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Sustainable Water Management in the City of the Future

Integrated Project
Global Change and Ecosystems

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Water quality enhancement in Sloterbinnenpolder (Amsterdam, The Netherlands) by adopting ecological engineering approaches

Ahmed Medhat Mohamed Ali

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Water quality enhancement in Sloterbinnenpolder (Amsterdam, The Netherlands) by adopting ecological engineering approaches

Master of Science Thesis
by
Ahmed Medhat Mohamed Ali

Supervision
Prof. Piet Lens, PhD (UNESCO-IHE)
Diederik Rousseau, PhD (UNESCO-IHE)

Examination committee
Prof. Piet Lens, PhD (UNESCO-IHE), Chairman
Diederik Rousseau, PhD (UNESCO-IHE)
Rob Ververs, MSc (Waternet)

This research is done for the partial fulfilment of requirements for the Master of Science degree at the UNESCO-IHE Institute for Water Education, Delft, the Netherlands

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Dedicated to

My beloved mother, Tawheda
My love, Hanaa
My future, Omar
My brother, Mohamed

Abstract

Sloterbinnenpolder (SBP) is very problematic with regard to surface water quality. The water bodies in the polder are suffering of high load of nutrients (mainly phosphorus) which caused eutrophication problems especially in the Sloterplas lake. Therefore, the main goal of this research was to improve the water quality in Sloterbinnenpolder.

To achieve this goal, identification of the sources of phosphorus loads (as the main limiting nutrient) to the Sloterplas Lake and assessment of different proposed measures that can reduce external loads, and accelerate the self-purification capacity of the Lake were carried out. The research assessed ecological engineering measures and conventional engineering methods that could reduce phosphorus in the polder.

It was concluded that both internal phosphorus load (32% of the total load) and external phosphorus load (68% of the total load) to the lake are contributing to the lake eutrophication problem, so attention has to be paid for both internal and external loads in order to restore the lake. In addition, it was found that the runoff, pumps, and inlet are the most important external loads contributors, where they represent about 99% of the total external loads.

The research results showed that the external phosphorus loads reduction that could be achieved by runoff treatment by ecological engineering (BMPs) and treatment of the pumps and inlet flow rates was quite high (75%), but was not enough to restore the lake due to the natural background loads, and the extensive internal load.

Keywords: Sloterbinnenpolder, eutrophication, phosphorus, ecological engineering, best management practices (BMPs)

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1 Introduction

Due to the continuing processes of industrialization and urbanization coupled with the natural growth of population, and un-sustainability factors and risks inherent to conventional urban water management (UWM), cities will experience difficulties in efficiently managing the aqueous environment in general and scarcer and less reliable water resources in particular.

Moreover, the population growth in the urban areas is the major factor of land cover conversion and storm-water drainage that lead to degradation of aquatic ecosystem quality (Brezonik & Stadelmann, 2002; Ierotheos et al., 2003).

In this context Ecological Engineering can play an important role, and can be employed for effective UWM, this can be undertaken by the processes in natural systems relating to water purification and remediation ; these processes can be employed for effective UWM in two ways (SWITCH, 2007): a) by applying the processes of natural systems in 'Ecological Engineering' (e.g. buffer strips, Bio-manipulation, lake restoration), or, b) by employing full scale natural systems (e.g. phyto-technology, eco-hydrology).

The urban water system in the Sloterbinnenpolder (SBP) in Amsterdam, The Netherlands, the focus of this research, is very problematic with regard to surface water quality. So, this research will study the possible measures to improve the water quality and self-purification capacity in the aforementioned system. The research will help in achieving a water quality in accordance with the European Water Framework Directive (EWFD), which requests a program of measures to reach a good chemical and ecological condition before 2015.

The proposed research will be carried out within the European 6th Framework program, Sustainable Water management Improves Tomorrow's Cities' Health project (SWITCH), which calls for a paradigm shift in UWM, with a need to convert ad-hoc actions (problem/incident driven) into a coherent and consolidated approach (sustainability driven), in which the use of natural systems and processes for the effective management of the urban water cycle as a whole is one of its objectives.

1.1 Problem Statement

Waternet has to provide a detailed management plan for several areas to reach the goals of the EWFD. One of these areas is the SBP. SBP is an urban area with separate stormwater and sewerage systems. Surface water in the polder includes water bodies such as ditches, canals, and Sloterplas Lake. All water bodies are suffering of high load of nutrients,

mainly phosphate; especially The Sloterplas Lake has an extremely high phosphate load. The high nutrients load causes eutrophication problem in the lake in the summer. The source of these nutrients is not exactly known. Therefore, there is a need for carrying out research to identify the source of the nutrients load and to find the potential solutions of the problem.

1.2 Goal and Objectives

The goal of this MSc research is to improve the water quality in Sloterbinnenpolder. Therefore the objectives of the research are:

1. To identify the sources of high nutrients load in the polder;
2. To identify, describe and analyse a range of natural systems and processes that could be employed in stimulation of the self purification capacity of water bodies, and managing the urban water system;
3. To propose and assess potential Ecological Engineering measures, as an approach for utilizing the nature systems and processes, to decrease the loading of nutrients, particularly phosphorus, from various sources in the polder.

And the specific aims of the research are defined as follows:

1. To help in achieving a good chemical and ecological condition in Sloterbinnenpolder before 2015 as requested by EWFD; and
2. To maximize the utilization of Ecological Engineering for the effective management of the urban water system, as a contribution to one of the main SWITCH objectives.

2 Literature Review

As has been mentioned in the previous chapter the urban water system in the Sloterbinnenpolder (SBP) is very problematic with regard to surface water quality. The water bodies in the polder are suffering of high load of nutrients (mainly phosphorus) which caused eutrophication problems especially in the Sloterplas lake. So, this chapter will present the possible ecological engineering measures that can improve the water quality with regard to eutrophication problem of the Sloterplas Lake, in addition to the diversion or the treatment of the inflows. Moreover, the definition of self-purification capacity and the natural processes that play a role in self-purification capacity (one of the main objectives of the SWITCH Project) will be presented.

2.1 Natural Processes and Self-Purification Capacity in Aquatic Ecosystems

2.1.1 Water Self-Purification Capacity

Self-purification processes must be considered in the management of aquatic systems, particularly in the context of the EU Water Framework Directive, which emphasises the conservation or restoration of good ecological quality. The main problem comes from the difficulty of research about aquatic ecosystem functioning, because of multiple and complex interactions between physical, chemical and biological factors.

Ostroumov (2006) defined the Water Self-Purification Capacity as a system of natural processes that lead to improving water quality as a result of the interplay of natural factors without any man-made attempts to treat or purify water.

2.1.2 Natural Processes

Many physical, chemical, and biotic processes are important determinants of water quality and water purification in aquatic ecosystems. Table 2.1 shows a list of these processes.

Ostroumov (2006) stated that the dominant processes of the self-purification of aquatic ecosystems are: (1) filtration activity; (2) mechanisms of transferring chemical substances between ecological compartments (from one medium to another); and (3) degradation of pollutant molecules.

Table 2.1: Some processes playing a role in purification, remediation and upgrading the quality of water in aquatic ecosystems (Ostroumov, 2006).

Types	Processes
Physical	• Dilution • Adsorption • Sedimentation • Evaporation • Advection and mixing
Chemical	• Hydrolysis • Photochemical reactions, photolysis • Oxidation and reduction • Free radical-dependent destruction • Complexation by and binding to other molecules
Biological	• Microorganism-dependent oxidations, reductions and other biotransformations • Transformations performed by excretion of chemical substances and enzymes • Accumulation by organisms • Filtering of water by suspension-feeding organisms • Excretion of molecules that are instrumental in increasing the rate of some chemical processes of degradation of pollutants (e.g., photodegradation) • Producing oxygen that is involved in chemical oxidation of pollutants • Regulation of biological processes of water purification by other organisms

2.2 Ecological Engineering

Ecological engineering, a new emerging discipline, seeks a symbiotic mix of man-made and ecological self-design that maximizes productive work of the entire system. Allowing self-organization of plant, animal and microbial biota to develop cooperative mechanisms may develop better-adapted ecosystems to handle pollution and toxicity (Odum, 1983). So by minimizing human manipulation and the use of machinery, ecological engineering solutions aim to increase material recycling, enhance efficiency, reduce costs, and maximize the contributions of natural processes in the total system.

2.2.1 Definition

In 1962, H. T. Odum was among the first to define ecological engineering as "environmental manipulation by man using small amounts of supplementary energy to control systems in which the main energy drives are still coming from natural sources. Formulas for ecological engineering may begin with natural ecosystems as a point of departure, but the new ecosystems, which develop, may differ somewhat". Then, in 1971 he developed the concept of ecological engineering in his book Environment, Power and Society as follows: "The management of nature is ecological engineering, an endeavor with singular aspects supplementary to those of traditional engineering. A partnership with

nature is a better term." He later states in his comprehensive work Systems Ecology that "the engineering of new ecosystem designs is a field that uses systems that are mainly self-organizing."

Ecological engineering or Ecotechnology is defined as the design of human society with its natural environment for the benefit of both. It is engineering in the sense that it involves the design of this natural environment using quantitative approaches and basing our approaches on basic science. It is a technology with the primary tool being self-designing ecosystems.

The components are all of the biological species of the world (Hadjinicolaou, 2003).

Uhlmann, Straskraba and Gnauck have defined ecotechnology as the use of technological means for ecosystem management, based on deep ecological understanding, to minimize the costs of measures and their harm to the environment (Hadjinicolaou, 2003).

2.2.2 Ecological Engineering Principles

An extensive list of principles which any ecosystem design should consider was presented by Mitsch and Jørgensen in their book Ecological Engineering: An Introduction to Ecotechnology. A summary of these principles is given below (Mitsch and Jørgensen, 1989).

1. Ecosystem structure and function are determined by the forcing functions of the system. Alterations of the forcing functions cause the most drastic changes in ecosystems.
2. Ecosystems are self-designing systems. The more one works with the self-designing ability of nature, the lower the costs of energy to maintain that system.
3. Nutrients, metals, and other elements are recycled in ecosystems. Matching human activities and ecosystems in recycling pathways will ultimately reduce the effects of human activity by-products and pollution.
4. Ecosystem stability requires a balance between biological and chemical composition. A chemically overloaded ecosystem will not remain stable.
5. Processes in ecosystems have characteristic time scales that may vary over several orders of magnitude. Manipulation of ecosystems must be adapted to the ecosystem dynamics.
6. Ecosystem components have characteristic space scales. Manipulation of ecosystems must consider the appropriate size necessary to achieve the desired results.
7. Chemical and biological diversity contribute to the buffering capacities of ecosystems. It is necessary to take advantage of, and to protect this capability for greater efficiency.
8. Ecosystems are most vulnerable at their geographic edges. Ecological management has its best results in the ecosystems optimal geographic range.

9. Ecosystems are coupled with other ecosystems. This coupling should be maintained wherever possible and ecosystems should not be isolated from their surroundings.

10. All components of an ecosystem are interlinked. It is impossible to manage one component without affecting other parts.

11. Ecosystems have feedback mechanisms, resilience, and buffer capacities in accordance to their evolutionary history. Existing ecosystems do not easily adapt to human-made synthetic chemicals, although new emerging ecosystems can develop to deal with them in some cases.

2.2.3 Application of Ecological Engineering in Urban Water Management

The application of Ecological Engineering will be focused on the Sloterplas Lake restoration within the urban water system in the Sloterbinnenpolder (SBP). Lake restoration methods can be divided into two major categories: (1) external load control measures that can eliminate stress loadings, and (2) internal load control through in-lake measures that can accelerate the self-purification capacity of the Lake (Jørgensen and Vollenweider, 1988).

2.2.3.1 External Load Control Measures

One of the popular applications of ecological engineering in urban water management is the treatment of stormwater through the Best Management Practices (Hadjinicolaou, 2003).

The stormwater treatment practices fall into five major categories (Centre for Watershed Protection, CWP, 2000) .Within each category; there are several design variations, which will be discussed in more details in Section 2.3.

1. Stormwater ponds: Wet Pond, and Dry Extended Detention Pond
2. Stormwater wetlands.
3. Infiltration practices: Infiltration Trench, and Infiltration Basin.
4. Filtering practices: Surface Sand Filter, Underground Sand Filter, Perimeter Sand Filter, Organic Filter, and Bioretention.
5. Open channels: Vegetated Swale and Vegetated filter strips.

2.2.3.2 Internal Load Control Measures

Reduction of inflows is not sufficient to improve the quality of water in lakes and reservoirs, but recovery of lakes can be accelerated by in-lake methods. To remove long-lived contaminants, exotic species or to reintroduce ecosystem components lost through the impact of stress are some remedies in this category (Asaeda et al., 2004).

Methods in this category are dilution, Phosphorus inactivation, Sediment skimming, sediment oxidation, hypolimnetic withdrawal, biomanipulation, and artificial circulation. Each of these methods will be discussed in more detail in section 2.4.

2.3 Urban Runoff Treatment

Recent reports by the United States Environmental Protection Agency (EPA) indicate that stormwater runoff is one of the major sources of water quality impairment in water bodies across the United States (U.S. EPA, 1994; U.S. EPA, 2002a). The EPA classifies stormwater runoff as runoff from impervious surfaces including streets, buildings, lawns, and other paved areas that enter a sewer, pipe, or ditch before discharging into surface waters (U.S. EPA, 1994). Pollutants found in stormwater runoff range from oils and greases to nutrients and sediments.

2.3.1 Sources of Pollutants in Runoff

EPA reported that municipal wastewater (point source) and runoff from agricultural lands and hydrologic modifications (non-point source) were the primary sources of water quality impairment (U.S. EPA, 2002a). Other sources of non-point source pollution are atmospheric deposition, in-place contaminants, and natural sources (U.S. EPA, 1994).

The increase of impervious area due to urbanization also increases the volume and rate of urban stormwater runoff. One of the pollutant sources associated with heavy rainfall events is combined sewer overflows (CSOs). Combined sewer systems typically do not cause environmental problems for nearby waterways, but during intense rainfall events water flow in CSOs may exceed the wastewater treatment plant’s capacity (Slone and Evans, 2003; Bas and Baskaran, 2003 ;cited in Moran,2004). As a result of the CSO, raw, untreated human waste is emptied into nearby waterways.

Another stormwater pollutant source associated with rainfall events is the actual rainfall. Studies have revealed that a considerable portion of nitrogen found in stormwater runoff is from atmospheric deposition of nitrogen in rainfall (Line et al., 2002; Wu et al., 1998). Studies in Charlotte, North Carolina found that atmospheric deposition accounted for 10-30% of runoff pollutant loadings for phosphorus and total suspended solids, 30-50% of runoff pollutant loadings for copper and lead, and 70-90% of runoff pollutant loadings for nitrogen (Wu et al., 1998). An estimated 65% of the total phosphorus and nearly 100% of the total Kjeldahl nitrogen entering Irondequoit Creek basin was due to atmospheric deposition (Johnston and Sherwood, 1996; cited in Moran, 2004).

2.3.2 Best Management Practices (BMPs) and their Pollutant Removal Mechanisms.

The term "BMPs" encompasses all measures that remove pollution from runoff. An understanding of the removal mechanisms leads to better application of BMPs to achieve design goals, as well as the refinement of existing types of BMPs and the development of innovative BMPs. Six primary pollutant removal mechanisms are (NCDERN, 2005):

- Sedimentation: Settling is the primary pollutant removal mechanism associated with dry extended detention and wet detention basins. Settling is an important secondary removal mechanism in many other types of BMPs, especially those with stilling basins or other impoundment features (such as low-velocity areas in grass swales, pretreatment areas of infiltration trenches and sediment chambers in water quality inlets).
- Filtration: Filtration is a primary pollutant removal mechanism in infiltration trenches and basins, porous pavement, and bioretention. Filtration is also an important component of other types of BMPs, such as grass swales, filter strips, and other buffers.
- Adsorption: Adsorption also is important in the removal of soluble pollutants from other non-filtering types of BMPs. Phosphorus, for example, has been shown to adsorb to sediment and to settle out of the water column in many BMP studies. Sediments from BMP enhancements, such as shallow wetlands, are excellent media for pollutant adsorption.
- Infiltration: Infiltration is a particularly efficient means of pollutant removal if most of the sediment can be removed from the stormwater flowing into the infiltration area.
- Microbial action: Microbes that break down organic pollutants and transform nutrients colonize many BMPs, especially when organic matter is plentiful, temperatures warm, and conditions moist. Two important microbial processes: nitrification and denitrification.
- Plant uptake: Biological removal can be achieved in many BMPs via "enhancements" by adding artificial wetlands to the lower stages of dry ED basins and adding aquatic and emergent plants to littoral shelves around wet detention basins. Plants convert nutrients into biomass, and when the plants go dormant, the biomass settles into the detritus where the nutrients may be consumed by bacteria and, in the case of nitrogen, removed from the system.

General description, design feature, applications, removal efficiency, and cost for each practice will be discussed in details in the following sections.

2.3.3 Wet pond

General description

Wet ponds (Figure 2.1) can be fairly simple structures composed of a pretreatment basin and a permanent pond basin with an emergency spillway. They may also incorporate more complex devices such as hazardous material traps, spreader and separator boxes, and filtered outfall structures (Harlow et al., 2000).

The primary removal mechanism is settling while stormwater runoff resides in the pool. Nutrient uptake also occurs through biological activity in the pond. While there are several different versions of the wet pond design, the most common modification is the extended detention wet pond, where storage is provided above the permanent pool in order to detain stormwater runoff in order to provide greater settling. Wet ponds are among the most cost-effective and widely used stormwater treatment practices (CWP, 2006).

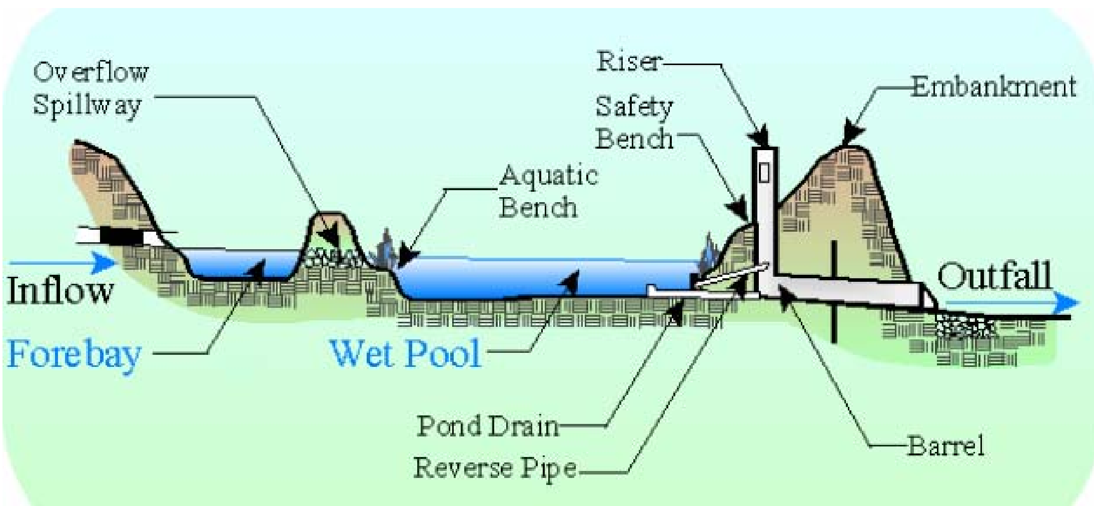

Figure 2.1 Wet Ponds (Harlow et.al. 2000)

Design features

- *Storage*: The permanent pool volume is often defined as the volume equivalent to three times the water quality volume or 12.7 mm of runoff from the contributing drainage area. Consensus is that the larger the permanent pool, the more effective the structure will be (Schueler 1987, and Young et al. 1996). Wet Ponds may be a good option for retrofitting existing dry detention basins. Wet Ponds are often organized into three groups (PCSL, 2006):
  - Wet Ponds primarily accomplish water quality improvement through displacement of the permanent pool and are generally only effective for small inflow volumes (often they are placed offline to regulate inflow).

