicting TN concentration ranging up to 0.8 mg/L.

6.5.2 HEA for P components

6.5.2.1 Prediction and forecasting of outflow P concentration

The prediction of P components concentration for the outflow using selected parameters from the inflow was performed the same way it has been performed for N components. For model 1 the selected input parameters included all physical parameters including the nutrient concentrations from the inflow and model 2 it only included the physical parameters. The prediction accuracy showed for both P components, that model 1 had slightly higher prediction accuracy than model 2. The R^2 for TP model 1 was 0.63 and for model 2 it was 0.44. For PO_4^{3-} the R^2 for model 1 was 0.57 and for model 2 0.53. Both models were able to predict the peak events in regards timing and magnitude quite accurate, but model 2 missed one of the peak events. The input parameter setup from model 2 was applied to develop a forecasting model for TP and PO_4^{3-}. The forecasting model developed by HEA had prediction accuracy for TP of R^2 = 0.32 and for PO_4^{3-} R^2 = 0.52. For TP the model was able to predict only one major peak event and missed three, but seems to predict relatively accurate the TP conc. ranging up 0.03 mg/L. For PO_4^{3-} the developed model mostly predicted the major peak events, but showed also accurate predictions for lower concentrations as well (Appendix E-3).

6.5.2.2 Knowledge discovery and generic rules for the prediction of P for the different sites

The input variables for the prediction of the output variables TP and PO_4^{3-} were: water temperature, conductivity, dissolved oxygen, redox potential, pH, turbidity, sediment TP, $P_{(prevS)}$, evaporation and rainfall. The prediction accuracy for the different P forms can be seen in Table 6-6.

Table 6-6: Prediction accuracy (R^2) for the different phosphorus components

	inflow	site 1	site 2	site 3	site 4	outflow
TP	0.55	0.44	0.20	0.43	0.41	0.53
PO_4^{3-}	0.69	0.45	0.33	0.58	0.12	0.66

The determination for the different P forms showed, that the comparison between the measured and determined revealed that HEA was able to peak events for most of the sites, with the exception for the prediction of PO_4^{3-} for site 4, where the HEA estimating major peak events during the first year from June – December 2005.

For the determination of TP the HEA has selected the following input variables, which are water temperature, conductivity, pH, DO, ORP, sediment TP, turbidity, evaporation and $TP_{(prevS).}$

The parameters, which were most sensitive for determining TP were $TP_{(prevS)}$, DO, ORP and water temperature. $TP_{(prevS)}$ had, like for the previous nutrient forms, a positive effect in determining TP. Water temperature increased towards the summer seasons and the monitoring showed that the concentration for P forms were highest during the summer seasons. Related to that DO and ORP had a negative effect, which means that with increase of the input range the output range is decreasing. During summer the DO levels are low and therefore creating an anaerobic condition, which would lead to a release of P from the sediment, which is support by the ORP.

For phosphate the HEA selected the following parameters to be influential on determining PO_4^{3-}, which are conductivity, DO, ORP, pH, turbidity, $PO_4^{3-}{}_{(prevS)}$, sediment TP, rain and evaporation. The input variable with greatest impact on the PO_4^{3-} concentration was like for the other nutrient form, the concentration level at the previous site, which is influencing the nutrient levels of the following sites. The sensitivity analysis showed that there was a positive relationship. Similar to TP, DO and ORP had a negative influence on the level of PO_4^{3-} concentration for all sites, which is corresponding with the observation on-site. The finding of generic rule to determine the P component concentrations for the all sites, was performed the same as for the N components. The results of these approach showed that the rule, which predicted the P component concentrations for site 1 & 2 was inaccurate in predicting the P components at the other sites, which can be seen in the low R^2 values. The predictions were showing overestimations and predicting peak events, which didn't occur under natural conditions. A possible explanation could be, that because of the significantly higher P concentrations at the site 1 & 2 the rule is applying the same pattern to the other sites, which wouldn't resulting in good results, because of different dynamics. Another factor would be sediment, because this is the main source and sink of the P components, which is showing a more complex condition than compared to the prediction of N components.