- Wet Detention Ponds are similar to Wet Ponds but use extended detention as another mechanism for water quality and peak rate control.
- Pocket Wet Ponds are smaller Wet Ponds that serve drainage areas between approximately 20,235 m^2 and 40,470 m^2 and are constructed near the water table to help maintain the permanent pool. They often include extended detention as well.
- Some designers have used wet ponds to act as a water source, usually for irrigation. In this case, the water balance should account for the water that will be taken from the pond. One study conducted in Florida estimated that a water reuse pond could provide irrigation for a 404,682 m^2 golf course at about one seventh the cost of the market rate of the equivalent amount of water (US$40,000 versus US$300,000) (CWP, 2006).

- *Pre-application treatment*: A sediment pretreatment area should be provided with a volume equal to 25 percent of the water quality volume. This recommendation is generally consistent across all sources (Schueler 1987, and Young et al, 1996). A sediment forebay is a small pool (typically about 10% of the volume of the permanent pool) located near the pond inlet. Coarse sediments are trapped in the forebay, and these sediments are removed from the smaller pool on a five to seven year cycle (CWP, 2006).

- *Site-specific characteristics:* Wet ponds need sufficient drainage area to maintain a permanent pool. In humid regions, a drainage area of about 100,000 m^2 is typically needed, but greater drainage areas are needed in arid and semi-arid regions. Wet ponds can be used on sites with an upstream slope up to about 15%. The local slope within the pond should be relatively shallow, however. While there is no minimum slope requirement, there must be enough elevation drop from the pond inlet to the pond outlet to ensure that water can flow through the system by gravity. Wet ponds can be used in almost all soils and geology, with minor design adjustments for regions of karst topography (CWP, 2006).
- *Loading and operating cycle:* Typically, ponds are sized to be equal to the water quality volume (i.e., the volume of water treated for pollutant removal). Designers may consider using a larger volume to meet specific watershed objectives, such as phosphorous removal. Wet ponds should always be designed with a length to width ratio of at least 1.5:1. In addition, the design should incorporate features to lengthen the flow path through the pond, such as underwater berms designed to create a longer flow path through the pond. (CWP, 2006).
- *Vegetation type*: Vegetation is an integral part of a Wet Pond system. Vegetation in and adjacent to a pond may enhance pollutant removal, reduce algal growth, limit erosion, improve aesthetics, create habitat, and reduce water warming (Mallin et al., 2002). Wet

Ponds should have varying depths to encourage vegetation in shallow areas. The emergent vegetation zone (areas not more than 0.46 m deep) generally supports the majority of aquatic vegetation and should include the pond perimeter (PCSL, 2006).

- *Maintenance*: Wet Ponds should have a maintenance plan and privately owned facilities should have an easement, deed restriction, or other legal measure to prevent neglect or removal. During the first growing season or until established, vegetation should be inspected every 2 to 3 weeks. Wet Ponds should be inspected at least 4 times per year and after major storms (greater than 5 cm in 24 hours) or rapid ice breakup.

Application

Wet extended detention ponds can be applied in most regions, with the exception of arid climates. In arid regions, it is difficult to justify the supplemental water needed to maintain a permanent pool because of the scarcity of water. Wet ponds can accept runoff from stormwater hotspots, but need significant separation from groundwater if they are used to treat hotspot runoff. Wet ponds are widely used for stormwater retrofits, and have two primary applications as a retrofit design (CWP, 2006).

Removal efficiencies

The performance of wet ponds varies somewhat more than other BMPs based on the size of the permanent pool and the contributing watershed. The values are given in Table 2.1.

Table 2.2: Pollutant Removal Performance (%): Wet Ponds (*Source*: Harlow et al., 2000).

Pollutant	FHWA Evaluation and Management of Highway Runoff Quality	National Pollutant Removal Performance Database	12.7 mm Runoff per impervious 4047 m^2	Runoff from 25.4 mm* Runoff coefficient * Area	2 Years Runoff Volume
TSS	74	79	60-90	60	85-90
TP	49	49	40-60	35-40	65
TN	34	32	N/A	N/A	N/A

FHWA: USA Federal Highway Administration

Advantages and disadvantages (AES, 2006)

Advantages:

- Improve runoff control, including reductions of overall runoff from adjacent sites with proper design.
- Create wildlife habitat. Aesthetically pleasing.

- May increase property values.
- Requires significantly less expense for maintenance if natural vegetation is used along the banks.

Disadvantages:

- May require approval from dam safety authorities.
- May require maintenance at regular intervals to remove sediments deposited
- If not designed or maintained correctly, could become a mosquito vector.

Application in urban environment

Wet detention basins are applicable in low-density residential, industrial, and commercial developments where enough space and a reliable source of water are available (NCDENR, 2005). It is difficult to use wet ponds in ultra urban areas because enough land area may not be available for the pond. Wet ponds can, however, be used in an ultra-urban environment if a relatively large area is available downstream of the site (CWP, 2006).

Cost

Average typical base capital construction costs for wet ponds basin is US\$27/m^3 (Brown and Schueler, 1997b). Average annual maintenance cost (% of construction cost) is 4.5% (Wiegand et al, 1986; Schueler, 1987).

2.3.4 Dry Extended Detention Pond

General description

Dry extended detention (ED) ponds, (Figure 2.2) is a permanent stormwater management facility that temporarily stores incoming stormwater, trapping suspended pollutants, and reducing the frequency and severity of erosive runoff events (NCDENR, 2005).
The primary means of removing pollutants is sedimentation which results from the stilling effect of detention, allowing heavier sediments to settle out of suspension. The longer the detention time, the greater the pollutant removal will be (Harlow et al., 2000). If detention of the water quality volume can be extended to 48 hours or greater, removal of up to 90 percent of suspended solids is possible (Young et al. 1996). The removal of nutrients is also reasonably effective for detention times of 48 hours or more (Harlow et al., 2000).

Design features

- *Storage*: The discharge structure should be designed to detain the water quality volume for 24 to 48 hours and must have a release rate that will not exacerbate downstream flooding for estimated peak discharges of one or more storm return frequencies.

Detention structures can be used for watersheds of 40,000 m^2 to 12,000 m^2 (Harlow et al., 2000).

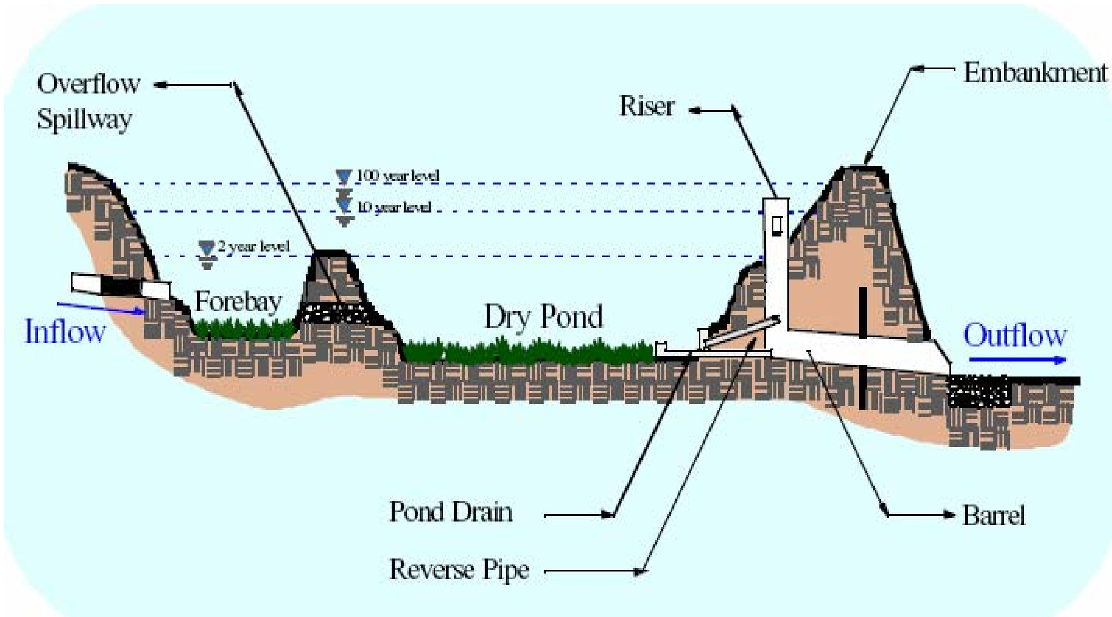

Figure 2.2 Dry extended detention ponds (Harlow et al., 2000).

- *Pre-application treatment*: Pretreatment is intended to capture and remove coarse sediment particles before they enter the practice. Maintenance burden of the pond is reduced when these particles are removed from runoff before they reach the pool. In ponds, pretreatment is achieved with a sediment forebay. A sediment forebay is a small pool (typically about 10% of the volume of water to be treated for pollutant removal) (CWP, 2006).
- *Site-specific characteristics*: The soil should have low infiltration rates if detention occurs over ground water reservoirs that could be contaminated (Harlow et al., 2000). Except for the case of hotspot runoff, the only consideration regarding groundwater is that the base of the extended detention facility should not intersect the groundwater table. Dry extended detention ponds can be used on sites with slopes up to about 15% (CWP, 2006).
- *Loading and operating cycle*: Land requirement is typically 0.5 to 2.0 percent of drained area .The inflow and outflow hydrographs must be computed and the outlet structure hydraulics must be evaluated to design the dry ED basin for controlling stormwater quantity. Dry ED basins typically are designed to control runoff peak rates for rainfall events with return frequencies of. By causing turbulence and eddies in the flow, flow short-circuiting can interfere with the function of the basin outlet system and should therefore be minimized. The most direct way of minimizing short-circuiting

is to maximize the distance between the riser and the inlet. Larger length to width ratios should be used if sedimentation of particulates during low flows is desirable (NCDENR, 2005).

- *Vegetation type*: It is recommended that basin bottoms be vegetated in a diverse native planting mix to reduce maintenance needs, promote natural landscapes, and increase infiltration potential. Vegetation may include trees, woody shrubs and meadow/wetland herbaceous plants. Meadow grasses or other deeply rooted herbaceous vegetation is recommended on the interior slope of embankments (PCSL, 2006).
- *Maintenance*: Maintenance is necessary to ensure proper functionality of the extended detention basin and should take place on a quarterly basis. A basin maintenance plan should be developed which includes the following measures (PCSL, 2006):
  - All basin structures expected to receive and/or trap debris and sediment should be inspected for clogging and excessive debris and sediment accumulation at least four times per year, as well as after every storm greater than 25 mm.
  - Structures include basin bottoms, trash racks, outlets structures, riprap, and inlets.
  - Sediment removal should be conducted when the basin is completely dry.

Application (PCSL, 2006)

- Low Density Residential Development Urban Areas.
- Industrial and Commercial Development.

Removal efficiencies

As seen in Table 2.3, the performance of extended detention ponds increases significantly for TSS with time. According to most sources, there is little significant change in the removal of other pollutants after a 24-hour period.

Advantages and disadvantages (EAS, 2006)

Advantages:

- Reduces peak flow rate and energy of stormwater discharges, therefore limiting downstream erosion and scouring.
- Good potential for removal of sediments.
- Can be used for recreation when dry.
- Can serve as green space, supporting wet prairie functions and wildlife habitat.

Disadvantages:

- Generally not prescribed for drainages less than 40,000 m^2.
- Potential for clogging of outlets.
- Can be considered unattractive by residents if not designed or maintained correctly.
- Limited ability to remove pollutants.

Table 2.3: Pollutant Removal Performance (%): Extended Detention Ponds (Harlow et al., 2000).

Pollutant	FHWA Evaluation and Management of Highway Runoff Quality (HRT)			National Pollutant Removal Performance Database	Controlling Urban Runoff (Schueler 1996) (HRT)		
	12 hr	24 hr	48 hr	N/A	6 h	12 hr	24 hr
TSS	68	75	90	61	55	69	75
TP	42	45	50	20	25	44	45
TN	28	32	40	31	22	25	32

FHWA: USA Federal Highway Administration

Application in urban environment

It is difficult to use dry extended detention ponds in the ultra urban watershed because of the land area each pond consumes. They can, however, be used in these environments if a relatively large area is available downstream of the site (CWP, 2006).

Cost

Average typical base capital construction costs for dry extended detention basins is US$27/$m^3$ (Brown and Schueler, 1997b). Average annual maintenance cost (% of construction cost) is 0.7% (Livingston et al, 1997; Brown and Schueler, 1997b).

2.3.5 Stormwater Wetland

General description

Stormwater wetlands (Figure 2.3) are engineered wetland systems designed to treat runoff. They are typically designed to provide some of the functions of natural wetlands, e.g., wildlife habitat, in addition to mitigate the impacts of urbanization on stormwater quality and quantity (USEPA, 2005, NCDENR, 2005). Stormwater wetlands provide an efficient method for removing a wide variety of pollutants, such as suspended solids, nutrients (nitrogen and phosphorus), and petroleum compounds.

Pollutant removal in a wetland is accomplished by physical treatment, which includes vitalization and sedimentation, adsorption, and filtration. In addition, chemical processes such as chelation, precipitation, and chemical adsorption occur in wetlands. These chemical processes, paired with biological processes like decomposition, nutrient utilization, and degradation are the primary advantage of the wetland over a wet pond (Harlow et al. 2000).

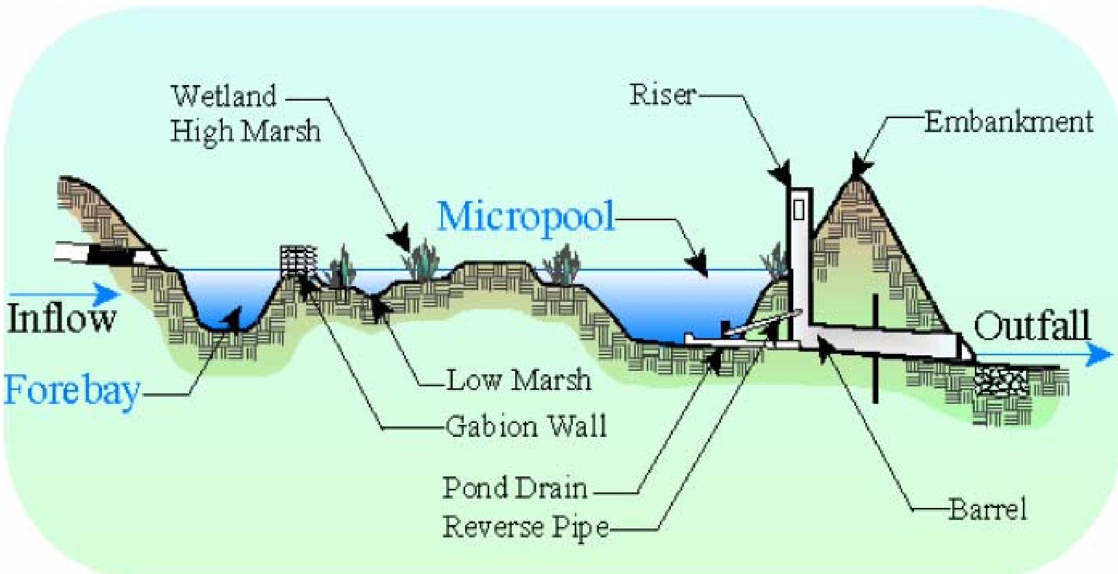

Figure 2.3 Stormwater wetlands (Harlow et al. 2000)

Design features

- *Storage*: Some typical design features include (CWP, 2006):
  - The surface area of wetlands should be at least 1% of the drainage area.
  - Wetlands should have a length to width ratio of at least 1.5:1. Making the wetland longer than it is wide helps prevent "short circuiting".
  - Effective wetland design depends on "complex microtopography". In other words, wetlands should have zones of both very shallow (<15 cm) and moderately shallow (<46 cm) wetlands are incorporated, using underwater earth berms to create the zones. This design will provide a longer flow path through the wetland to encourage settling, and provides two depth zones to encourage plant diversity.
- *Pre-application treatment*: Pretreatment is used to settle out coarse sediment particles prior to entry in the main wetland cell. By removing sediments before they reach the wetland, the maintenance burden of the wetland is reduced. In wetlands, pretreatment is achieved with a sediment forebay. A sediment forebay is a small pool (typically about 10% of the volume of the permanent pool). Coarse particles remain trapped in

the forebay, and maintenance is performed on this smaller pool, eliminating the need to dredge or clean out sediments from the entire wetland (CWP, 2006).

- *Site-specific characteristics:* Stormwater wetlands are feasible at most sites and drainage areas where there is enough rainfall and/or snowmelt to maintain a permanent pool. In areas with highly permeable soils, other impermeable barriers, such as synthetic liners or clay, sometimes can be used to maintain enough water or moisture to support the wetland (USEPA, 2005). Unless they receive hotspot runoff, wetlands can often intersect the groundwater table.

- *Loading and operating cycle*: Wetlands are useful water quality tools for watersheds of 20,000 to 200,000 m^2 in size. The depth of the permanent pool should be 1 m to 2 m; shallower depths may result in resuspension of pollutants. For safety reasons, a moderately sloped bench (3-4 %), at least 3.1 m wide, should be provided and the 2 m depth should be considered maximum (Harlow et al. 2000).

- *Vegetation type*: Vegetation can be established by allowing volunteer (i.e. not planted) vegetation to become established, or, from planting nursery stock (AES, 2006). Stormwater wetlands should initially be planted with emergent plants and woody shrubs, and the wetlands should be allowed to succeed to a system dominated by woody shrubs and trees. (USEPA, 2005).

- *Maintenance*: Maintenance requirements are summarized in Table 2.4.

Table 2.4: Maintenance: Stormwater wetlands (CWP, 2006).

Activity	Schedule
Replace wetland vegetation to maintain at least 50% surface area coverage in wetland plants after the second growing season.	One-Time (after construction)
Inspect for invasive vegetation and remove where possible.	Semi-Annual Inspection
• Inspect for damage to the embankment and inlet/outlet structures. Repair as necessary. • Monitor for sediment accumulation in the facility and forebay.	Annual Inspection
• Clean and remove debris form inlet and outlet structures • Mow side slopes.	Frequent (3-4 times/year) Maintenance
• Supplement wetland plants if a significant portion have not established (at least 50% of the surface area). • Harvest wetland plants that have been "choked out" by sediment build-up.	Annual Maintenance (if needed)
Removal of sediment from the forebay.	5 to 7 year Maintenance
Monitor sediment accumulations, and remove sediment when the pool volume has become reduced significantly, plants are "choked" with sediment, or the wetland becomes eutrophic.	20 to 50 year Maintenance

Removal efficiencies

The performance (Table 2.5) of wetlands varies somewhat more than other BMPs based on the size of the permanent pool and the contributing watershed.

Table 2.5: Pollutant Removal Performance (%): Stormwater Wetland (Harlow et al., 2000).

Pollutant	FHWA Evaluation and Management of Highway Runoff Quality	National Pollutant Removal Performance Database	12.7 mm Runoff per 4047 m^2	12.7 mm Runoff per impervious 4047 m^2	2.5 Times the Runoff of the Mean Storm
TSS	74	79	60-90	60	85-90
TP	49	49	40-60	35-40	65
TN	34	32	N/A	N/A	N/A

FHWA: USA Federal Highway Administration

Table 2.6 indicates the average range of pollutant removal efficiencies that have been reported in the literature for constructed wetlands receiving highway runoff in the UK, France, Canada and the United States. Two types of wetland are represented in Table 2.6, Free Water Surface (FWS) or Subsurface Flow (SSF) systems. In FWS systems, the flow of water is above the ground, and plants are rooted in the sediment layer at the base of water column. In SSF systems, water flows though a porous media such as gravels or aggregates, in which the plants are rooted.

Table 2.6: Pollutant Removal Performance (%): Constructed Wetland for high way runoff (MU, 2003)

Pollutant	Sub Surface Flows (SSF)	Free Surface Flows (FWS)
TSS	85 (67-97)	73 (13-99)
TP	50 (20-97)	43 (2-98)
TN	44 (25-98)	33 (10-99)

Advantages and disadvantages (AES, 2006)

Advantages:

- Improvements in downstream water quality through settlement of particulates, and removal of pollutants.
- Flood attenuation and reduction of peak discharge.
- Enhancement of biological diversity and wildlife habitat in urban areas.
- Aesthetic enhancement and valuable addition to community green space.
- Relatively low maintenance costs.