6.5.3 HEA for DOC

6.5.3.1 Prediction and forecasting of outflow DOC concentration

The prediction of the outflow DOC concentration by using the selected measured input variables showed that the HEA was able to produce reasonable prediction models estimating the timing and magnitudes of peak events. Two experiments were run simultaneously with the difference of one input variable. Model 1 included the DOC concentration of the inflow as an input, whereas model 2 didn't include DOC concentration of the inflow as an input. The prediction performance of model 1 was much more accurate in predicting DOC at the outflow than model 2, but still model 2 was able to predict the DOC in a reasonably manner. Both prediction models selected similar input variables for their sensitivity analysis with the exception that model 1 selected the DOC conc. from the inflow as a key variable, which was selected as a key variable by the HEA, therefore the results of model 2 and the input variable setup was used to carry out a 7 days ahead forecasting experiment. Other variables selected were water temperature, pH, turbidity and conductivity. The first three variables are having an influence on the microbial community, which is responsible for the production of carbon due to decomposition of organic matter and utilization as well. The variable conductivity is a more site-specific condition, which is related to pumping of aquifer (ground-) water into the reed bed pond during the summer seasons. The results of model 2 were used to do further experiments, based on the fact that the HEA was still able to produce a good prediction of the DOC concentration at the outflow, by just using only physical variables. But it demonstrates that a more accurate prediction model can be achieved. The next step is to develop a forecasting model.

For the forecasting model the input set up from model 2 was selected and the DOC conc. at the outflow was forecasted 7 days ahead and results showed that the HEA developed a very predictive model able to estimate most of the major peak events, by slightly underestimating them. Two events were missed, but the forecasting model was accurate in predicting DOC concentration up to a concentration of around 15 mg/L.

6.5.3.2 Knowledge discovery and generic rules for the prediction of DOC for the different sites

The input variables for the prediction of DOC concentrations in the reed bed pond were: water temperature, conductivity, dissolved oxygen, redox potential, pH, turbidity,

$DOC_{(prevS)}$, chl-a, macrophyte biomass, evaporation and rainfall. The prediction accuracy for the different P forms can be seen in Table 6-7.

Table 6-7: Prediction accuracy (R^2) for the different carbon component

	inflow	**site 1**	**site 2**	**site 3**	**site 4**	**outflow**
DOC	0.40	0.31	0.36	0.50	0.51	0.66

The predictions of DOC for the different sites revealed, that most of the peak events was well predicted in regards timing and magnitude, whereas there are over- and under-estimations for the medium range concentrations (10-20 mg/L). The driving variables for the prediction of DOC for the different sites showed, that different parameters are influencing the DOC concentration.

The DOC concentration at the inflow area showed that the parameters water temperature and $DOC_{(prevS)}$, which was selected for the "THEN" branch. The "ELSE" branch selected the parameters water temperature, pH and chl-a having a sensitive relationship to DOC. For all parameters the relationship on DOC was positive, which means that an increase of input range results in an increase of DOC concentration. For the following sites the variables of rainfall and $DOC_{(prevS)}$ had a positive relationship on DOC, which are used in the function of the "THEN" branch, whereas the "ELSE" branch only selected the variable $DOC_{(prevS)}$. Rainfall didn't seem to have a positive or negative influence regards the DOC concentration, whereas the $DOC_{(prevS)}$ had a positive relationship for either the "THEN" and "ELSE" branches. DOC concentrations at site 3 and 4 were influenced by the variables water temperature, conductivity and OPR selected for "THEN" branch and $DOC_{(prevS)}$ for the "ELSE" branch. With the exception of conductivity, which had a negative influence, all the parameters selected to determine DOC had a positive effect. At the outflow site the DOC concentration was determined by the variables DO and water temperature selected for "THEN" branch and $DOC_{(prevS)}$ for the "ELSE" branch. DO had a negative effect on DOC and therefore water temperature and $DOC_{(prevS)}$ showed to have a positive effect. The findings of the relationship between the selected variables and the determined DOC concentrations were corresponding with observations and measurements. An increase of the water temperature towards the summer seasons showed that the DOC concentrations were significantly higher, which was reflected as a positive influence due to the sensitivity analysis. Linked to water temperature an increase will lead to a decrease of DO due consumption and therefore higher concentrations of DOC were measured caused by the leaching from the organic matter by decomposition. DOC concentrations from the previous sites had a positive effect based on the sensitivity analysis, which is corresponding that higher concentration at the previous site will lead to higher concentrations than compared to the situation when the concentration at the previous site was lower.

Based on using the HEA to determine the DOC concentrations for the different sites showed that water temperature and DOC concentration form the previous site had the biggest influence in determining and therefore to control the concentrations these variables have to controlled, including the other parameters having an influence on determining DOC.

Additional to analysis of the key parameters influencing the DOC concentrations for the different sites the following approach was made to find a generic rule set predicting DOC concentration for all sites. The generic rule was selected by applying the rule sets found for knowledge discovery to all the other sites. The rule for the prediction of DOC concentration of site 1 & 2 showed the best results, which showed reasonably accurate predictions of major peak events as well as the prediction of lower and medium concentrations.

7 Conclusion

This study was conducted in the period from March 2005 to August 2007 and involved the collection and analysis of water, sediment and plant samples from the field sites and laboratory experiments, including statistical analysis. The purpose of the study in general was to get a better understanding of the nutrient dynamics in the water column, sediment and macrophytes in order to assist in informed water quality control.

7.1 Nutrient dynamics of Stormwater runoff

7.1.1 Temporal variation of nutrients in stormwater runoff

The nutrient dynamics which are affecting the water quality showed temporal and spatial differences, which were statistically significant. Most of the nutrient forms measured during the study showed highest concentrations during the summer seasons and lowest concentrations during the autumn and winter seasons. Seasonality is affecting the physical parameters, like temperature, DO, ORP and turbidity, which in turn affect chemical and biological processes determining the dynamics of the nutrients. During summer with an increase of water temperature the microbial activity stimulates processes like nitrification and denitrification that increase oxygen demand, resulting in lower concentrations of DO and levels of ORP creating an anaerobic environment. Anaerobic conditions at the sediment-water interface cause P release from the sediment into the water column. Other transformation processes like the ammonification of org. N will also be influenced with a lower efficiency under anaerobic conditions. Other important N transforming processes are nitrification and denitrification, which are controlled by microorganisms under aerobic conditions for nitrification and anaerobic conditions for denitrification.

7.1.2 Spatial variation of nutrients in stormwater runoff

Additionally to the temporal variations, spatial differences between the different collection sites were observed as well. The levels of concentrations for the different nutrients were significantly higher at sites 1 and 2, which are located near the inflow area, than the sites 3 and 4, which are located near to the outflow. Site 1 and 2 are dominated the macrophyte species *P. australis*, which is also the case for site 3. Site 4 was dominated by *B. articulata*. Sites near the inflow have to deal with higher loadings of nutrients indicated by high primary productivity in form of macrophyte biomass. The monitoring of macrophyte biomass revealed that sites 1 and 2 had significantly greater primary productivity than sites 3 and 4. Due to the higher productivity of macrophyte community the uptake rates were also higher at these sites, but without proper management the utilized nutrients will re-enter the system by senescence or decomposition. The analysis of the removal performance showed that sites 1and 2 were showing negative removal performances, especially site 1. The removal efficiency and retention capacity at site 1 showed for all nutrient forms with the exception of DOC negative performances, therefore site 1 is considered to be an internal source for most of the nutrients. However nutrient uptake rates were highest at site 1, which means that the internal production of nutrients was greater than the utilization, which is linked to the plant density. Macrophytes are considered to be main autochthonous source in a treatment system, providing organic materials, which will be decomposed and therefore leaching back nutrients. The amount of organic material will increase the turbidity levels, which especially during the summer seasons will cause anaerobic conditions, which is leading to a release of P from the sediment into the water column.