Disadvantages:

- May be difficult to maintain vegetation under a variety of flow conditions.
- May require larger land requirements than other BMPs.
- Pollutant removal efficiencies may be low until vegetation is established.
- Relatively high construction costs.
- If not designed properly, wetlands may not receive favorable community attention.

Application in urban environment

It is difficult to use wet ponds in the ultra urban watershed because of the land area each wetland consumes. They can, however, be used in these environments if a relatively large area is available downstream of the site (CWP, 2006).

Cost

Average typical base capital construction cost for stormwater wetlands is US$33/$m^3$ (Brown and Schueler, 1997b). Average annual maintenance cost (% of construction cost) is 2% (Livingston et al, 1997; Brown and Schueler, 1997b).

2.3.6 Infiltration Basin

General description

An infiltration basin is a surface structure that captures a predetermined water quality volume and treats the water by allowing it to infiltrate into the native soil (Harlow et al., 2000) or through a specially constructed under-drain system containing gravel and/or sand filter beds (MD, 2003). Infiltration basins (Figure 2.4) are impoundments created by excavation or creation of berms or small dams. They are typically flat-bottomed with no outlet and are designed to temporarily store runoff generated from adjacent drainage areas (from 8,094 to 202,350 m^2, depending on local conditions). Infiltration basins are often used as an off-line system for treating the first flush of runoff flows or the peak discharges of the two-year storm (USEPA, 2005), because sediment accumulation and particulates from stormwater runoff can clog the system (FHWA, 2005).

Design features

- *Storage*: Runoff gradually infiltrates through the bed and sides of the basin, ideally within 72 hours, to maintain aerobic conditions and ensure that the basin is ready to receive runoff from the next storm (USEPA, 2005). Infiltration basins only contain water immediately after a storm and should be dry within 48 to 72 hours depending on the soil and the desired drawdown time.

- *Pre-application treatment*: Infiltration basins should only be used as part of a "treatment train," where soluble organic substances, oils, and course sediment are

removed by other management practices prior to stormwater entering the infiltration basin (MDEQ, 1999). In order to ensure that pretreatment mechanisms are effective, designers should incorporate "multiple pretreatment," using practices such as grass swales, sediment basins, and vegetated filter strips in series, prior to the infiltration basin (CWP, 2006).

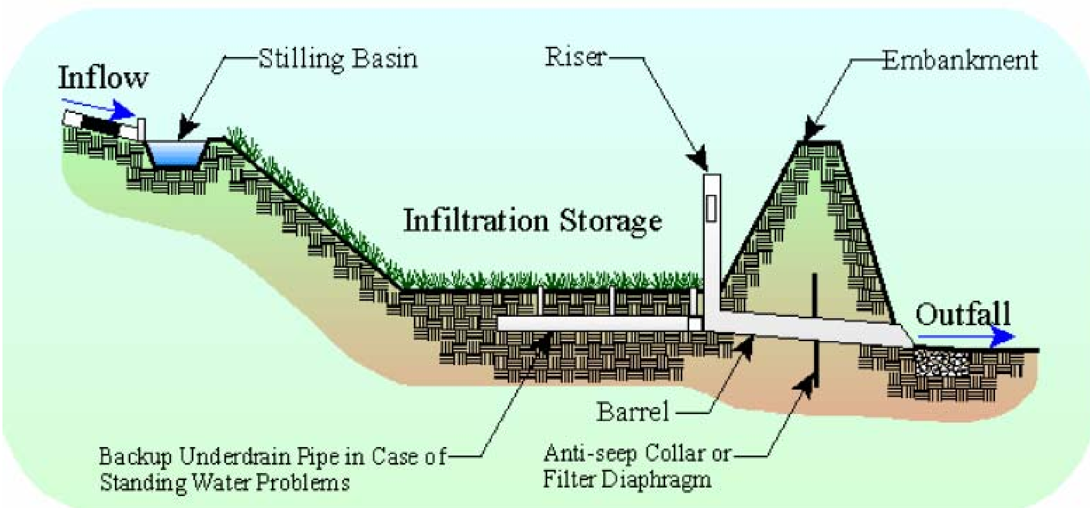

Figure 2.4 : Infiltration Basin (Harlow et. al. 2000)

- *Site-specific characteristics:* Infiltration practices need to be located extremely carefully. In particular, designers need to ensure that the soils on the site are appropriate for infiltration, and that designs minimize the potential for groundwater contamination, and long term maintenance problems (CWP, 2006). The following site characteristics are required for practical usage of infiltration basins (USEPA, 2005):
  - A minimally compacted sub-base;
  - Minimum of 1.22 m or more of soil cover over the substrate;
  - Slope of the basin should be less than 5 percent; depth of the basin should be limited to provide drawdown times of 48-72 hours;
  - Provide pretreatment equal to 25 % of the basin volume;
  - Soils should have an infiltration rate of 12.7 mm/hr; and
  - Provide an emergency spillway to bypass volumes greater than the designed water quality volume
- *Loading and operating cycle*: The infiltration rate for the soil should range between 12.7 mm and 76.2 mm per hour. Detailed soil tests are needed to determine if hardpans or other confining layers are present. In addition, the soils should have no greater than 20% clay content, and less than 40% silt/clay content (CWP, 1998). The rate at which

water percolates into the ground can be estimated by using Darcy's Law (NCDENR, 2005).

- *Vegetation type*: Vegetation is a key to success of the infiltration basin. Deep-rooting vegetation will enhance infiltration of water while also staying well anchored against disturbance from water or other factors (AES, 2006). In infiltration basins, the most important purpose of vegetation is to reduce the tendency of the practice to clog. Upland drainage needs to be properly stabilized with a thick layer of vegetation, particularly immediately following construction. In addition, providing a thick turf at the basin bottom helps encourage infiltration and prevent the formation of rills in the basin bottom (CWP, 2006).
- *Maintenance*: Maintenance requirements are summarized as follows (NCDENR, 2005):
  - Grass areas, grassed swales, and filter strips leading to infiltration devices should have a dense vegetation cover and be mowed at least twice a year.
  - Sediment deposits should be removed from pretreatment devices at least annually.
  - The infiltration facility must be dismantled and reconstructed when the infiltration rate drops to unacceptable levels.

Application

Infiltration basins can be utilized in most regions, with some design modifications in cold and arid climates. In regions of karst topography, these infiltration basins should not be applied due to concerns of sink hole formation and groundwater contamination. Infiltration basins should never receive runoff from stormwater hotspots, unless the stormwater has already been fully treated by another stormwater treatment practice. This is due to potential groundwater contamination (CWP, 2000).

Infiltration Basins can be incorporated into new development. Ideally, existing vegetation can be preserved and utilized as the infiltration area. Existing grassed areas can be converted to an infiltration basin. If the soil and infiltration capacity is determined to be sufficient, the area can be enclosed through creation of a berm and runoff can be directed to it without excavation (PCSL, 2006).

Removal efficiencies

There is very little in the literature to substantiate the performance levels of infiltration basins. The values (Table 2.7) given in the 1996 FHWA study are repeated from Schueler's 1987 document. In the section on infiltration basins, Schueler clearly states that the values are estimates of removal rates that might be achieved under various sizing rules. The National Pollutant Removal Performance Database (Winer, 2000) provides no values for infiltration basins. In this publication they caution that while infiltration practices tend

to show very good results, it is difficult to monitor infiltration BMPs, and very few have actually been monitored.

Table 2.7: Pollutant Removal Performance (%): Infiltration Basins (Harlow et al., 2000).

Pollutant	FHWA Evaluation and Management of Highway Runoff Quality	National Pollutant Removal Performance Database	12.7 mm Runoff per impervious 4047 m^2	Runoff from 25.4 mm* Runoff coefficient * Area	2 Years Runoff Volume
TSS	75	N/A	75	90	99
TP	50-55	N/A	50-55	60-70	65-75
TN	45-55	N/A	45-55	55-60	60-70

FHWA: USA Federal Highway Administration

Advantages and disadvantages (AES, 2006)

Advantages:

- Reduces peak flow rate and energy of stormwater discharges, therefore limiting downstream erosion and scouring.
- Can be used for recreation when dry.
- Can help to maintain baseflow of nearby streams.
- Can serve as greenspace, supporting wet prairie functions and wildlife habitat.
- Reduces local flooding.

Disadvantages:

- Generally not prescribed for drainages greater than 40,468 m^2.
- Potential for fouling infiltration capacity of the soil if runoff is sediment-laden.
- Can be considered unattractive by residents if not designed or maintained correctly.

Application in urban environment

Infiltration basins can rarely be applied in the ultra urban environment. Two features that can restrict their use are the potential of infiltrated water to interfere with existing infrastructure, and the relatively poor infiltration capacity of most urban soils. In addition, while they consume about the same amount of space as infiltration trenches, they need a continuous, relatively flat area. Thus, it is more difficult to fit them into small unusable areas on a site.

Cost

Average typical base capital construction cost for infiltration basins is US$46/$m^3$ (SWRPC, 1991). Average annual maintenance cost (% of construction cost) is 6% (Livingston et al, 1997; Schueler, 1997; SWRPC, 1991; Wiegand et al, 1986).

2.3.7 Infiltration Trench

General description

Infiltration trenches are shallow, linear excavations backfilled with coarse material (Figure 2.5) (Harlow et al., 2000). Infiltration trenches are filled with stone or rubble and in comparison to soakaways require lower volumes of infiltration material for a given water inflow (ME, 2003). Infiltration trenches are shallow (0.61 to 3.05 m deep) excavated ditches with relatively permeable soils that have been backfilled with stone to form an underground reservoir.

The trench surface can be covered with a grating or can consist of stone, gabion, sand, or a grass covered area with a surface inlet. Runoff diverted into the trench gradually infiltrates into the subsoil and, eventually, into the ground water. More expensive than pond systems in terms of cost per volume of runoff treated, infiltration trenches are best-suited for drainage areas of less than 20,234 to 40,468 m^2 (USEPA, 2005). In developing areas, infiltration trenches can help minimize the change in predevelopment hydrology by helping to maintain interflow and recharge.

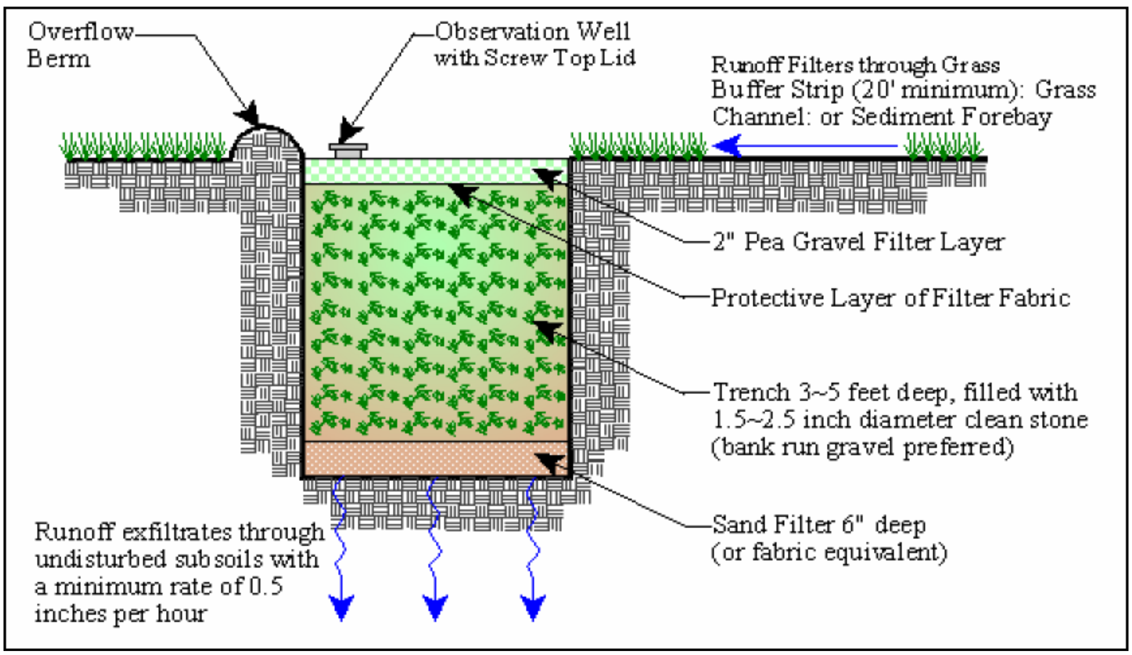

Figure 2.5: Infiltration Basin (Harlow et. al., 2000)

Design features

- *Storage*: Storage volume should be based on the median design storm for the region, and is dependent on the coarseness of the backfill material. LCRA suggests a value of 35 percent of the excavated volume of the trench as a reasonable value (Harlow et al., 2000). Recommended drawdown time, 48 hours (LCRA 1998) to 72 hours (Schueler,

1987), dependent on the probability of the recurrence of a storm event that would produce runoff equal to the storage volume of the infiltration trench.

- *Pre-application treatment*: Pretreatment controls such as vegetated filter strips should be incorporated into the design to remove sediment and reduce clogging of soil pores (USEPA, 2005). The vegetative filter strip should be at least 6 meters wide between the runoff source and the trench (MDEQ, 1999). In order to make pretreatment effective, designers should incorporate "multiple pretreatment" into every trench, using grass swales, vegetated filter strips, sedimentation basins, or a plunge pool in series (CWP, 2006).

- *Site-specific characteristics:* Designers always need to provide significant separation (0.6 to 1.5 meters) from the bottom of the infiltration trench and the seasonally high ground water table, to reduce the risk of contamination and trench failure. In addition, infiltration practices should be separated at least 46m from adjunct drinking water wells. Infiltration trenches generally are applied to relatively small sites (less than 20,234 m^2) that have relatively high impervious cover. Infiltration trenches should be placed on flat ground, but the slopes of the site draining to the practice can be as high as 15%. Soils must be significantly permeable to ensure that trenches can infiltrate quickly enough to reduce the potential for clogging. The infiltration rate should range between 13 and 76 mm/hr (CWP, 2006). In addition, the soils should have no greater than 20% clay content, and less than 40% silt/clay content (CWP, 1998).

- *Loading and operating cycle*: Designers need to be particularly careful in ensuring that channels leading to infiltration trenches are designed to minimize erosion. Infiltration trenches should be designed to treat only small storms, (i.e., only for water quality or recharge). Thus, trenches should be designed "off-line," using a structure to divert only small flows to the trench. Finally, the sides of an infiltration trench should be lined with a geotextile fabric to prevent adjacent soils from clogging the matrix.

- *Maintenance*: Maintenance requirements are summarized as follows (NCDENR, 2005):

  - The top several millimeters of aggregate and the filter cloth along the top of the trench should be replaced annually or at least when the facility shows evidence that infiltration rates have declined. Proper disposal of the materials removed is necessary; the aggregate and cloth should be appropriately packaged and delivered to the local landfill, if the operating authority approves the disposal.

  - The surface of infiltration trenches must be kept in good condition. Colonization by grass or other plants should be discouraged, since this can lead to reduced surface infiltration rates. In many instances, it is convenient to cover infiltration trenches with concrete grid pavers or similar permeable paving systems that can be removed easily and replaced as necessary to service the trench.

Application

Infiltration basins can be utilized in most regions, with some design modifications even in cold and arid climates. In regions of karst topography, these infiltration trenches should not be applied due to concerns of sink hole formation and groundwater contamination. Infiltration basins should never receive runoff from stormwater hotspots, unless the stormwater has already been fully treated by another treatment unit. This is due to potential groundwater contamination. Infiltration trenches may be used as a stormwater retrofit (CWP, 2006). Roof leaders may be connected to Infiltration Trenches. An Infiltration Trench may be preceded by or used in combination with a Vegetative Filter, Grassed Swale, used to reduce sediment levels (PCSL, 2006).

Removal efficiencies

According to the literature, pollutant performance of infiltration trenches varies with design, soil type, backfill, and age. The current EPA Pollutant Removal Database (EPAPRD) gives a TSS removal rate of 100 percent. However, data points are limited and there is no allowance for aging. The earlier values by Schueler (1987) and others seem to be more reasonable for estimating purposes. The values from Schueler, EPA, and FHWA are shown in Table 2.8.

Table 2.8: Pollutant Removal Performance (%): Infiltration Trenches (Harlow et al., 2000).

Pollutant	FHWA Evaluation and Management of Highway Runoff Quality	National Pollutant Removal Performance Database	12 mm Runoff per impervious 4047 m^2	Runoff from 25.4 mm* Runoff coefficient * Area	2 Years Runoff Volume
TSS	99	100	60-80	80-100	80-100
TP	65-75	42	40-60	40-60	60-80
TN	60-70	42	40-60	40-60	60-80

FHWA: USA Federal Highway Administration

Advantages and disadvantages (AES, 2006)

Advantages:

- Effectively removes or reduces many pollutants, including suspended solids, bacteria, and trace metals.
- Reduces runoff volumes during storm events.
- Increases baseflow in nearby streams.

Disadvantages:

- Generally not prescribed for drainages greater than 20,234 m^2.
- Potential for fouling infiltration capacity of the soil if runoff is sediment-laden.

- Infiltration trenches may require periodic maintenance to prevent clogging.

Application in urban environment

Infiltration trenches can seldom be applied in the ultra urban environment. The two main reasons are the potential of infiltrated water to interfere with existing infrastructure, and the relatively poor infiltration capability of most urban soils (CWP, 2006). Because of the cost and the need for pretreatment, infiltration trenches have very limited application in highway transportation (Harlow et al., 2000).

Cost

Average typical base capital construction cost for infiltration trenchs is US\$141/m^3 (SWRPC, 1991). Average annual maintenance cost (% of construction cost) is 12% (Schueler, 1997; SWRPC, 1991).

2.3.8 Bioretention

General description

Bioretention (Figure 2.6) is the use of plants and soils for removal of pollutants from stormwater runoff via adsorption, filtration, sedimentation, volatilization, ion exchange, and biological decomposition. In addition, bioretention provides landscaping and habitat enhancement benefits (NCDENR, 2005). They are commonly located in parking lot islands or within small pockets in residential land use (CWP, 2006). Treated water is allowed to infiltrate into the surrounding soil, or is collected by an underdrain system and discharged to the stormwater system or directly to receiving waters (AES, 2006).

Design features

- *Storage:* Prince George's County, Maryland, Department of Environmental Resources (1993), suggests a design based on a four-day maximum ponding period (appropriate for the Mid- Atlantic region). This four-day period is based on hydrologic, horticultural, and maintenance constraints such as plant tolerance of flooded conditions and mosquito-breeding concerns. Other considerations include infiltration rates for the root zone, sand layer, and in-situ material (USEPA, 2005). The volume of a Rain Garden has 3 components (PCSL, 2006):

  Bioretention Volume = Surface Storage Volume + Soil Storage Volume

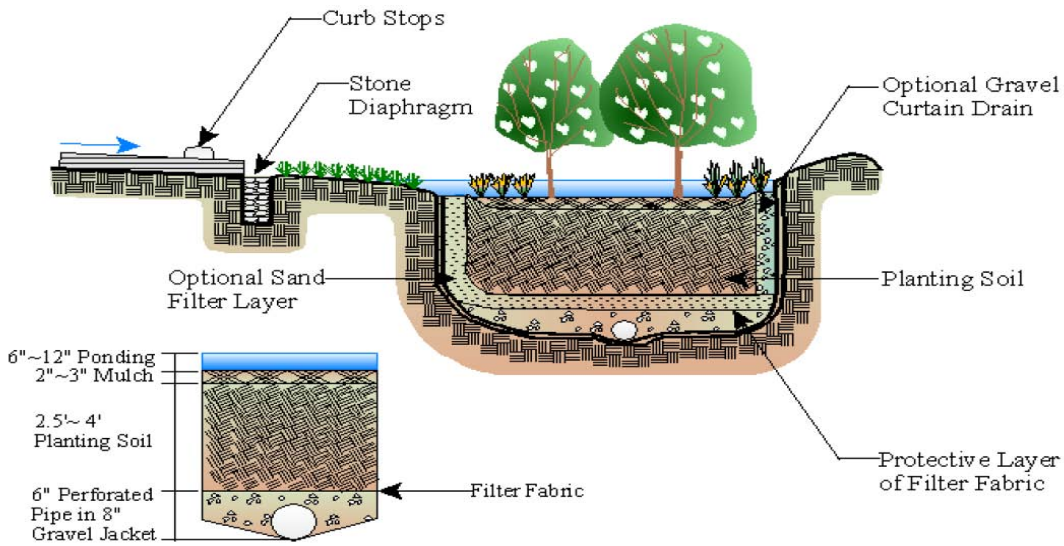

Figure 2.6: Bioretention (Harlow et al., 2000)

- *Pre-application treatment*: Incorporating pretreatment helps to reduce the maintenance burden of bioretention, and reduces the likelihood that the soil bed will clog over time. Several different mechanisms are used to provide pretreatment in bioretention areas. Runoff can be directed to a grass channel or filter strip to settle out coarse sediments before the runoff flows into the filter bed of the bioretention area. Other features may include a pea gravel diaphragm, which acts to spread flow evenly and drop out larger particles. (CWP, 2006).

- *Site-specific characteristics:* Bioretention areas should usually be used on small sites (i.e., 20,234 m^2 or less). Bioretention areas are best applied to relatively shallow slopes (usually about 5%). Bioretention areas can be applied in almost any soils or topography, since runoff percolates through a made soil bed, and is returned to the stormwater system. Bioretention should be separated from the watertable to ensure that the groundwater never intersects with the bottom of the bioretention area, which prevents possible groundwater contamination and practice failure (CWP, 2006). Bioretention should not be used in areas (USEPA, 2005): with mature trees; with slopes greater than 20 percent; with a water table within 2 meter of the land surface; with easily erodible soils; below outfalls; where concentrated flows are discharged; or where excavation or cutting will occur.

- *Loading and operating cycle*: Surface area is dependent upon storage volume requirements but should generally not exceed a maximum loading ratio of 5:1 (impervious drainage area to infiltration area). Surface Ponding depth should not exceed 15 cm in most cases and should empty within 72 hours (PCSL, 2006).

- *Vegetation type*: Typically, native floodplain plant species are best suited to the variable environmental conditions encountered in a bioretention. If shrubs and trees are

included in a bioretention area (which is recommended), at least three species of shrub and tree should be planted at a rate of approximately 700 shrubs and 300 trees per 4046 m^2 (shrub to tree ratio should be 2:1 to 3:1) (PCSL, 2006). Plants should be selected that can withstand the hydrologic regime they will experience (i.e., plants that tolerate both wet and dry conditions) (CWP, 2006).

- *Maintenance*: Properly designed and installed bioretention require some regular maintenance (PCSL, 2006).
  - While vegetation is being established, pruning and weeding may be required. Weeds should be removed thereafter by hand.
  - Detritus may also need to be removed approximately twice per year. Perennial plantings may be cut down at the end of the growing season.
  - Mulch should be re-spread when erosion is evident and be replenished annually. Once every 2 to 3 years the entire area may require mulch replacement.
  - Bioretention should be inspected at least two times per year for sediment buildup, erosion, vegetative conditions, etc.

Application

Bioretention areas can be used in a variety of applications (PCSL, 2006): Residential On-lot; Tree and Shrub Pits; Roads and highways; Parking Lots; Parking Lot Island Bioretention; Commercial/Industrial/Institutions; and Roof leader connection from adjacent building.

Removal efficiencies

The biofiltration concept is included in practically all recent literature on BMPs. Little pollutant removal data has been collected on the pollutant removal effectiveness of bioretention areas. One study has been conducted (Davis et al., 1998). The data from this study is presented in Table 2.9.

Table 2.9: Pollutant Removal Performance (%): Bioretention (CWP 2006).

Pollutant	Pollutant Removal (%)
TSS	81
TP	29
TN	49

As for the performance of bioretention areas (USEPA, 2005), in this research study, simulated runoff was pumped continuously into an area of 5 m^2 in six bioretention cells,

and effluent samples were collected from the perforated drainpipes underlying the bioretention media. All six bioretention facilities showed greater than 99 percent removal efficiency for oil and grease. Total lead removal efficiency decreased when the TSS level in the effluent increased because lead was adsorbed onto the surface of the solids. TSS removal ranged from 72 to 99 percent, and lead removal rates ranged from 80 to 100 percent. For total phosphorus, the removal efficiency was found to be highly variable, ranging from 37 to 99 percent. Nitrate-nitrogen and ammonium nitrogen removal efficiencies ranged from 2 to 7 percent and 5 to 49 percent, respectively.

Advantages and disadvantages (AES, 2006)

Advantages

- Provides effective stormwater flood control by slowing down runoff and increasing water infiltration into the soil.
- Minimally consumes land.
- Provides aesthetic enhancement.
- Increases groundwater recharge.
- Can be used as a stormwater retrofit.

Disadvantages:

- Should not be installed until the entire contributed drainage area has been stabilized.
- Require proper plant selection and maintenance.
- Susceptible to clogging by sediment, may require pretreatment.
- Treat a relatively small drainage area.

Application in urban environment

Bioretention facilities are ideally suited to many ultra urban areas, such as parking lots. While they consume a fairly large amount of space (approximately 5% of the area that drains to them), they can fit into existing parking lot islands or other landscaped areas (CWP, 2006).

Cost

Average typical base capital construction costs for a bioretention is US$187/m^3 (Brown and Schueler, 1997b). Average annual maintenance cost (% of construction cost) is 6% (SWRPC, 1991).

2.3.9 Filtration basins and Sand Filters

General description

Filtration basins are impoundments lined with a filter medium such as sand or gravel. Runoff drains through the filter medium and through perforated pipes into the subsoil. Detention time is typically four to six hours. Sediment-trapping structures are often used to prevent premature clogging of the filter medium (NVPDC, 1980; Schueler et al., 1992). Sand filters are usually two-chambered practices: the first is a settling chamber and the second is a filter bed filled with sand or another filtering medium. As runoff flows into the first chamber, large particles settle out and finer particles and other pollutants are removed as runoff flows through the filtering medium (CWP, 2006).

There are several modifications of the basic sand filter design, including the surface sand filter, underground sand filter, perimeter sand filter, organic media filter, and multi-chambered treatment train (Robertson et al., 1995). All of these filtering practices operate on the same basic principle. Modifications to the traditional surface sand filter were made primarily to fit sand filters into more challenging site designs (e.g., underground and perimeter filters) or to improve pollutant removal (e.g., organic media filter) (CWP, 2006). The following are design variations for sand filtration devices (USEPA, 2005):

(1) Surface sand filter. The surface sand filter (Figure 2.7) is an aboveground filter design. Both the filter bed and the sediment chamber are aboveground. The surface sand filter is designed as an off-line practice; only the water quality volume is directed to the filter. The surface sand filter is the least-expensive filter option and has been the most widely used.

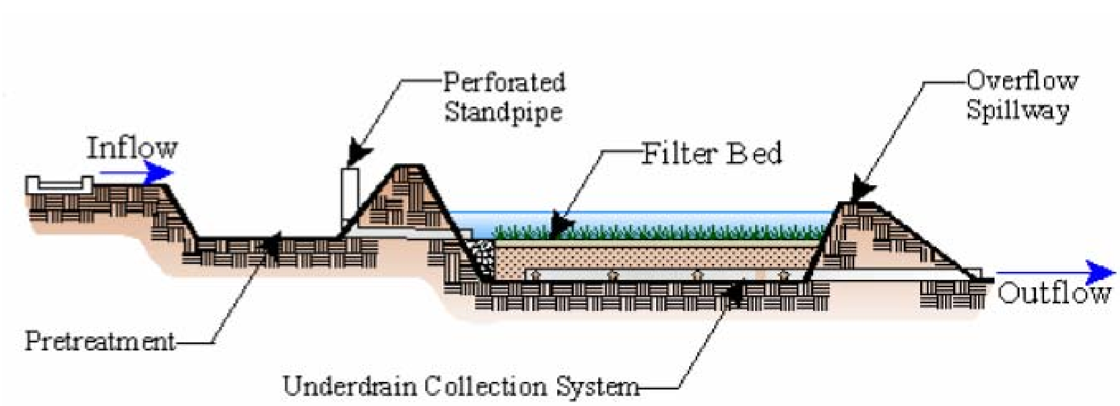

Figure 2.7: Surface (Austin) Sand Filter (Harlow et al., 2000).

(2) Underground sand filter. The underground sand filter is a modification of the surface sand filter, where all of the filter components are underground. Like the surface sand filter, this practice is an off-line system that receives only flows from small rainstorms. Underground sand filters are expensive to construct but consume very little space. They

are well-suited to highly urbanized areas, and often included in groups of practices known as "ultra-urban BMPs."

(3) Perimeter sand filters. The perimeter sand filter also includes the basic design elements of a sediment chamber and a filter bed. In this design, however, flow enters the system through grates, usually at the edge of a parking lot. The perimeter sand filter is the only filtering option that is on-line; all flow enters the system, but a bypass to an overflow chamber prevents system flooding. One major advantage of the perimeter sand filter design is that it requires little hydraulic head and thus is a good option in areas of low relief.

(4) Organic media filter. Organic media filters are essentially the same as surface filters, with the sand replaced with or supplemented by another medium. Two examples are the peat/sand filter (Galli, 1990) and the compost filter system. It is assumed that these systems will provide enhanced pollutant removal for many compounds because of the increased cation exchange capacity achieved by increasing organic matter content.

(5) Exfiltration/partial exfiltration. In exfiltration designs, all or part of the underdrain system is replaced with an open bottom that allows infiltration to the ground water.

Design features

- *Storage*: Sediment Chamber; Volume = 36.86 m^3 / ha of drainage area. Sand Filter Chamber: Filter volume = 36.86 m^3 / ha of drainage area (NCDENR, 2005). Forty to 48 hours is reasonable for the water to pass through the filter bed (Harlow et al., 2000).
- *Pre-application treatment*: Pretreatment is a critical component of any stormwater treatment practice. In sand filters, pretreatment is achieved by a sedimentation chamber that precedes the filter bed. In this chamber, the coarsest sediment particles settle out before they reach the filter bed. Pretreatment reduces the maintenance burden of sand filters by reducing the potential for clogging of the filter bed surface (CWP, 2006). Two types of pretreatment designs are used for sand filters; Full Sedimentation, and Partial Sedimentation (Harlow et al., 2000).
- *Site-specific characteristics:* The most desirable sites for sand filters are those with slopes in the range of 3 to 5 percent and sufficient right-of-way to allow all earthen containment. Rocky, karst sites will complicate excavation. Therefore, basins must be lined to prevent contamination of the groundwater. Filtration structures must not encroach on natural wetlands (Harlow et al., 2000). Designers should provide at least two feet of separation between the bottom of the filter and the seasonally high groundwater table (CWP, 2006).
- *Loading and operating cycle*: The infiltration rate through the filter medium should be established by lab testing the proposed material. Experience in the Austin district

suggests that there is such wide variation in the performance of natural materials that testing is the only way to determine the infiltration rate (*k*). For preliminary estimates, a value of 1 m/day can be used. This is based on testing conducted by the City of Austin in 1988 (Harlow et al., 2000).

- *Vegetation type*: Grass should be established on the filter bed. For most situations, sodding over the bed should be avoided since this will likely introduce clay soils and impair the permeability of the sand bed. The sand bed should be seeded during the growing season with an appropriate seed mix (Harlow et al., 2000).
- *Maintenance*: The maintenance guidelines are as follows (NCDENR, 2005):
  - The sediment chamber outlet devices should be cleaned or repaired when drawdown times within the chamber exceed 24 hours. Trash and debris should be removed as necessary. Sediment should be cleaned out of the sedimentation chamber when it accumulates to a depth of more than 25 cm.
  - When the filtering capacity of the filter diminishes substantially (e.g., when water ponds on the surface of the filter bed for more than 24 hours), the topsoil and underlying 8 cm of filter material should be removed and replaced with fresh material. The removed sediments should be tested and disposed of in an acceptable manner (e.g., landfill). Silt/sediment should be removed from the filter bed when the accumulation exceeds 3 cm.
  - Vegetation within the sedimentation chamber of sand filters should be mowed

Application

- Sand filters have been successfully used in diverse applications for small (less than 40,680 m^2) tributary areas (Debo et al., 1994).
- Recommended for "ultra-urban" areas where area is limited and runoff is poor quality; not recommended for new construction sites.
- Most sand filters are limited to an impervious tributary area of 20,235 to 40,680 m^2. Follow-up sand filters, placed at the outlet of detention basins, may treat tributary areas in excess of 406,800 m^2 (Urbonas and Ruzzo, 1986).

Removal efficiencies

As seen in Table 2.10, the performance of extended detention ponds increases significantly for TSS with time. According to most sources, there is little significant change in the removal of other pollutants after a 24-hour period.

Table 2.10: Pollutant Removal Performance (%): Surface sand Filter (Harlow et al., 2000).

Pollutant	FHWA Evaluation and Management of Highway Runoff Quality	National Pollutant Removal Performance Database	Controlling Urban Runoff (Schueler ,1987)
TSS	70-86	87	99
TP	50-65	59	65-75
TN	31-47	32	60-70

FHWA: USA Federal Highway Administration

Advantages and disadvantages (NCDENR, 2005)

Advantages:

- Highly effective at filtering TSS.
- Larger units can attenuate runoff peaks, particularly if the design storm is not large (e.g., less than 10-year return period).
- Underground closed filters are useful where space is limited.

Disadvantages:

- Need to integrate trash screens or grated inlets in all designs so materials that can cause premature failures are kept out of filter chambers. Frequent cleaning of these screens may be required. Clogging of the sand filter materials can limit BMP's life-span.
- If anoxic conditions develop in the sand filter, phosphorus levels can increase as water passes through the sand filter. (Avoid this by the maintenance described below when the filter takes longer than 24 hours to drain.)
- May not be effective in controlling peak discharges.

Application in urban environment

Sand filters, in general, are good options for these areas because they consume little space. Underground and perimeter sand filters are particularly are well suited for ultra urban watersheds as they consume no surface space (CWP, 2006).

Cost

Average typical base capital construction cost for a sand filter is US\$159/m^3 (Brown and Schueler, 1997b). Average annual maintenance cost (% of construction cost) is 12% (Livingston et al, 1997; Brown and Schueler, 1997b).

2.3.10 Vegetated Swale

General description

Vegetated swales are broad, shallow, channels designed to slow runoff, promote infiltration, and filter pollutants and sediments in the process of conveying runoff. Vegetated Swales (Figure 2.8) are often heavily vegetated with a dense and diverse selection of native, close-growing, water-resistant plants with high pollutant removal potential. The various pollutant removal mechanisms of a swale include: sedimentary filtering by the swale vegetation (both on side slopes and on bottom), filtering through a subsoil matrix, and/or infiltration into the underlying soils with the full array of infiltration-oriented pollutant removal mechanisms (PCSL, 2006).

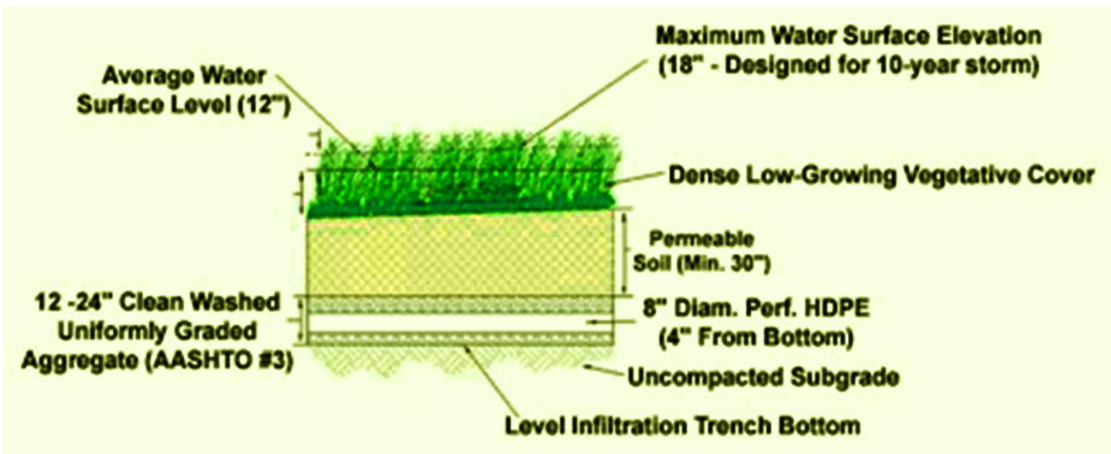

Figure 2.8: Vegetated Swale (PCSL, 2006).

Design features

- *Storage*: Vegetated Swales are sized to temporarily store and infiltrate the 25 mm storm event, while providing conveyance for up to the 10-year storm with freeboard; flows for up to the 10-year storm are to be accommodated without causing erosion. Swales should maintain a maximum ponding depth of 0.46 meter at the end point of the channel, with a 0.31-meter average maintained throughout. 0.15 meter of freeboard is recommended for the 10-year storm (PCSL, 2006).
- *Pre-application treatment*: A small forebay should be used upstream of the channel to trap incoming sediments. A pea gravel diaphragm (a small trench filled with river run gravel) can also be used to pretreat runoff that enters the sides of the channel (CWP, 2006).
- *Site-specific characteristics:* The average slope of the watershed should be 5 percent or less. Maximum use should be made of natural topographic features such as natural swales, draws, and depressions. Soils should have infiltration rates of 4.5 mm/hr. Heavy clays are generally not acceptable. The seasonal high groundwater table should be at least 3 m below the surface of the channel (Harlow et al., 2000).

- *Loading and operating cycle*: Residence times between 5 and 9 minutes are acceptable for swales without check-dams. The maximum ponding time is 48 hours, though 24 hours is more desirable. Studies have shown that the maximum amount of swale filtering occurs for water depths below 15 cm (PCSL, 2006).

- *Vegetation type*: A dense grass cover is the best vegetation to maximize the performance of a grass swale. However, native plants and wetland vegetation, where they already exist or where they can be established before substantial discharge begins, can be effective, provided that erosion will not result from the increased runoff (NCDENR, 2005).

- *Maintenance*: Maintenance requirements are summarized as follows (CWP, 2006):

  - Inspect pea gravel diaphragm for clogging and correct the problem.

  - Inspect grass along side slopes for erosion and formation of rills or gullies and correct.

  - Remove trash and debris accumulated in the inflow forebay.

  - Inspect and correct erosion problems in the sand/soil bed of dry swales.

  - Based on inspection, plant an alternative grass species if the original grass cover has not been successfully established. Mow grass to maintain a height of 7 to 10 cm

Application (PCSL, 2006)

- Parking, Commercial and light industrial facilities.

- Roads and highways, Residential developments ,and

- Pretreatment for volume-based BMPs.

Removal efficiencies

The pollutant removal performance (Table 2.11) depends on whether or not a grass swale or channel has been designed to specifically provide water quality functions.

Table 2.11: Pollutant Removal Performance (%): Vegetated Swale (Harlow et al., 2000).

Pollutant	FHWA Evaluation and Management of Highway Runoff Quality 61 m length	National Pollutant Removal Performance Database	Controlling Urban Runoff (Claytor and Schueler , 1996)
TSS	93	81	88
TP	29	34	49
TN	25	84	74

FHWA: USA Federal Highway Administration

The key to pollutant removal may be soils with high infiltration rates and flow velocities of less than 0.15 m/sec (Urbonas and Stahre, 1993).

Advantages and disadvantages (NCDENR, 2005)

Advantages:

- Can reduce runoff peak rates and increase opportunities for filtration, partially infiltrating runoff from small storm events if the underlying soil is not compacted or saturated.
- Can reduce the use of costly development infrastructure, e.g., curb and gutter.
- Can be aesthetically pleasing.
- Low-slope swales can create wetland areas.

Disadvantages:

- Could be subjected to standing water and mosquito infestations.
- May be subject to channelization due to concentrated flows
- Roadside swales may pose traffic hazards in residential subdivisions.

Application in urban environment

Grassed swales are generally not well suited to ultra urban areas because most runoff is conveyed in underground storm drain pipes rather than open channels on the surface (CWP, 2006).