7.1.3 Allochthonous and autochthonous sources

The Parafield Stormwater Harvesting Facility is receiving stormwater runoff containing high concentrations of pollutants in form of nutrients, heavy metals and suspended solids. The majority of nutrients entering the system are allochthonous sources, but due to the nature of the treatment system autochthonous loadings are occurring due to the dense macrophyte community.

Based on the results and data analysis site 1 is considered to be the major source for autochthonous loadings for most of the nutrients, with the exception of DOC. The main internal source for DOC was site 2. Both sites differ significantly with respect to the nutrient levels in the water column and plant community. The loading amounts for the different nutrients were for TN 0.39 mg/m^2/day, NO_3^- 0.07 mg/m^2/day, NH_4^+ 0.12 mg/m^2/day, TP 0.076 mg/m^2/day and PO_4^{3-} 0.063 mg/m^2/day. The internal DOC loading at site 2 amounted to 1.06 mg/m^2/day. These results are proving that the high plant density at these two sites were responsible for internal loadings in the reed bed ponds. Higher primary productivity in form of macrophyte biomass is transformed after senescence into organic matter, which will be decomposed. However the permanent inundation of the wetland results in slow decomposition rates, which leads to an accumulation of organic matter causing anaerobic conditions. As a result nutrients are released and transformation processes are slowed down. Therefore to minimize internal loadings water quality parameters and macrophyte must be controlled.

7.2 Removal Performance of the Reed Bed Pond

The overall removal performance of the reed bed pond for the different nutrients showed a medium to low efficiency. The removal performance for the nutrients is influenced by a variety of factors and one of the most important factors is residence time.

7.2.1 Seasonal impact of residence time on nutrient removal performance in the reed bed pond

The semi-arid or mediterranean climate in South Australia causes hot and dry summer, with sporadic rain events, and more frequent rain events during the autumn and winter seasons.

Therefore seasonal differences in the water residence time are typical with longer residence times during summer and shorter residence time during the winter. Statistically the difference of residence time between the seasons was not significant, but the residence time was longer during the summer season and was shortest during spring.

The removal performances based on the seasonality showed that the removal efficiencies for the different forms of nutrients were higher during autumn and winter and lower during spring and summer. In case of P components the removal efficiency was most efficient during autumn and summer.

The influence of the flow regime on the removal performance for the different nutrients were most efficient at >15 days for TN (RE: 29.6%), NO_3^- (RE: 51.3%) and DOC (RE: 28.7%), <5days for NH_4^+ (RE: 10.7%), 5-10days for TP (RE: 40.0%) and PO_4^{3-} (RE: 51.6%).

7.3 Macrophytes

The macrophyte community in the reed bed pond of the Parafield Stormwater Harvesting Facility showed that there were spatial differences in the primary productivity and nutrient storage capacity between the sites near inflow and outflow. Sites 1 and 2 near the inflow had a significantly higher primary productivity and nutrient storage than sites 3 and 4 near the outflow. High productivity during the growing seasons resulted in higher nutrient uptake rates at sites 1 and 2, but created a problem in autumn by recycling nutrients due to senescence or decomposition. Slow decomposition causes accumulation of organic matter, which has an impact on the nutrient dynamics in the water column, creating an environment of high turbidity and low oxygen conditions, which in turn slows down chemical transformation, like ammonification and nitrification processes. Anaerobic conditions favour the release of nutrients from the sediments, especially P, and microbial decomposition of organic matter.