Cost

Average typical base capital construction cost for vegetated swales is US$18/m^3 (SWRPC, 1991). Average annual maintenance cost (% of construction cost) is 6% (SWRPC, 1991).

2.3.11 Vegetated Filter Strip

General description

The EPA defines a Vegetated Filter Strip (figure 2.9) as a “permanent, maintained strip of planted or indigenous vegetation located between non-point sources of pollution and receiving water bodies for the purpose of removing or mitigating the effects of non-point source pollutants such as nutrients, pesticides, sediments, and suspended solids.” Pollutant removal mechanisms include sedimentation, filtration, absorption, infiltration, biological uptake, and microbial activity (PCSL, 2006).

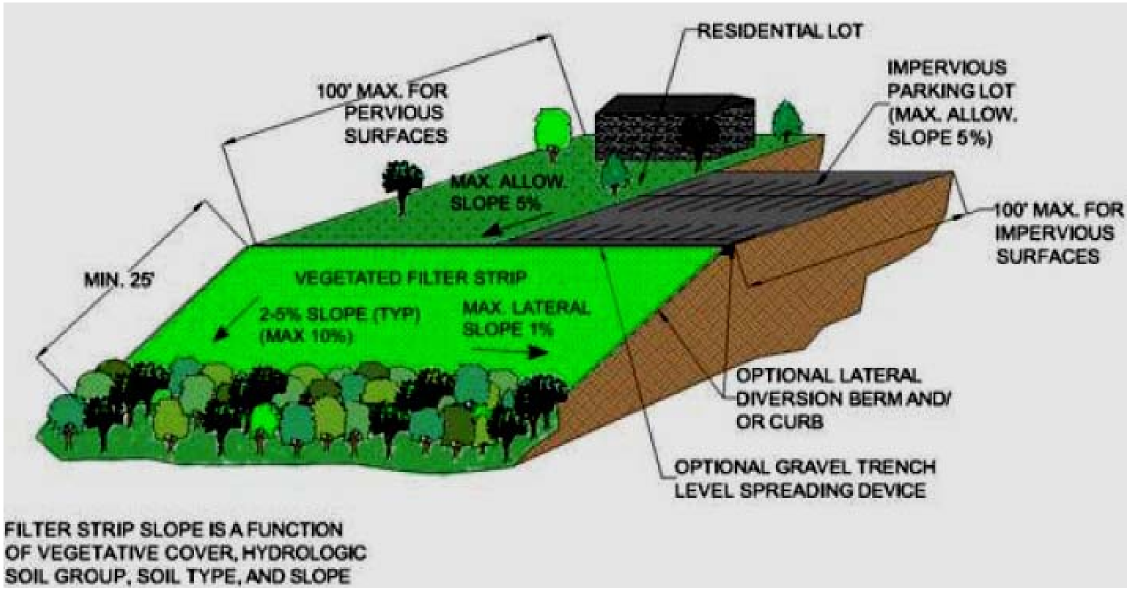

Figure 2.9: Vegetated Filter Strip (PCSL, 2006).

Design features

- *Storage*: The filter strip should be designed with a pervious berm of sand and gravel at the toe of the slope. This feature provides an area for shallow ponding at the bottom of the filter strip. Runoff ponds behind the berm and gradually flows through outlet pipes in the berm. The volume ponded behind the berm should be equal to the water quality volume. The water quality volume is the amount of runoff that will be treated for pollutant removal in the practice. Typical water quality volumes are the runoff 12 mm of runoff over the entire drainage area to the practice (CWP, 2006).

- *Pre-application treatment*: To enhance the effectiveness of the filter strip, a pervious berm of sand and gravel should be installed at the toe of the slope (AES, 2006). Use a pea gravel diaphragm at the top of the slope. The pea gravel diaphragm (a small trench running along the top of the filter strip) serves two purposes. First, it acts as a *pretreatment* device, settling out sediment particles before they reach the practice. Second it acts as a level spreader, maintaining sheet flow as runoff flows over the filter strip (CWP, 2006).

- *Site-specific characteristics:* Filter strips are most applicable for small watershed areas, typically less than 20,234 m^2 (EAS, 2006) that is, rather than moving uniformly over the surface, it forms rivulets which are slightly deeper and cover less area than the sheet flow. When flow concentrates, it moves too rapidly to be effectively treated by a grassed filter strip. As a rule, flow concentrates within a maximum of 22 m for impervious surfaces, and 46 for pervious surfaces (CWP, 1996). Filter strips should be designed on slopes between 2% and 6%. Filter strips should not be used on soils with high clay content because they require some infiltration for proper treatment. Another

possible limiting factor would be very poor soils that cannot sustain a grass cover crop. The seasonal high water table should be at least 0.6 to 1.2 m lower than any point along the filter strip (PCSL, 2006).

- *Loading and operating cycle*: Width of filter strip perpendicular to flow (should be at least 30 m for every acre of drainage area).Vegetated filter strip flowpath should be at least 15 m long. A good rule-of-thumb is to increase the flow length of the strip by 1.2 m for each 1 percent of slope, particularly if the strip is forested (NCDENR, 2005). The ratio of contributing drainage area to filter strip area should not exceed 6:1(PCSL, 2006).
- *Vegetation type*: The following are suggested criteria for selecting vegetation for filter strips (NCDENR, 2005): Deep-rooted; Dense, well-branched top growth; and Resistant to damage from either saturation or drought. The vegetation for Filter Strips may be comprised of (PCSL, 2006): Turf Grasses; Meadow grasses, shrubs, and native vegetation, including trees; Indigenous areas of woods and vegetation.
- *Maintenance*: Maintenance requirements are summarized as follows (NCDENR, 2005):
  - At least annually, remove deposited sediment, especially from the upstream edge, to maintain original contours and grading.
  - Repair gullies and rills that form and regrade the filter strip to ensure that the runoff flows evenly in a thin sheet over the filter strip.
  - Repair the level spreader, if necessary, to prevent the formation of channels in the filter strip.
  - Reseed and regrade the filter strip to maintain a dense growth of vegetation, especially if the strip has been used for sediment control (Schueler et al., 1992).
  - Mow filter strips vegetated with grasses and harvest the clippings two to three times a year to promote the growth of thick vegetation with optimum pollutant removal efficiency. Turf grass should not be cut shorter than 8 to 13 cm and may be allowed to grow as tall as 30 cm depending on aesthetic requirements (NIPC, 1993).

Application (PCSL, 2006**)**

- Residential development, Commercial and industrial facilities
- Roads and highways, Parking
- Pretreatment for other structural BMPs (Infiltration Trench, Bioretention, etc.)

Removal efficiencies

Studies from agricultural settings suggest that a fifteen foot wide grass buffer can achieve a 50% removal rate of nitrogen, phosphorous and sediment, and that a 30m buffer can

reach closer to 70% removal of these constituents (Desbonette et al., 1994). It is unclear how these results can be translated to the urban environment, however. The characteristics of the incoming flows are radically different both in terms of pollutant concentration and the peak flows associated with similar storm events.

Yu et al. (1992) have investigated the effectiveness of a vegetated filter strip to treat runoff from a large parking lot. The study found that the pollutant removal varied depending on the length of flow in the filter strip. The narrower (23 m) filter strip had moderate removal for some pollutants and actually appeared to export lead and nutrients (Table 2.12).

Sheet flow over the grass surface was achieved using a level spreader and the associated average percentage removal efficiencies were 71%, 38%, and 10%, for TSS, TP, and TN respectively. USA EPA (1993) has stated that the average TP removal efficiency is 65%. The achievement of maximum contact and residence times through the use of level spreader devices to obtain sheet flow over the grassed surface is a desirable factor in the use of these systems (Schueler, 1987; Livingston et al, 1984).

Table 2.12: Pollutant Removal Performance (%): Urban Vegetated Filter Strip (CWP, 2006).

Pollutant	**Pollutant Removal (%)**	
	23 m Filter Strip	**50 m Filter Strip**
TSS	54	84
TN	-27	20
TP	-25	40

Advantages and disadvantages (AES, 2006)

Advantages:

- Provides effective stormwater flood control by slowing down runoff and storing water, including water infiltration into the soil.
- Improves water quality by filtering pollutants from stormwater (oils, greases, metals, and sediments that can be picked up from paved surfaces).
- Can be used as a system by itself, or in conjunction with other BMPs.
- May help maintain temperature of receiving waters.
- Flexible to incorporate existing natural features and a variety of vegetation types.
- Preserves natural/native vegetation and provides habitat for wildlife.

Disadvantages:

- Need to maintain vegetative cover for controlling erosion and reducing particulates in the runoff.
- Not appropriate for hilly or highly impervious terrain.

- Requires maintenance to remove trash.

Application in urban environment

Filter strips are often impractical in Ultra Urban areas since they require more space than is often available (CWP, 2006).

Cost

Average typical base capital construction cost for filter strips is US$23/m^3 (SWRPC, 1991). Average annual maintenance cost (% of construction cost) is 0.5% (SWRPC, 1991).

2.3.12 Overview of the Best Management Practices

The average expected phosphorus removal efficiency, costs, and land consumption for the aforementioned Best management Practices are presented in Table 2.13.

Table 2.13 BMPs average phosphorus removal efficiency, costs, and land consumption.

BMPs	Removal Efficiency (%)	Costs		Land consumption (%)
		Capital Construction Cost (US$/m^3)	Maintenance Cost (% of Construction Cost)	
Infiltration Basins	65	46	6	2.5
Infiltration Trenches	58	141	12	2.5
Bioretention	55	187	6	5
Sand Filters	63	159	12	1.5
Vegetated Swale	37	18	6	10
Vegetated Filter Strips	45	23	0.5	100
Stormwater Wetland	46	33	2	4
Wet Pond	48	27	4.5	2.5
Dry Extended Detention Ponds	39	27	0.7	2.5

* Base year for all cost data: 1997

*Land consumption represents the amount of land needed as a percent of the impervious area that drains to the practice to achieve effective treatment (Claytor and Schueler, 1996).

2.4 Diversion or Treatment of inflow streams

The goal in diversion/treatment is to reduce lake P concentration and case histories show that mean concentration can be expected to decrease nearly in proportion to the reduction in inflow concentration on an annual basis. However, lake quality (algal biomass and transparency) is a function of the absolute equilibrium lake P concentration, not the proportional change in concentration. If P does not reach a limiting level, and there is some suggestion that SRP must reach 10μg/L or less to limit algal biomass, then one cannot expect quality improvements following diversion or treatment. Cooke et al., (2005) stated that the response of lakes and reservoirs to stream diversion or treatment has been varied, from lake P concentration being easily predictable from models with adequate change in trophic state to those showing no change in trophic state, as indicated by algal biomass and transparency.

Ferric iron has been successfully used to remove P, metals, and organics from inflows to drinking water supplies in the U.S., U.K., and The Netherlands (Cooke et al., 2005). An iron system was established to improve raw water quality of the Amsterdam Rhine Canal and Bethune Polder before their discharge into Lake Loenderveen, part of the water supply of Amsterdam, The Netherlands. The system has been in operation since 1984. Water is treated with $FeCl_3$ (7 mg Fe/L) and detained in a settling basin (mean residence time of 4 h) before it enters the lake. When P content of the raw water is very high, two in-line coagulation and settling systems are used. The basins store floc, which is routinely removed with a hydraulic dredge to drying fields. The Loosdrecht Lakes receives a similar treatment. The process is highly effective with 90% P reduction, and little final treatment in the potable water supply plant is needed (van der Veen et al., 1987). Effective chemical interception of P for water supply is therefore feasible. There is no technical reason why this procedure could not be applied to the inflows of recreational lakes and reservoirs.
Stream treatments are difficult and expensive because they must be continuous as long as the stream has high nutrient concentrations (Cooke et al., 2005).So upstream measures are needed to reduce the source of nutrients.

Cooke et al. (2005) stated that Harper et al. (1983) may have been the first to devise a system to treat stormwater inflows with alum. The lower volume and duration of storm flow (versus river flow) allowed treatment of the entire discharge. Harper's early system led to development of a more sophisticated system with sonic flow meters and variable speed pumps that automatically injected alum at a flow-proportioned rate, based on jar tests for dose determination. The floc was discharged to the lake, providing sediment P inactivation, apparently without a significant floc build-up after three years of operation.The system reduced P loading and lake TP fell from > 200 μg P/L to about 25 μg P/L. Algal biomass decreased, and transparency, macrophyte biomass, and dissolved oxygen increased. The USEPA 7 day Chronic Larval Survival Growth Test on fathead

minnows (Pimephales promelas) demonstrated no chronic toxicity of the alum-treated stormwater as long as pH remained at pH 6.0–6.5. High mortality was evident at pH 7.5 in this low alkalinity system. Floc disposal in the lake was a problem solved by collecting floc in a separate basin, and drying it. The floc is a Grade 1 wastewater sludge that can be disposed of via land application (Harper, 1990).

The use of diversion was limited to the diversion of treated sewage or industrial wastewater involves installing interceptor lines to convey the wastewaters away from the degraded water body to waters that have greater assimilative capacity (Cooke et al., 2005), but no case studies were found where the diversion was used to divert streams (canals or ditches) from discharging to a certain water body.

2.5 In-Lake Measures

To restore or improve the aquatic ecosystem itself, in-lake measures are necessary for the long-term success of restoration programs, but they are not restorative actions per se and moreover may not be sufficient. It is done to accelerate the return to earlier (more natural) conditions, to remove long-lived contaminants. The in-lake measures are divided into two categories as follows:

2.5.1 In-Lake Methods to Reduce Phosphorus Concentrations and Cycling

- *Dilution:* is a procedure that can lower water column phosphorus concentrations by adding water that is low in phosphorus (National Research Council 1992). It will also increase washout of algal cells from a lake. In principle, addition of dilution water to a lake will increase its total phosphorus loading rate but decrease the mean inflow phosphorus concentration. The lake's flushing rate is also increased, and this tends to decrease phosphorus sedimentation. A sufficient rate of flushing causes a great loss of algae, without reducing the limited nutrient concentration. Even a minimal amount of flushing water can be effective in diluting nutrients (Asaeda et al., 2004). The advantages of using dilution water include (1) relatively low cost if water is available, (2) an immediate and proven effectiveness if the limiting nutrient can be decreased, and (3) some success even if only moderate- to high-nutrient water is available, through physical limitations to large algal concentrations. However, the principal limitation for use of this technique is the availability of low-nutrient dilution water (Cooke et al., 2005).

- *Phosphorus Inactivation:* A significant reduction in nutrient loading to a eutrophic lake is a necessary but sometimes insufficient step in order to decrease water column phosphorus concentrations enough to reduce the amount of algae (National Research Council 1992). Phosphorus release from lake sediments at high pH, or when dissolved oxygen in overlying water is low or zero, can be a major source of phosphorus to the

water column. Under certain conditions, phosphorus released from lake sediments will be transported to the upper layers of a lake and stimulate an algal bloom. This process, in which sediments enriched in organic and inorganic matter from external loading and in-lake production cause dissolved oxygen consumption and phosphorus release, is known as internal loading. It can be great enough to delay or prevent a lake's recovery from nutrient diversion or interception. Phosphorus inactivation reduces the rate of phosphorus release from lake sediments by the addition of aluminum salts (sodium aluminate, aluminum sulfate) to them (Cooke et al., 1986). Aluminum hydroxide is formed and appears as a visible floc that settles on sediments and binds with phosphate ions to form a solid that is insoluble under low or zero dissolved oxygen. Phosphate ions diffusing from the sediment are trapped by the floc. The process has proved to be effective and long lasting (Asaeda et al., 2004).

- *Sediment Skimming:* Phosphorus release from lake sediments is greatest from the most recent phosphorus-rich surface layers. Sediment skimming involves the use of a hydraulic dredge to remove this layer. This procedure, although effective, is more costly than phosphorus inactivation. It does have a restorative effect without the addition of potentially toxic materials, especially when nutrient inflows have been reduced or eliminated. Once the equipment is set up for sediment skimming, it might be reasonable to proceed with a full-scale sediment removal to accomplish both lake deepening and control of internal loading.

- *Sediment Oxidation:* Phosphorus release from lake sediments can be controlled by accelerating the oxidation of sediment organic matter and providing a chemical environment that favors the binding of phosphorus by iron in the top 5 to 10 cm of lake sediment but, adding potentially toxic elements, such as aluminum, should be avoided. Instead, calcium nitrate is injected into the sediments. The nitrate serves as an electron acceptor in the absence of oxygen, and decomposition of organic matter proceeds via denitrification. At the same time, iron sulfide is oxidized and phosphate ions are bound to the resulting ferric oxide. In some lakes, calcium hydroxide is added to bring the pH to the optimum for denitrification, and ferric chloride may be added if the lake is iron deficient (Asaeda et al., 2004) .

- *Hypolimnetic withdrawal:* The impact of phosphorus release from lake sediments can be controlled by siphoning the nutrient-rich deep (hypolimnetic) water from a lake or discharging the hypolimnetic water of a reservoir through a deep gate in its dam. If release exceeds new external loading, the procedure should gradually deplete the sediments of phosphorus and could reduce the amount of nutrients entrained from deep to surface waters each summer. Summer and early autumn algal blooms should be reduced (Nurnberg, 1987). Continued high nutrient loading to a lake is likely to negate the effects of this technique. The advantages of hypolimnetic withdrawal are threefold: (1) relatively low capital and operational costs, (2) evidence of effectiveness in a large

fraction of cases, and (3) potentially long-term and even permanent effectiveness (Cooke et al., 2005).

2.5.2 Management of Symptoms

- *Biomanipulation:* Biomanipulation was broadly defined by Shapiro et al. (1975) to include a wide array of biological controls for water quality problems. They distinguished these from the many chemical and engineering approaches that exist for water quality improvement. More recently, a narrower definition, derived from the pioneering studies of Hrbacek et al. (1961), has been adopted by some limnologists: the manipulation of fish community structure to permit large herbivorous zooplankton grazers to flourish and to control nuisance algae (Shapiro, 1990b). This approach to biomanipulation is currently the object of substantial research programs in the United States, Canada, and several European nations (Gulati et al., 1990). Biomanipulation is not regarded as a substitute for reduction of nutrient loads. Important questions revolve around the capacity of biomanipulation to (1) reduce algal biomass where loads cannot be controlled and (2) augment or accelerate the effects of load reductions.

- *Artificial Circulation:* Artificial circulation is a management technique whose goal is to achieve and maintain an isothermal and isochemical water column in a lake or reservoir that otherwise would exhibit stratification during summer. This is accomplished by injecting compressed air into a pipeline tethered at the lake's bottom in the deep zone. The last several meters of the pipe are perforated so that a vigorous bubble curtain is created, with enough energy to mix the water column rapidly. Even on the warmest days, a properly sized system will have a temperature vertical difference of less than 3ºC (versus 20º C in typical stratified lakes). Pumps and whirling blades may also accomplish this goal (Cooke et al., 1986).

3 Materials and Methods

In this chapter the identification of the sources of nutrient loading in the polder, and assessment of the proposed Ecological Engineering measures to decrease the loading of nutrients, particularly phosphorus, from various sources and identification of the best measures will be presented.

3.1 Study area

3.1.1 Sloterbinnenpolder (SBP)

SBP is located in the west of Amsterdam (from 52.342 to 52.401º N, and from 4.793 to 4.862º E) and lies between the Haarlemmerweg and the Sloterweg and west of the ring road A10. In the middle of the polder the Sloterplas Lake (89 ha) is located (Map of the polder is shown in Fig.3.1). The total area of the polder is about 1830 ha, in which paved area, green area, and water courses area (including channels and Sloterplas Lake) account for 50%, 40%, and 10% of the total area respectively. The desired water level of the polder is - 2.10 m. NAP.

The polder consists of a number of areas with similar water level (sub polder systems). Since most water levels are below sea level, ground water has to be drained out off the polder with the use of a complex system of surface water bodies varying from small ditches to larger canals. The main water system of a polder is called the boezem. It has the highest water level in the polder. Water is pumped from the smaller and lower lying sub polder systems into the boezem system and from there eventually pumped into the main Dutch water system. In dry warm summer periods however the whole system is used the other way around. Water from the main Dutch water system is pumped into the polder to maintain constant ground water levels in order to prevent growth reduction of grass and crops, and also to prevent subsidence. The water movement in a polder is thus almost completely determined by water level management.