Annual comparison showed an increase of primary productivity. Nutrient concentration in form of TP showed that the concentrations of 2006 and 2007 were significantly higher than 2005, whereas the concentrations of TN showed no significant differences

7.3.1 Plant Harvesting

The plant harvesting performed at the end of January 2006, where the above-ground biomass (above the sediment level) was harvested, showed to have an impact on the nutrient dynamics in the water column and sediment. The average nutrient concentrations for all measured nutrients in the water column, with the exception of NO_3^-, showed a decreasing trend after the harvesting. The nutrient dynamics for the sediment before and after the harvesting changed as well. The average nutrient concentrations of TC and TN showed an increasing trend after the harvesting whereas TP concentrations decreased. Therefore the harvesting had a positive impact in reducing nutrient levels in water column and increasing nutrient concentration in sediment due absorption and precipitation.

It also showed a positive effect on key driving water quality parameters, like DO and ORP. The DO concentration after the harvesting increased slightly from 3.92 to 4.41 mg/L (2005: 3.92 mg/L, 2006: 5.69 mg/L and 2007: 3.12 mg/L). ORP also increased after the harvesting from -19.42 to 19.38 mV (2005: -19.42 mV, 2006: 33.78 mV and 2007: 4.99 mV).

But the plant harvesting performed was unsuccessful in order to stimulate regrowth of macrophyte community; therefore the recovery of the macrophyte community after the harvest couldn't be determined and an effective harvesting regime wasn't achieved. Instead of the regrowth of the macrophyte community at the harvested area, the open water area stimulated the growth of the green algae *Spirogyra,* indicated by an increase of chl-a concentrations (from 34.31 to 53.96 µg/L) after harvesting, which inhibited the growth of macrophytes.

Therefore another harvesting regime must be applied to effectively harvest the nutrients through macrophytes and stimulate growth rates at the same time. Additionally to that the harvesting regime must avoid creating open water area, which would stimulate the growth of *Spirogyra*.

7.4 Sediment

The sediment survey revealed also distinctive seasonality. The nutrient concentrations in form of TC, TN and TP in the sediment were significantly lower during the spring and summer seasons than the autumn and winter seasons.

Higher nutrient concentrations during the autumn and winter seasons indicate the dominance of precipitation or sorption mechanisms for the different nutrients therefore the sediment was functioning as a sink, because utilisers' in form of the macrophytes, algae and microorganisms weren't active or showed decreased activities. In case of the macrophytes they were acting as an internal source of nutrients leaching nutrients back to the system due to senescence and decomposition. Lower nutrient concentrations during the spring and summer seasons indicated that nutrients were released or relocated from the sediment due release of nutrients into the water column, absorption by plants and volatilization.

The inter-annual comparison of nutrient retention showed that the overall function of the sediment was to be a source of nutrients and that there was no accumulation of nutrients in the sediment, with the exception of year 2006. Over the whole study the sediment in the reed bed pond was functioning as a source.

7.5 Data Modelling by means of HEA

The HEA using the "Bootstrap" method was to test if it could development a reasonable prediction model utilizing data from a smaller nature which had a weekly interval.

7.5.1 Prediction nutrient condition at outflow

The prediction of the different nutrients at the outflow by using selected input variables from the inflow showed that relatively accurate prediction model can be produced by using the "Bootstrap" method running a small dataset. By developing two prediction models for each nutrient forms, which varies by one input variable showed, that a more accurate prediction model is possible.

7.5.2 Forecasting

The development of a forecasting model, in this case 7 days ahead forecast, showed that the “Bootstrap” method was able to utilize different patterns and conditions, which made it possible to create forecasting models of moderate prediction accuracies based on 2 1/2 years of monitoring data.

7.5.3 Knowledge Discovery

The application of HEA allowed determining the key parameters having an impact on the output variables in form of the different nutrients.

The results of using HEA for predicting the different nutrient concentration based on the “Bootstrap method” revealed R^2 levels ranging between 0.12 – 0.89. It demonstrated that the “Bootstrap method” is able to predict patterns within a short data set.

For the prediction of the different N components in the water column the key parameters were DO, turbidity and the nutrient concentration from the previous site. With the exception of DO the other two parameters had a positive impact on the N components, which that an increase in levels of these parameters is linked to an increase of N component concentrations; therefore the levels of these parameters have to be kept low. Whereas DO had a negative impact, which means that with an increase of DO levels the nutrient levels will decrease.