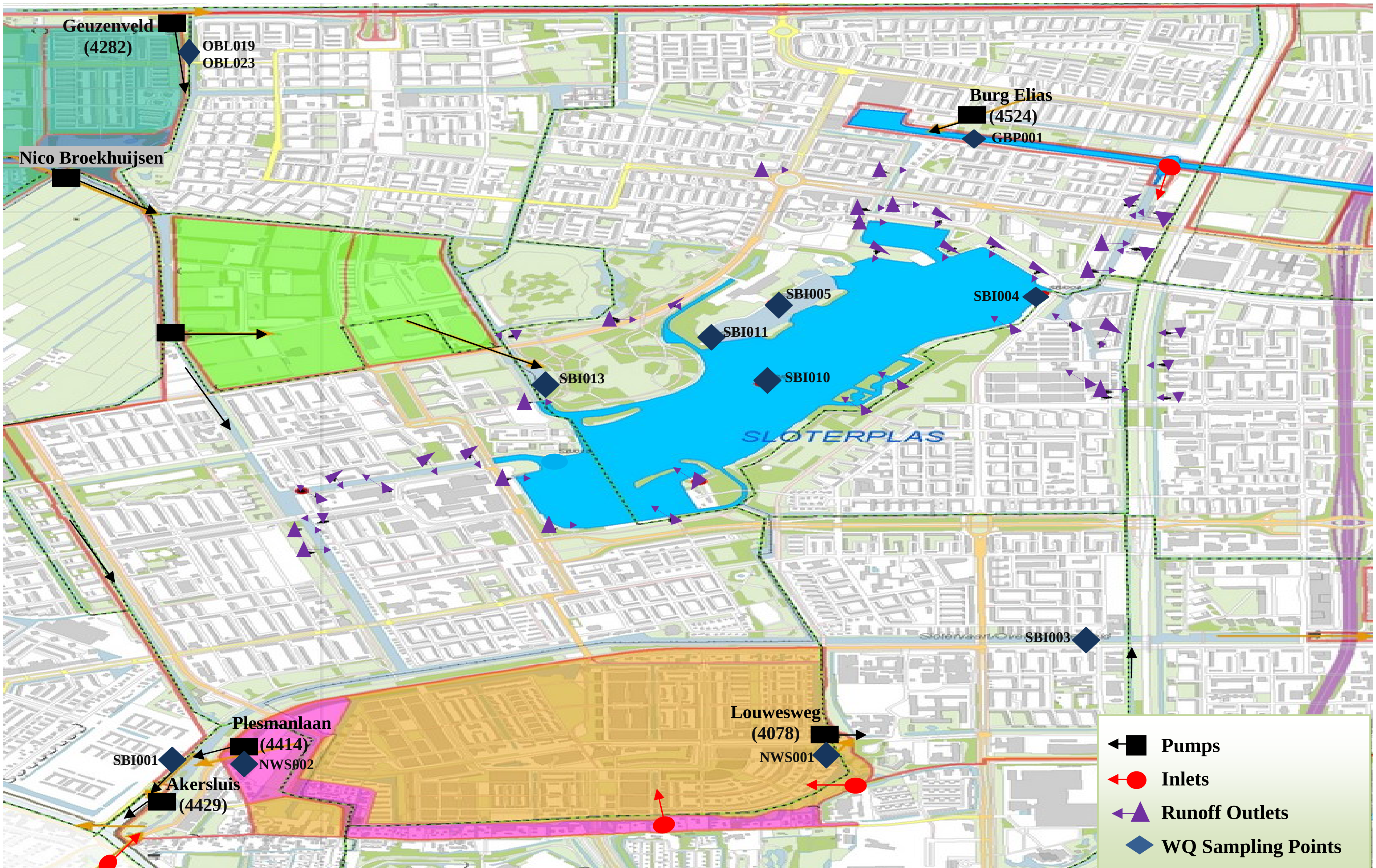

Figure 3.1 Sloterbinnenpolder Map

3.1.2 Sloterplas Lake

The Sloterplas is part of the combined urban water system Sloterbinnen and Middelveldsche polders. The Lake was dug between 1948 and 1956 for sand winning purpose. Sand winning wells are generally characterized by the high depth, the hard banks and the stratification which appears in the summer six-month period. The Sloterplas has a surface area of 89 ha and average depth of 30 meters. The maximum depth is 35 meters. Along the bank the average depth is one meter. The bank length of the Lake is 6 km. The desired water level of the Lake is - 2.10 m. NAP

3.2 Identification of the sources of nutrients load

In this section the available data, water balance, load calculation method, and sources of the uncertainty will be introduced.

3.2.1 Data Collection

Most of the data was collected from Waternet Company (www.waternet.nl) which includes: daily precipitation and evaporation, daily pumping flow rate (inflow and outflow), calculated daily runoff, calculated inflows from other polders through inlets, assumed ground water seepage 1 mm/day which is a reasonable value (Jacobs, 2006), and discrete water quality samples for the water courses. The schematic of all inflows and outflows of the Sloterbinnenpolder is shown in Figure 3.2

Daily precipitation data were taken from Schiphol station-Amsterdam. Daily evapo-transpiration data was calculated using Makkink (1957) equation, which is based on global radiation and temperature data.

$$ET = C * C_M \frac{s}{s+\gamma} \frac{R_s}{L} \tag{3.1}$$

Where:

ET	Evapotranspiration mm/day
C	constant to convert units (C=86400)
C_M	Makkink constant (0.8, for open water, 0.65 for crop evaporation)
L	latent heat of vaporization ($2.45*10^6$ J/Kg)
s	slope of temperature-saturation vapour pressure curve (kPa/K)
γ	psychometric constant (0.067 kPa/K at sea level)

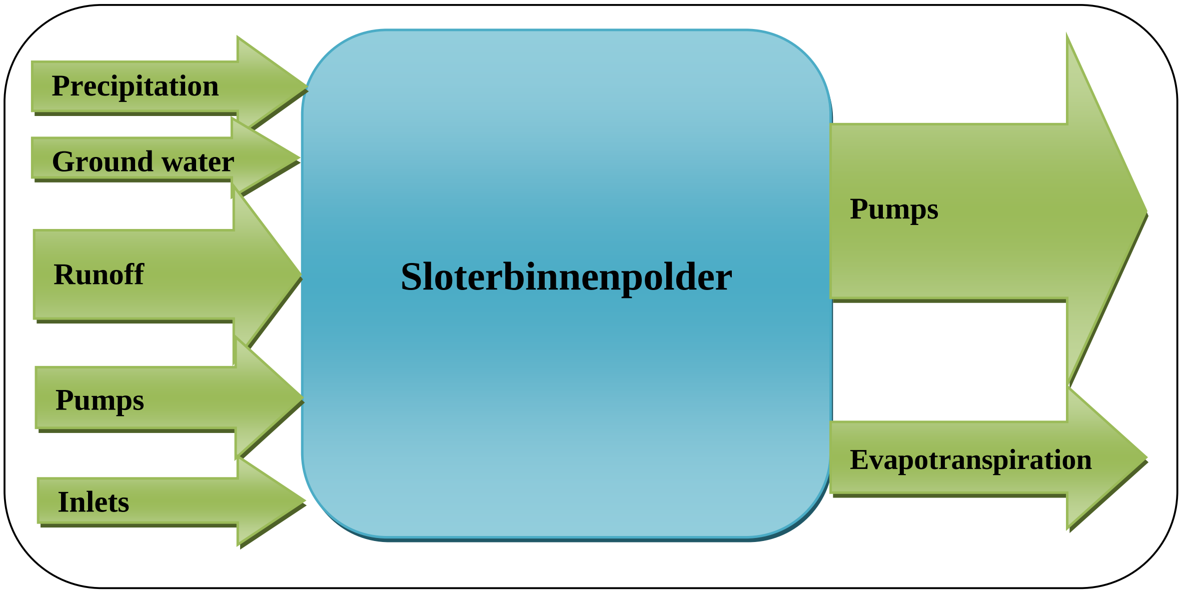

Figure 3.2 Schematic of inflow and outflows of Sloterbinnenpolder.

Daily pump data for the inflow pumps (Louwesweg, Plesmanlaan, Geuzenveld, and Nico Broekhuijsen) and outflow pumps (BurgElias, and Akersluis) as shown in Figure 3.1, was calculated as a product of the pumps capacity and the daily working hours.

Runoff data was calculated as the sum of the surface runoff, subsurface runoff (inter flow), and deep percolation (base flow).

Based on the measured levels, in the case of low water level (lower than -2.1 NAP) a certain amount of water is allowed to be discharged to the polder through sluice gates to maintain the minimum level. This amount of water (inlet) was calculated from the water balance equation (3.2). As well as the inlet; the outlet pump was calculated for maintaining the highest water level at – 2.05 NAP.

All the above data was for the period from 1998 to 2002, on the other hand the available water quality data (heavy metals, nutrients, and physical parameters) for the channels, ditches, and the lake was for the period from 2000 to 2004. The water quality data locations are shown in Figure 3.1. Water quality data for the runoff, ground water, and precipitation were collected from literature.

3.2.2 Water Balance

The water balance of Sloterbinnenpolder was done by Waternet, while the water balance of Sloterplas Lake was carried out according to an approach proposed by this research. The two water balances are presented here after.

3.2.2.1 Sloterbinnenpolder (SBP)

The existing water balances in the polder are based on the principle of continuity. This can be expressed with the equation 3.2:

$$I(t) - O(t) = \frac{\Delta S}{\Delta t} \tag{3.2}$$

Where I is the inflows in [L3/T], O is the outflows in [L3/T], and ΔS/Δt is the rate of change in storage over a finite time step in [L3/T] of the considered control volume in the system. The inflows comprise the inlet from the boezem systems, water pumped from the adjacent polders, surface runoff, seepage of groundwater into the polder, and precipitation, while the outflow comprises water pumped out of the polder, and evapotranspiration.

3.2.2.2 The Sloterplas Lake

It was noticed that the water balance done by Waternet was for the whole polder and there was nothing specific for the Sloterplas Lake. So there is a need to make a water balance for the Sloterplas Lake, but there was a difficulty to do this water balance because the flow rates to and from the Lake are unknown. So the following approach was used to estimate these unknown flow rates.

Lake assessment for management usually requires a model that adequately predicts TP in the lake/reservoir in question. Mass balance models for P are based on the kinetics of continuously stirred tank reactors (CSTR), which are commonly used in chemical engineering (Reckhow and Chapra, 1983). By continuously mixing the volume in such a reactor, holding that volume constant and maintaining the water inflow rate equal to the water outflow rate, Vollenweider (1969b) proposed the TP mass balance for lakes based on the abovementioned model:

$$\frac{d\,TP}{d\,t} = \frac{L}{\bar{Z}} - \rho\,TP - \sigma\,TP \tag{3.3}$$

The steady state equation is:

$$TP = \frac{L}{\bar{Z}(\rho + \sigma)} \tag{3.4}$$

Where:

TP= Predicted total phosphorus concentration of the lake (mg/m^3)

L= Total phosphorus loading (mg/m^2 /yr), summation of the internal and external loading.

z: the mean depth (m)

ρ: The flushing rate (1/year), it is calculated from dividing the inflow by the lake volume

σ: The sedimentation rate coefficient (1/year)

The principal limitation of this model is determining the sedimentation rate coefficient. All other variables can be determined directly. However, to develop a model to describe a large number of lakes, it is useful to have some general way to estimate sedimentation. One approach is to use a unitless retention coefficient RTP (Vollenweider and Dillon, 1974; Dillon and Rigler, 1974a).

$$\sigma = (R_{TP} * \rho) / (1 - R_{TP}) \quad (3.5)$$

RTP is the retention coefficient, and it is equal to (15/ (18+ ρ z)) (Nürnberg, 1984).

So by using the equation 3.4 and 3.5, the Lake TP concentration could be predicted, and then it can be compared with the observed concentration. Scenarios (from 0% to 100% of the total flow rate) for the flow rates of runoff, inlet and inflow pumps to the lake were proposed. The predicted concentration calculated from the scenarios analysis was then compared with the observed concentration which led to the identification of the better scenario (the closest to the reality) for the inflow flow rates.

3.2.3 Trophic State of the Sloterplas Lake

Trophic state is defined as the total weight of living biological material (biomass) in a water body at a specific location and time. The Trophic State Index (TSI) of Carlson (1977) uses algal biomass as the basis for trophic state classification. Three variables, chlorophyll pigments, Secchi depth, and total phosphorus, independently estimate algal biomass.

The index is relatively simple to calculate and to use. Three equations are used: Secchi disk, TSI (SD); chlorophyll pigments, TSI (CHL); and total phosphorus, TSI (TP).

$$\mathbf{TSI = 10\ (6 - \log_2 SD)} \quad (3.6)$$

$$\mathbf{TSI = 10\ (6 - \log_2 7.7/CHL\ \mathit{a}^{\ 0.68})} \quad (3.7)$$

$$\mathbf{TSI = 10\ (6 - \log_2 96/TP)} \quad (3.8)$$

Where SD in meter, CHL a in μg/L, and TP in μg/L

A major strength of TSI is that the interrelationships between variables can be used to identify certain conditions in the lake or reservoir that are related to the factors that limit algal biomass or affect the measured variables. When more than one of the three variables is measured, it is possible that different index values will be obtained. Because the

relationships between the variables were originally derived from regression relationships and the correlations were not perfect, some variability between the index values is to be expected. However, in some situations the variation is not random and factors interfering with the empirical relationship can be identified. These deviations of the total phosphorus or the Secchi depth index from the chlorophyll index can be used to identify errors in collection or analysis or real deviations from the "standard" expected values (Carlson 1981). Some possible interpretations of deviations of the index values are given in the Table 3.1 (Carlson 1983).

The effect of Total Nitrogen (TN) limitation can be estimated by having a companion index to the Total Phosphorus TSI. Such an index was constructed by Kratzer and Brezonik (1981) using data from the National Eutrophication Survey on Florida lakes. This index is calculated using the formula:

$$\mathbf{TSI = 54.45 + 14.43 \ln (TN)} \qquad \mathbf{(3.9)}$$

Where TN in μg/L

Table 3.1: Interpretation of Deviations of Trophic State Index

Relationship Between TSI Variables	**Conditions**
TSI(Chl) = TSI(TP) = TSI(SD)	Algae dominate light attenuation; TN/TP ~ 33:1
TSI(Chl) > TSI(SD)	Large particulates, such as *Aphanizomenon* , dominate
TSI(TP) = TSI(SD) > TSI(CHL)	Non-algal particulates or color dominate light attenuation
TSI(SD) = TSI(CHL) > TSI(TP)	Phosphorus limits algal biomass (TN/TP >33:1)
TSI(TP) >TSI(CHL) = TSI(SD)	Algae dominate light attenuation but some factor such as nitrogen limitation; zooplankton grazing or toxics limit algal biomass.

3.2.4 Phosphorus Mass Balance

Nutrient loads from the various sources were identified as a result of multiplying the flow rates obtained from the water balance by the concentration. The nutrient concentrations that were used in this research are shown in Table 3.2. The research focused only on the TP concentration, the reason for that will be explained in the next chapter. The data in the following table were based on water quality measurements by Waternet company (pumped in and inlet concentrations), and literature data (runoff, precipitation, and ground water concentrations).

Table 3.2 Total Phosphorus Concentrations used in calculations.

Total Phosphorus(TP) mg/l			
	Minimum	**Maximum**	**Mean**
Precipitation (EEA, 2005)	0.004	0.04	0.022
Ground water (Witteveen Bos, 1996)	0.24	0.43	0.335
Runoff (CPW, 2006)	0.4	1.0	0.5
Pump (measured data)	0.913	2.16	1.3
Inlet (measured data)	0.913	2.16	1.3

3.3 Assessment of the Proposed Ecological Engineering Measures

The mass balance models were then used to provide the data to select restoration alternatives. Two main concepts were taken into consideration in proposing the possible alternatives: (1) The possibility of accomplishing external phosphorus load diversions or treatment (sufficient to protect the lake from further deterioration or sufficient to accomplish a significant change in trophic state or lake quality), (2) the in-lake procedures that can be used to accelerate recovery following external load controls or to accomplish further improvement.

The assessment of the proposed alternatives in addition to the alternative of "no action" was based on the following criteria:

- The projected effectiveness, reliability and longevity,
- The amount of restoration that can be accomplished, and
- The cost.

The appropriate technique, or techniques, to apply to the lake requires a decision based largely on judgment. Although cost may be the principal criterion, reliability and longevity of the technique(s) will also be important.

4 Results and Discussions

In this chapter the identification of the sources of nutrients load in the polder, and assessment of the proposed Ecological Engineering measures to decrease the loading of nutrients, particularly phosphorus, from various sources and identification of the best measures will be presented.

4.1 Present Status

The Trophic State Index for Sloterplas Lake was calculated with average data of year 2000 according to Equations 3.6, 3.7, and 3.8. The TSI was 60, 60, 80 for the Secchi disk, TSI (SD); chlorophyll pigments, TSI (CHL); and total phosphorus, TSI (TP) respectively, and they are shown in Figure 4.1.

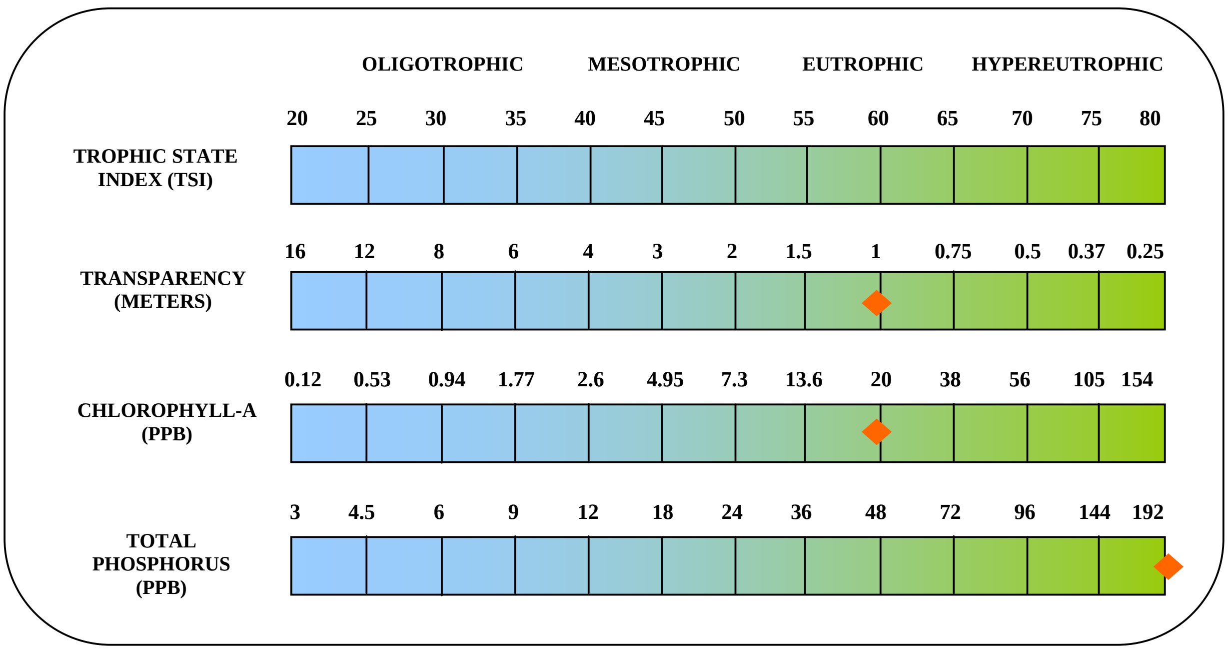

Figure 4.1: The Sloterplas Lake Trophic State Index (TSI).

The deviation of TSI (TP) is indicating that Algae dominate, but some factor such as nitrogen limitation; zooplankton grazing or toxics limit algal biomass (Carlson 1983).

The TSI (TN) was calculated using Equation 3.9, and it was 60 in which it is equal with TSI (CHL) and TSI (SD). This together with the deviation of TSI (TP) indicates nitrogen limitation. Although nitrogen limited growth rate during summer, fixation of atmospheric N by blue-green algae should have supplied enough N to complement the available P supply. N fixation is not a rapid process; maximum rates of cell N replacement and growth

of about 5% and 10%/d have been reported (Horne and Goldman, 1972; Horne and Viner, 1971).