For the prediction of different P components in the water column the key parameters were DO, ORP and the nutrient concentration from the previous site. DO and ORP had a negative impact on the nutrient levels, which means that for an effective control the levels of these parameters have to be kept as high as possible. Considering that OPR is influenced by the DO concentration it will be recommended to keep the DO concentration levels above the 4 mg/L threshold, because DO concentrations under this threshold will consider the system to be anaerobic.

The key parameters for influencing the concentration of DOC were water temperature and the nutrient levels from the previous site. Both parameters showed a positive effect on predicting the DOC concentrations.

Overall it can be concluded that the key parameter for determining the different nutrient concentrations is DO, which has a negative impact on the nutrient concentrations and therefore DO levels have to be controlled to keep nutrient concentrations low and therefore improve the water quality.

7.5.4 Generic Rule Development

The development of a generic rule, which can be applied to predict the nutrient concentrations for all the sites showed success for nitrogen and dissolved organic carbon, but for the phosphorus components it showed limitations based on spatial differences in the nutrient dynamics, at which a more site-specific rule would be more applicable.

7.6 Implication for Water Quality Control

Based on the results from this research the following implications for management can be concluded.

7.6.1 Residence Time

Water residence time is a key control option used in treatment wetlands. Considering that the removal performance for the different showed to be efficient at different flow regimes a regulation of the residence time would be able to keep the removal performance of the reed bed pond on an efficient level. The measured nutrients concentrations were highest during the summer season with an average residence time of 13.7 days. The residence time during that period is not creating the optimal environment for the most efficient removal performance; therefore residence times should be adjusted. Nutrient forms which will be targeted by the removal performance have to be decided, but it would be most effective to control the limiting nutrient source, which is in this system P. Therefore the optimum residence time will be between 5-10 days. Under the current water residence time the removal performance is 10% less effective. The optimal control of other nutrients such as DOC and NH_4^+ affecting the water quality requires also adjusted flow regimes. During the summer period the reed bed pond is supplied with aquifer water. The effect of the aquifer supply showed that the level conductivity was increasing almost two times from the level of stormwater runoff during the rainy seasons (Figure 7-1).

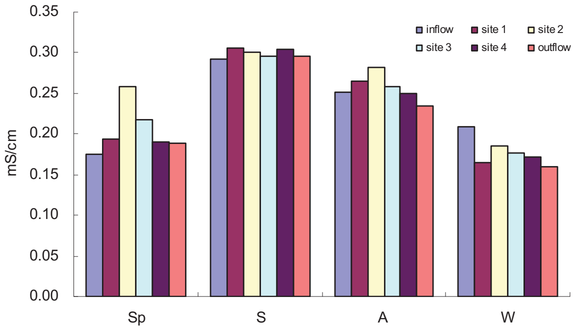


Figure 7-1: Seasonal comparison of electrical conductivity for the different collection sites (Sp = Spring, S = Summer, A = Autumn and W = Winter)

Since conductivity levels are representing ion state it can be used as salinity indicator. The supply of water during the summer period can only be uphold by pumping aquifer water into the system, but higher levels of conductivity could have long term effects on the infrastructure like corrosion of pipes. At the same time the supply water, which is used for secondary purpose such as irrigation could cause a salinity problem in the soil of terrestrial systems. Also it was revealed that aquifer water supply had a negative impact on RE with the exception for P RE. Therefore a direct input into the reed bed pond should be avoided or reduced which can be done by mixing the aquifer water with the water in the storage pond, which is only a possibility when water is present.