However, either phosphorus is already the most limiting nutrient, or in the case of highly eutrophic or hypertrophic lakes where nitrogen is often limiting, phosphorus can be made to limit if its concentration is sufficiently reduced (Cooke et al., 2005). So this research will focus on the phosphorus sources reduction in the urban water system restoration, specifically the Lake restoration.

4.2 Water Balance

4.2.1 SBP Water Balance

The SBP yearly water balance was conducted using the Equation 3.2. The calculations of the water balance are shown in Table 4.1. Clearly the inflows flow rates are equal to the outflows flow rates for all years, except in year 1998 the outflow was slightly smaller than inflow, which imply that the system in balance. That is due to the fact that the system is managed in a way where the water levels are maintained in the range from -2.1 to -2.05 NAP, these water levels are controlled by the amount of water that is pumped out from the system and the amount of inlet water from the boezem systems.

Table 4.1 Yearly Water Balance (all units are in millions cubic meters per year).

	Inflows Flow rate					**Outflows Flow rate**		**ΔS**	**Total In**	**Total Out**
	P	**GW**	**In**	**R**	**Pumps**	**E**	**Pumps**			
1998	1.90	0.58	0.34	12.20	2.28	1.00	16.26	0.04	17.31	17.27
1999	1.59	0.58	0.74	9.00	1.74	1.15	12.51	0.00	13.66	13.66
2000	1.67	0.58	0.32	9.84	1.83	1.09	13.17	0.00	14.25	14.25
2001	1.62	0.58	0.30	10.43	2.13	1.12	13.94	0.00	15.06	15.06
2002	1.48	0.58	0.00	8.80	2.55	1.10	12.31	0.00	13.41	13.41

E: Evaporation P: Precipitation In: Inlet GW: Ground Water R: Runoff S: Storage

4.2.2 The Lake Water Balance

The Lake water balance was conducted by adopting the approach explained in section (3.2.2.2). Following are the procedures of the calculations.

A. Scenarios (from 0% to 100%) for the flow rates of runoff, inlet and inflow pumps to the lake are presented in Table 4.2. The year 2000 was chosen, where the lake concentration data are available. The flow rates of ground water and precipitation

directly to Lake are 581216, and 1674570 cubic meters per year respectively. Ground water seepage is assumed to be 1 mm/day.

Table 4.2 Runoff, Inlets, and Pumps flow rates Scenarios, showing total volume of water passing through the lake (m^3/year).

		Runoff through lake				
		65%	70%	75%	80%	85%
Inlet & Pumps through lake	65%	7796673	8288652	8780631	9272610	9764589
	70%	7904438	8396417	8888396	9380375	9872354
	75%	8012203	8504182	8996161	9488140	9980119
	80%	8119969	8611948	9103927	9595906	10087885
	85%	8227734	8719713	9211692	9703671	10195650

B. Calculation of the Predicted TP of the lake: The predicted TP was calculated using equation 3.8 which calculates the TP as a function of the internal load, the external load, the flushing rate, the main depth, and sedimentation rate. The aforementioned parameters were calculated as follows:

i. External Loads were calculated as a product of multiplying the flow rates of year 2000 (Table 4.2) by the average concentrations (Table 3.1). The external loads are shown in Table 4.3.

ii. The mean internal load was calculated by Waternet using the iron-bound P (BD-P) in sediment (2500 mg/m^2/year) (Witteveen Bos, 1996) due to the fact that the sediment release rate (internal load) in anoxic cores is directly related to iron-bound P (BD-P) in sediment, where the iron cycle usually controls sediment P release in stratified anoxic lakes (Nürnberg, 1988).

iii. The flushing rate was calculated by dividing the inflows by the Lake volume. It has been noticed that all the values of the flushing rate was in the range from 0.34 to 0.43 which is considered as a low flushing rate which could increase the nutrients in the sediment

Flushing rate has an indirect effect on Lake P concentration as shown with a Vollenweider steady state, mass balance P model (Equation 3.4). Adding more water with lower nutrient content also increases nutrient loading, while the resulting increased flushing rate decreases nutrient loss through sedimentation (Uttormark and Hutchins, 1980). These processes could be counteracting the

dilution effect in some instances, because, as the authors stated, “a reduction in the influent concentration tends to reduce in-lake concentration, but a reduction in phosphorus retention tends to increase in-lake concentration.” They showed that a large increase in the combined flushing rate obtained by adding low-nutrient water (40% of the normal inflow nutrient content) could theoretically increase the lake nutrient concentration if the original flushing rate ρ is low enough, e.g., 0.1/yr. If the flushing rate is relatively large (≥ 1.0/yr) initially, the effect of reduced sedimentation rate is minimized and a reduction in lake concentration will result, but large quantities of water are necessary. Of course, the amount of water needed to achieve a given reduction in inflow concentration is a function of the concentration difference between the normal inflow and dilution water source.

Table 4.3 External Loads (mg/m^2/year) Calculations with different scenarios.

			Runoff Scenario				
			65%	70%	75%	80%	85%
		Precipitation	23.20	23.20	23.20	23.20	23.20
		Ground Water	122.61	122.61	122.61	122.61	122.61
Inlet & Pumps Scenario	65%	Runoff , Inlet & Pumps	5068.016	5298.804	5529.592	5760.38	5991.168
	70%		5227.075	5457.863	5688.651	5919.439	6150.227
	75%		5386.135	5616.923	5847.711	6078.499	6309.287
	80%		5545.194	5775.982	6006.77	6237.558	6468.346
	85%		5704.254	5935.042	6165.83	6396.618	6627.406
	65%	Total Load	5213.83	5444.62	5675.40	5906.19	6136.98
	70%		5372.89	5603.67	5834.46	6065.25	6296.04
	75%		5531.95	5762.73	5993.52	6224.31	6455.10
	80%		5691.01	5921.79	6152.58	6383.37	6614.16
	85%		5850.07	6080.85	6311.64	6542.43	6773.22

iv. The sedimentation rate was calculated based on Equation 3.5 which relates the sedimentation rate with the flushing rate (calculated above) and the retention coefficient for TP (R_{TP}). The R_{TP} depends mainly on the mean depth and the sedimentation rate (R_{TP} = 15 / (18+ ρ*z)). The R_{TP} decreases as flushing rate increases. The negative relation between flushing rate and R_{TP} is logical. That is, as flushing rate increases there is less time for TP to settle, so R_{TP} decreases

accordingly. That is analogous with using other relationships between the R_{TP} and the flushing rate (Cooke et al., 2005). A R_{TP}–flushing rate relation may be relatively constant with loading change (Edmondson and Lehman, 1981), or vary with loading (Kennedy, 1999).

C. Comparing the predicted TP with the observed TP in the lake (average P concentrations in the lake in year 2000): The direct comparison between the predicted TP and the observed TP showed that the two values could be approximately the same at 65% runoff flow rate (6.4*106 m^3/year of the total runoff 9.8*106 m^3/year) and 85% inlet and pumps flow rates (1.8*106 m^3/year of the total inlet and pumps 2.2*106 m^3/year) (Table 4.4). Accordingly, it can be concluded that 65% of the runoff and 85% of inlet and pumps flow rates are entering the Lake.

Table 4.4 Prediction of Lake Vollenweider parameters and TP concentration with different scenarios.

			Runoff Scenario				
			65%	70%	75%	80%	85%
Inlet &Pumps Scenario	65%	External Load (mg/m^2/year) (from Table 4.3)	5213.83	5444.62	5675.40	5906.19	6136.98
	70%		5372.89	5603.67	5834.46	6065.25	6296.04
	75%		5531.95	5762.73	5993.52	6224.31	6455.10
	80%		5691.01	5921.79	6152.58	6383.37	6614.16
	85%		5850.07	6080.85	6311.64	6542.43	6773.22
		Internal Load (mg/m^2/year)	2500	2500	2500	2500	2500
	65%	Flushing rate, ρ (1/year)	0.339	0.358	0.376	0.395	0.413
	70%		0.343	0.362	0.380	0.399	0.417
	75%		0.347	0.366	0.384	0.403	0.421
	80%		0.351	0.370	0.388	0.407	0.425
	85%		0.356	0.374	0.392	0.411	0.429
		Mean Depth, z (m)	30	30	30	30	30
	65%	Sedimentation rate, σ (1/year)	0.386	0.391	0.395	0.399	0.403
	70%		0.387	0.392	0.396	0.400	0.403
	75%		0.388	0.393	0.397	0.401	0.404
	80%		0.389	0.394	0.398	0.401	0.405
	85%		0.390	0.395	0.398	0.402	0.406
	65%	Predicted Conc., TP (mg/m^3)	354.38	353.77	353.35	353.10	352.98
	70%		359.18	358.46	357.94	357.59	357.38
	75%		363.92	363.10	362.49	362.04	361.74
	80%		368.60	367.69	366.98	366.44	366.04
	85%		**373.23**	372.22	371.42	370.79	370.31
		Observed Conc.(mg/m^3)	**373.23**	373.23	373.23	373.23	373.23

By assuming that the percentages of the runoff and the inlet and pumps flow rates that entering the lake is constant for all the years, the Lake inflows for all the years from 1998 to 2002 was calculated. Inflow rates to lakes are presented in Figure 4.2.

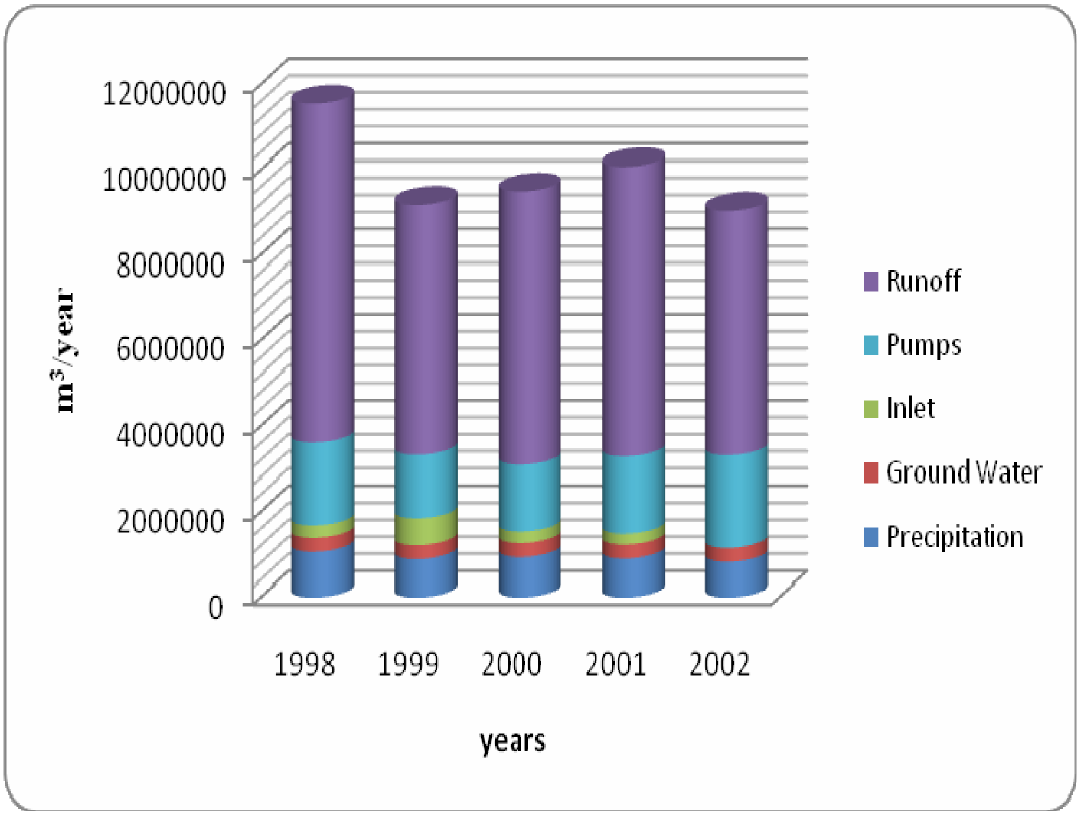

Figure 4.2 The Sloterplas Lake's Inflow rates.

As mentioned before, the principal limitation of the steady state mass balance model is determining the sedimentation rate coefficient. The steady state mass balance model illustrated by Equation 3.6 has been verified for a large population of lakes (Cooke et al., 2005). This suggests that the general form (Eq. 3.7) of the sedimentation term is reasonable, although the error for predicting the TP content in any given lake may be quite large (about ± 50 μg/L).

Taking into account the uncertainty of the used approach (± 50 μg/L) led to the result that the predicted TP concentration is equal to 373.23± 50 μg/L. The first probability is that the predicted TP concentration is equal to 423.23 μg/L, the corresponding scenario is 65% of pumps and inlets and 5% of runoff flow rates. The second probability is that the predicted TP concentration is equal to 323.23 μg/L, the corresponding scenario is 30% of pumps and inlets and 100% of runoff flow rates.

Obviously, the two aforementioned probabilities gave unreasonable results, where the first one showed that almost no runoff is entering the lake and the second one showed that 100% of the runoff is entering the Lake, these percentages could not be correct according to the system pattern shown in Figure 3.1.

According to Cooke et al., (2005), there still may be a problem with using the mass balance model for stratified lakes even if it can be verified for whole-lake TP. From predicted TP, Chl *a* and transparency are usually predicted as biological and physical factors defining trophic state and lake quality, and are a function of TP in the productive zone (i.e., epilimnion) and not of whole-lake TP. Usually, epilimnetic TP declines during the stratified period while hypolimnetic TP increases. Thus, either the epilimnion and hypolimnion must be modeled separately with diffusion between the two strata included to account for exchange of TP, or mean epilimnetic TP must be estimated from a relationship between that and whole lake TP. The latter may be satisfactory, because relationships among Chl *a*, TP, and transparency are usually based on summer means, which are in turn most often used for management purposes (Shuster et al., 1986). So it is stressed again that the results from the lake water balance have a high degree of uncertainty.

4.3 Phosphorus Sources Assessment

The sources of the phosphorus loads in the SBP polder and in addition to the sources of phosphorus loads in Sloterplas Lake will be presented in this section.

4.3.1 SBP phosphorus loads Sources

Areal Loads from the different phosphorus sources were calculated according the following equation:

$$\textbf{TP Areal Load(mg/m}^2\textbf{/day)} = \frac{\textbf{Flow rate(m}^3\textbf{/day) * Concentration(mg/l) * 1000}}{\textbf{Area(m}^2\textbf{)}} \qquad \textbf{(4.1)}$$

Due to the variation of TP concentration during the year, both the minimum and the maximum concentration (Table 3.2) were used in the calculations in order to calculate the minimum and maximum Loads.

The contribution of the runoff, the precipitation, and the groundwater seepage to the total phosphorus load (Fig.4.3) showed almost the same trend in the period from 1999 to 2002. It should be is noticed that the pumps loads increased in the same period due to the increase of the pumps flow rates, and as a direct consequence of the increase of pumps

flow rates the inlets flow rates decreased to maintain the water levels in the polder, hence the inlet loads decreased during the same period.

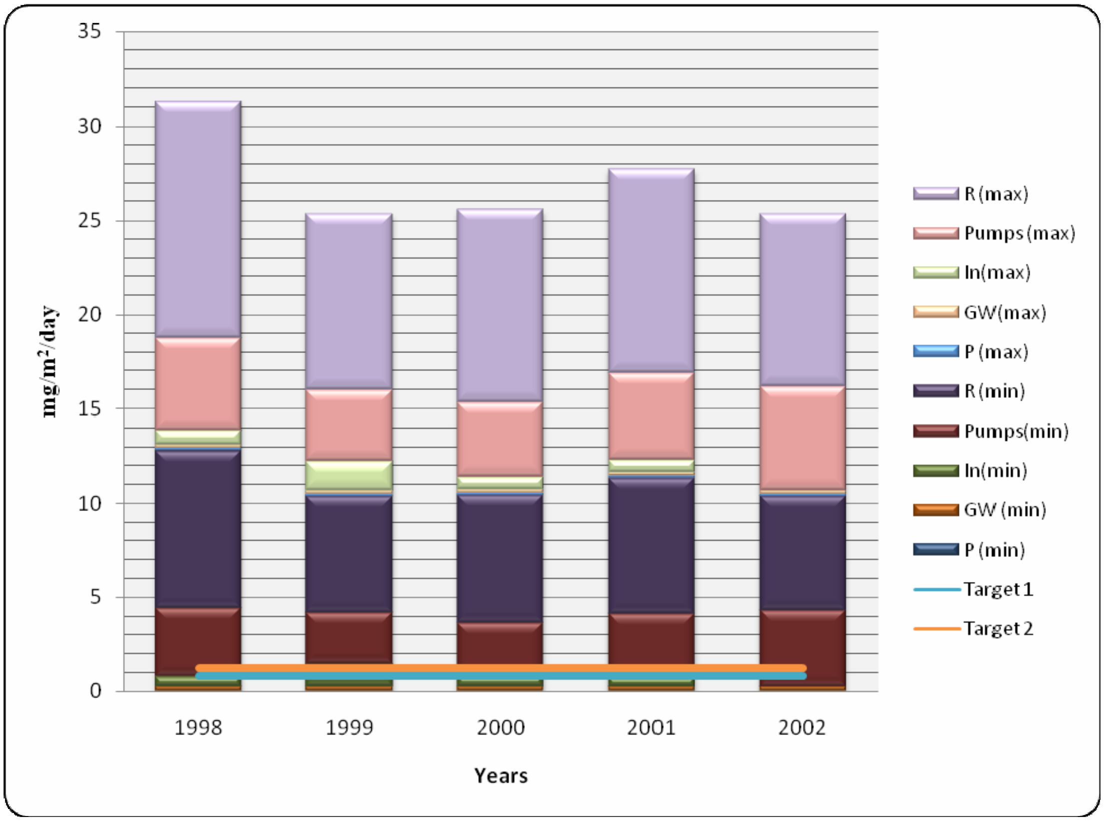

Figure 4.3 SBP's External Areal TP Loads

The target conditions (Target 1 and Target 2) were set by Waternet to 0.8 mg/m^2/day (30 μg/L), and 1.25 mg/m^2/day (50 μg/L); these values were assumed to achieve good ecological status for the lake. The selection of the two target values depended mainly on the ecological status of a lake near to Sloterplas Lake with similar conditions, it was found that the phosphorus concentration in this lake is about 50 μg/L and this concentration resulted in a clear lake and no eutrophication problem.

As the two lakes have similar conditions the company assumed that the same phosphorus concentration could achieve good ecological status for the Sloterplas Lake. In addition, it was reported by some researchers (Scheffer, 1998) that the recovery from the turbid (eutrophic) state to clear (oligotrophic) state doesn't follow the same path from clear to turbid state in terms of response to eutrophication (which is known as alternative stable states), so the value of 30 μg/L was assumed to be the alternative stable state for Sloterplas Lake.

The target conditions are consistent with the impact and pressure criteria used in EU Water Framework Directive for the risk assessment of eutrophication in lakes which indicated that the actual cut-off for total phosphorus between at risk and not at risk varies from < 10 μg/L to > 100 μg/L (EUWFD, 2005). Whilst, Vollenweider (1976) used the $TP_{e/m}$ 20 and $TP_{m/o}$ 10 μg/L for identifying the eutrophic–mesotrophic or mesotrophic–oligotrophic threshold, respectively.

The input percentages of the different TP loads to the SBP are shown in Figure 4.4. Clearly the contribution of the internal load has a high input (34%), while the contribution of all external loads represent 66%, so attention has to be paid for both internal and external TP loads in the management of the TP loads to the polder. Due to the low contributions of precipitation and ground water to the total load, in addition to the complexity of controlling their loads, they will not be considered in the Lake restoration.

The average current condition for TP loads to the polder is 20.4 mg/m^2/day (13.6 external, and 6.8 internal), while the target TP load is 0.8 mg/m^2/day. So for achieving the target, an overall load reduction of about 96% must be accomplished.