7.6.2 Plant harvesting

Plant harvesting is considered another management option in wetland treatment systems. The harvesting performed at the begin of 2006, which removed all above-ground biomass above the sediment showed a positive results in regards decreasing nutrient concentrations in the water column and increasing key-driving parameters like DO and ORP, but the regime failed to stimulate regrowth of macrophyte community after the harvesting, which was inhibited by the growth of algae and water stress caused by the water level. In general previous studies suggested to harvest multiple times during the growing seasons to effectively remove nutrients from the system. The harvesting regime in the reed bed pond at the Parafield Stormwater Harvesting Facility should consider, that the harvesting should avoid creating large open water areas, which is stimulating the growth of algae shown during the first harvesting regime, therefore patch harvesting and harvest of above-ground biomass over the water surface level, which would still allow to harvest a significant amount of nutrients, because during the growing seasons the highest nutrient storage is in the leaves. By cutting the plants above the water level the advantage is, that the water levels don't need to be changed (lowering and raising water levels), therefore water stress due to flooding is avoided and water is saved. It also would provide less open water area, inhibiting algae growth, therefore a stimulation of the macrophyte regrowth could be achieved, which allows multiple nutrient harvesting increasing the effect of removing nutrients, which would possibly re-enter the system and would create a management risk. The harvesting will not just remove nutrients from the system, but it also removes the source of organic matter. All undecomposed or inorganic material will accrete in the wetland system, which will have an impact on the turbidity level, which in high levels is known to have an inhibition impact on macrophyte growth. In shallow systems the accretion will be resuspended due to turbulences creating high levels of turbidity, which is linked to higher nutrient concentrations. Therefore the harvesting plan should include all these aspects and target sites responsible for autochthonous loadings, which is in the reed bed pond the sites located near the inflow area.

7.6.3 Sediment

The function of the sediment in the reed bed pond in regards the different nutrients showed a function shift from sink to source and vice versa with the change of the seasons. Overall the nutrient budgets showed that there was no accumulation of nutrients in the sediment. But the enormous amount of organic matter produced by the macrophyte community, which is slowly decomposing, creates more or less anaerobic conditions at the sediment-water interface, which during the growing seasons is a source of nutrients, especially for P components, due to the release from the sediment. On the other transformation process like ammonification and nitrification, which are performed by the microbial community will inhibited as well due to anaerobic conditions. Regards siltation rates no data is present therefore based on the results during the study period the management of sediment isn't necessary, but in future considerations of drying and wetting should be included, which will aerate the sediment, and therefore the sediment can act as sink for the nutrients rather than to be a source. But this would work more efficiently by having an additional pond, so that in case one pond is dried, that the other pond is kept inundated. An additional pond will also increase the treatment capacity of the system receiving and treating stormwater runoff allowing a greater water volume to be available for aquifer recharge.

7.6.4 HEA

The HEA results for the prediction of the different nutrient forms revealed, that the key driving variables were DO, ORP and turbidity. Based on their impact on the different nutrients the understandings can be used for decision making regards management. The development of models predicting the different nutrients in combination with management methods would lead to best practice management, for example to know the peak events for a certain nutrient components and the knowledge of the key driving parameter, would allow to choose a management tool to maximize the removal performance of the system, which is the main purpose of the system, by providing optimum conditions. Currently the results of HEA are to determine key driving variables only specific for the Parafield Stormwater Harvesting Facility for this kind of climate, but by establishing a monitoring program measuring the key variables a much more accurate model can be developed, which is used for decision making. Under the current situation online monitoring measures the following parameters, like turbidity, conductivity, water level and flow rate. The parameter in form of DO should be added to the list of parameters for online monitoring, because it is the most fundamental parameter in freshwater ecosystems, which controls most of the treatment processes, like oxidation, respiration and nitrification.

7.7 Future Considerations

- Monitoring of the water column, sediment and macrophytes should be continued to develop a greater dataset, which can be utilized for modelling and forecasting nutrient concentration.

- Research on the microbial community and processes will provide a better understanding of chemical transformation and utilization processes with regards to the different nutrients

- Research on species specific optimum macrophyte harvesting is essential in order to better control the influence on/of the green algae Spirogyra on the reed bed pond system

- Research on covering the surface and shading the open water of the sedimentation and storage pond and later for reed bed pond will allow preventing water loss by evaporation as well as photo-inhibition of the macrophyte community and reed bed system.