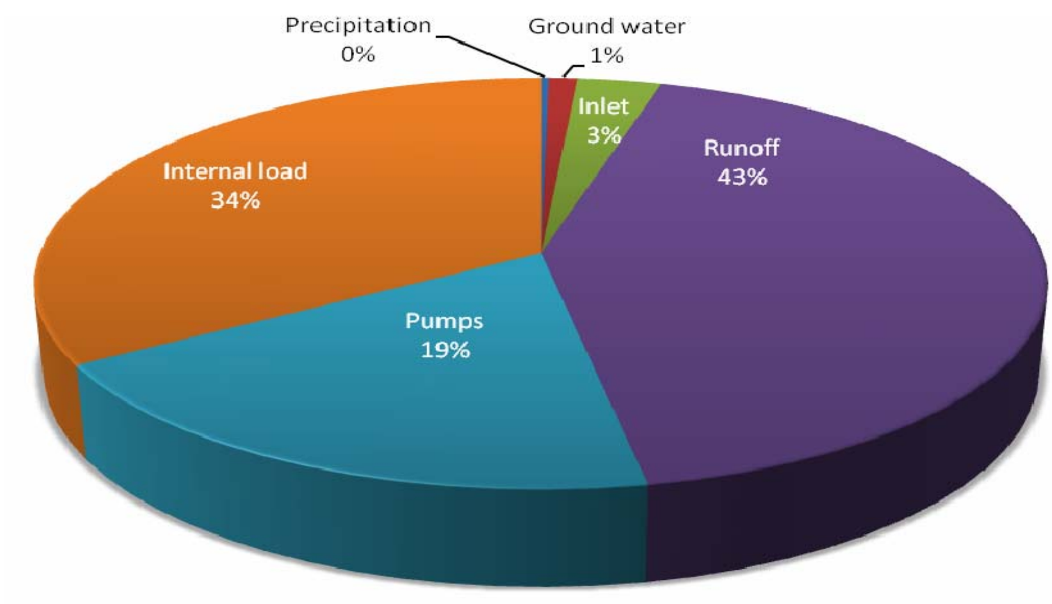

Figure 4.4 Average Contribution of External and Internal Phosphorus loading to SBP

4.3.2 The Lake Phosphorus loads Sources

The TP external loads (Fig 4.5) to the Lake were calculated as a result of multiplying each flow rate (obtained from the calculation in section (4.2.1)) by the corresponding concentration (Table 3.2).

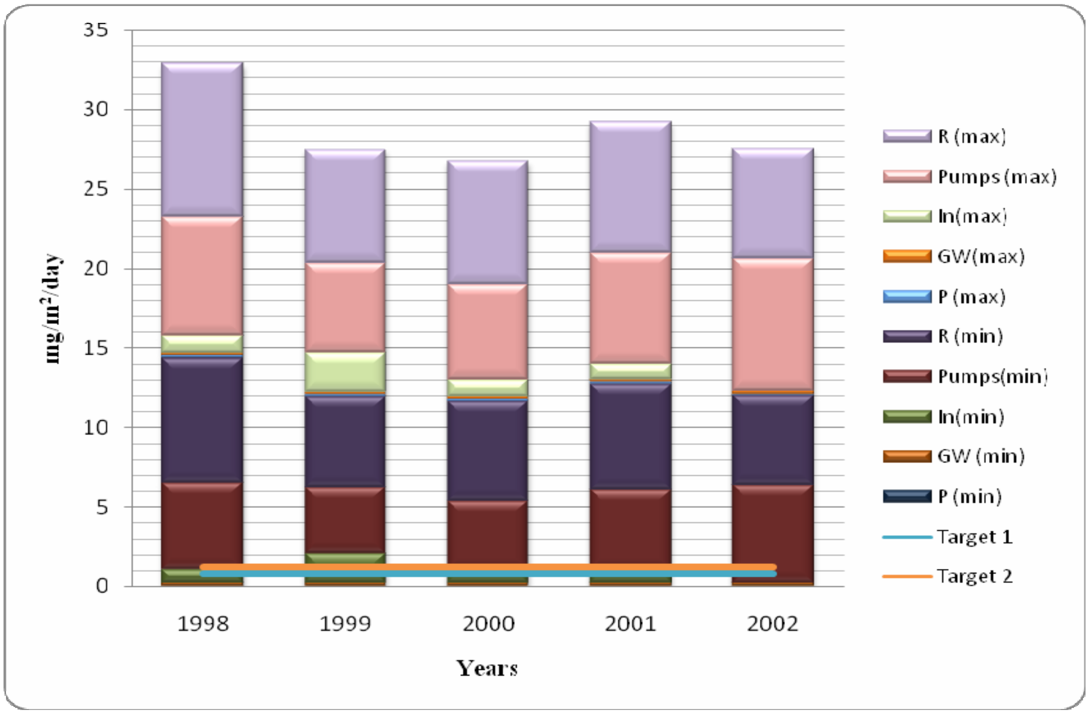

Figure 4.5 The Sloterplas Lake's External Areal TP Loads

The input percentages of the different TP loads to the Lake are shown in Figure 4.6. As the case of SBP, the contribution of the internal load in the Lake has a high input (32% of the total load), while the contribution of all external loads to the lake represent 68% of the total load, so attention has to be paid for both internal and external TP to restore the Lake.

The average current condition for TP loads to the Lake is 21.2 mg/m^2/day (14.4 external, and 6.8 internal), while the target TP load is 0.8 mg/m^2/day. So for achieving the target, an overall load reduction of about 96% must be accomplished.

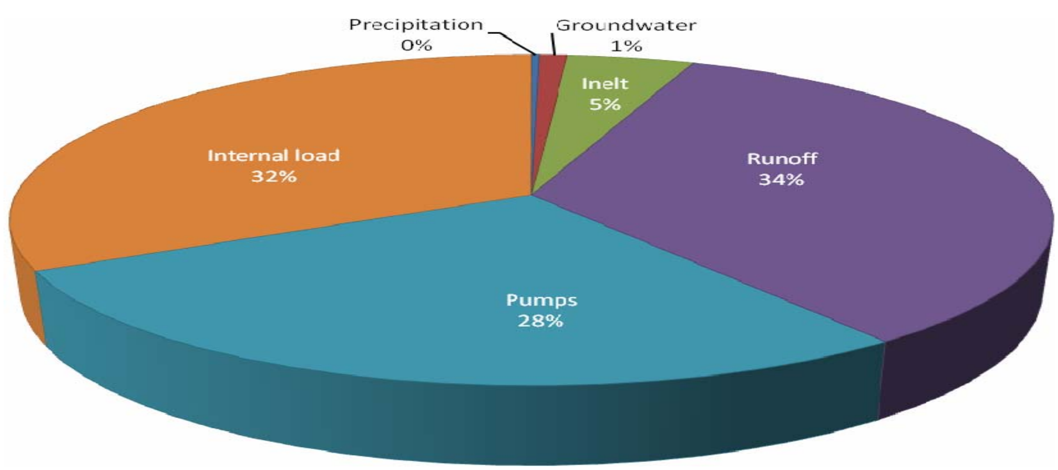

Figure 4.6 Average Contribution of External and Internal Phosphorus loading to Sloterplas

4.4 Assessment of the Proposed Measures

As mentioned in Chapter 2, the Lake restoration could be accomplished by reduction of external and internal loads. So in this section the assessment of urban water management measures to eliminate stress external loadings, and in-lake measures to accelerate the Lake return to earlier (more natural) conditions, will be discussed .

4.4.1 Diversion or treatment of pumps and inlet flow rates to the Lake

The average external load reduction to the lake that could be accomplished by diversion of pumps flow rate from the Lake is about 41% (from 14.4 mg/m^2/day to 8.4 mg/m^2/day) because in this case the phosphorus load of the pumps will not enter the lake. Figure 4.7 shows the total reduction in the lake external phosphorus loads when using the diversion.

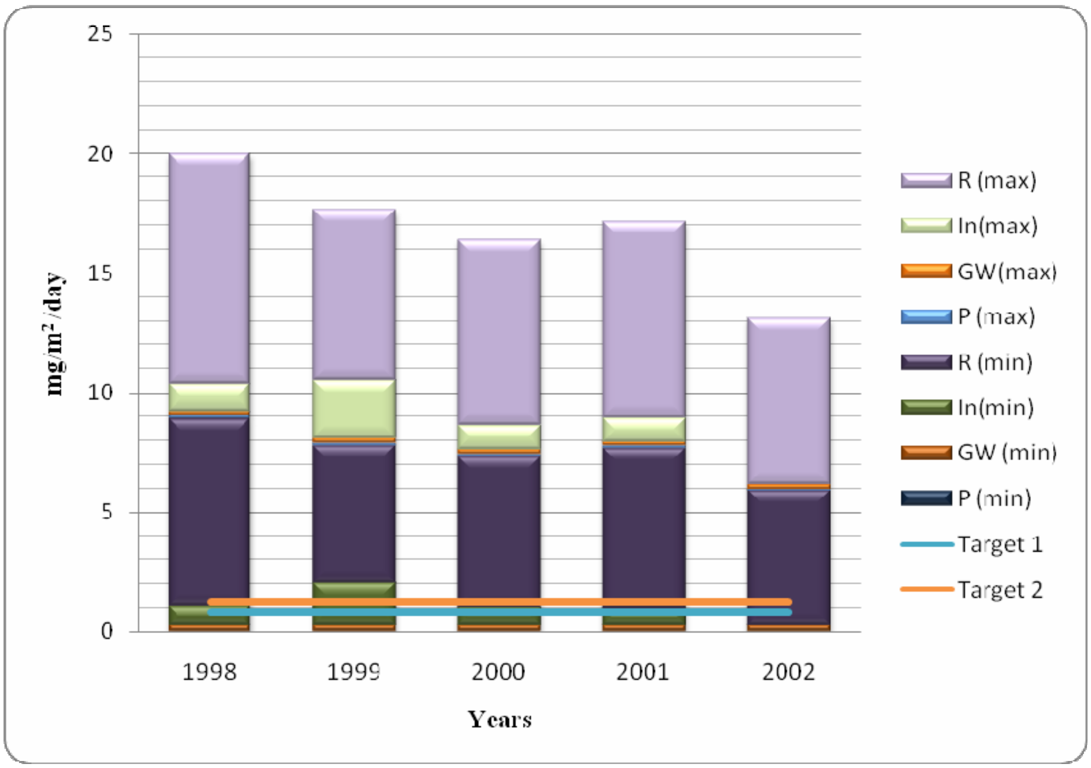

Figure 4.7 Lake External Phosphorus loads after pumps diversion

As mentioned in Chapter 2, the removal efficiency of the treatment of the pumps and inlets flow rates before it enters the lake is 90% removal of TP, so the treatment of the pumps and inlets flow rates to the Lake (Figure 4.8) could achieve a reduction of about 43% (from 14.4 mg/m^2/day to 8.2 mg/m^2/day).

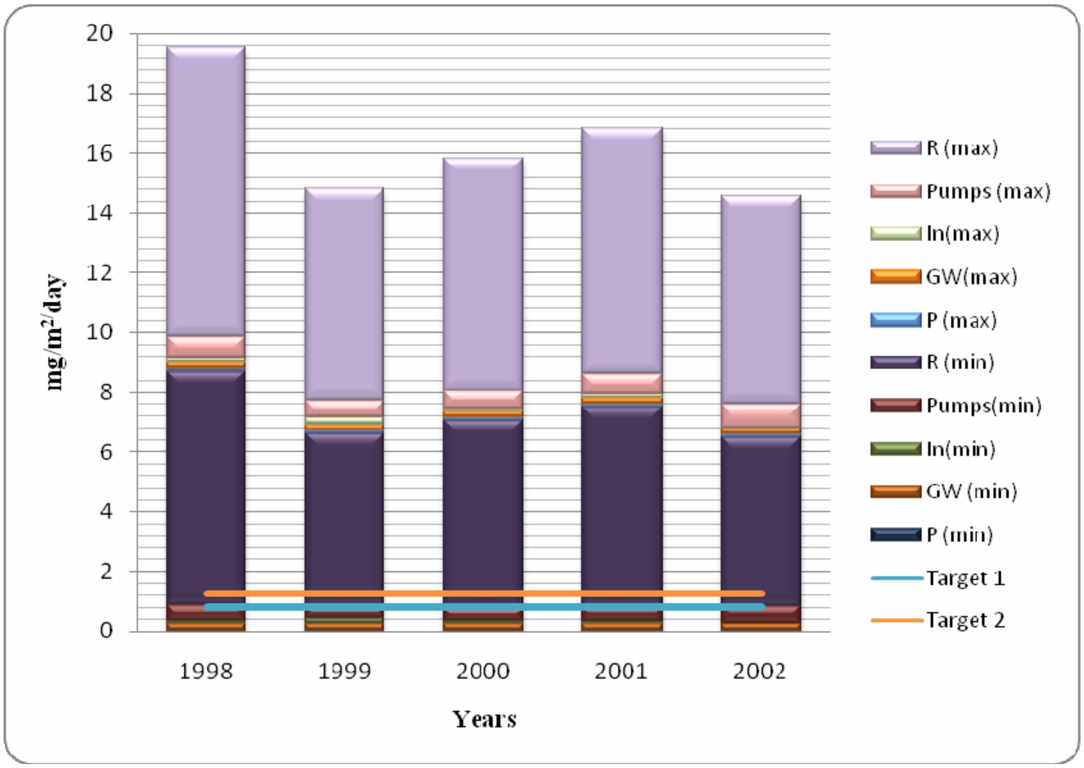

Figure 4.8 Lake External Phosphorus loads with pumps and inlet treatment

Although the load reduction achieved by diversion of pumps flow rate from the Lake is almost the same as the treatment of the pumps and inlet flow rates, the problem will remain for the quality of the receiving water after diversion, it is just problem shifting from one water course to another one without solving the problem of high TP load.

So the treatment will be better option to reduce the external load of pumps and inlet flow rates, even though the treatment will be difficult and expensive because it must be continuous as long as the stream has high nutrient concentrations.

4.4.2 Treatment of surface runoff

The treatment of surface runoff could be achieved either by ecological engineering (BMPs), or by conventional engineering (alum treatment). The results of using the two methods are presented the in following section.

4.4.2.1 Treatment of surface runoff by BMPs

From Table 2.13 the average phosphorus removal efficiency varied between different BMPs from 37% to 65%. Moreover, there’s no conflict between the uses of most of the above measures, so there’s a possibility for using two or more of the above measures in order to achieve the highest removal efficiency. In addition, in order to implement the above measures they have to be preceded by a pretreatment step, so it is recommended to use a combination of these measures. This suggests using treatment train sequential

components that will contribute to the treatment of stormwater before it leaves the site (AES, 2006).

The suggested treatment train will consist of vegetated swale followed by stormwater wet ponds. The treatment train represents an ecological approach to stormwater management and has proven effective and versatile in its various applications. Based on hydrologic modeling and published information on BMP effectiveness, the treatment train approach can be expected to reduce solids, nutrients and heavy metals loads by 85 percent to nearly 100 percent (AES, 2006).

The average removal efficiency of the treatment of 100% of the surface runoff flow rates before it enters the lake is about 90% removal of TP, the capital construction cost (without land cost) is about $22/m^3, (about $87 million for the total amount of the surface runoff) and the land needed is about 12.5 % of the impervious area that drains to the practice (99,000 m^2) to achieve effective treatment.

The average external load reduction to the lake (Figure 4.9) that could be accomplished by this treatment train by treating 100% of the surface runoff was about 32% (from 14.4 mg/m^2/day to 9.8 mg/m^2/day).

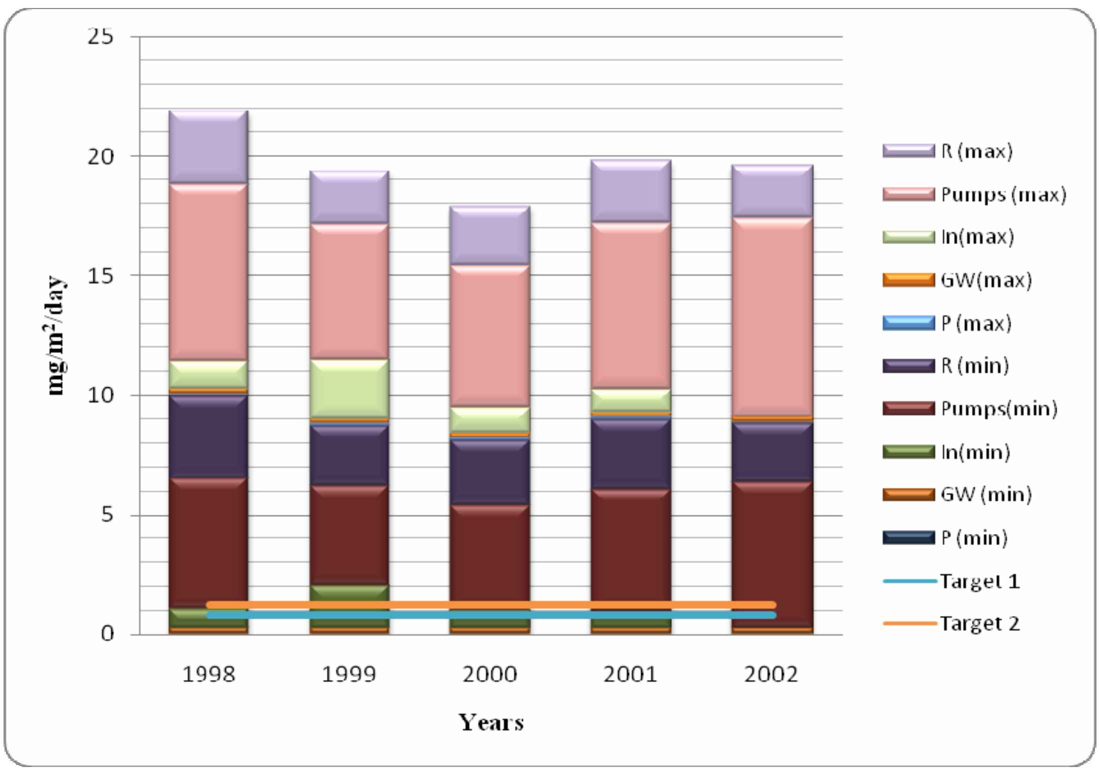

Figure 4.9 Lake External Phosphorus loads when 100% of surface runoff is treated using BMPs and assuming 90% removal efficiency

Although the load reduction accomplished by the suggested aforementioned measure is quite high, the assumption that all the surface runoff to the lake will be treated might not be possible due to the land constraints and the associated treatment cost.

As the land use is beyond the scope of this research, so there's no available data about the land use and the availability of land in the SBP that could be used in the treatment which is very essential in order to know the percentage of the surface runoff that could be treated. So, it is recommended that a detailed study about the land use has to be carried out in order to take the right decision about the percentage of the runoff that could be treated.

4.4.2.2 Treatment of surface runoff by Alum

The treatment of the surface runoff can also be done by alum treatment (As stated in section 2.5). The TP removal efficiency of the alum treatment is about 87%, so results quite similar (31% external load reduction) as treatment by BMPs

4.4.2.3 The best treatment method for surface runoff

Reviewing the above two methods, the following remarks can be drawn:

- The two methods can achieve almost the same removal efficiency,

- The construction cost of the ecological engineering method is quite less than the treatment by alum which is very complicated method that requires a lot of devices and equipments, but the ecological engineering method requires quite large land area; however it can be used for recreation when dry, or serves as green space, supporting wet prairie functions and wildlife habitat.

- The ecological engineering is better from the environmental point of view because it is natural system, while the use of alum will introduce strange substances to the system (chemicals) and also the sludge resulted from the use of alum method is difficult to be handled and it could it be environmentally harmful.

So in order to choose the best method for the treatment of the surface runoff, the trade-offs between the environmental aspects and the land requirements have to be taken into consideration.

4.4.3 Combined Scenarios

Obviously neither the treatment of pumps and inlet flow rates, nor the treatment of surface runoff will be enough to reduce the external loads to the Lake to reach the target condition.

So it is better to use both measures in order to achieve higher TP reduction of the external loads. The average external load reduction (Figure 4.10) achieved by the combined measures was 75% (from 14.4 mg/m^2/day to 3.6 mg/m^2/day).

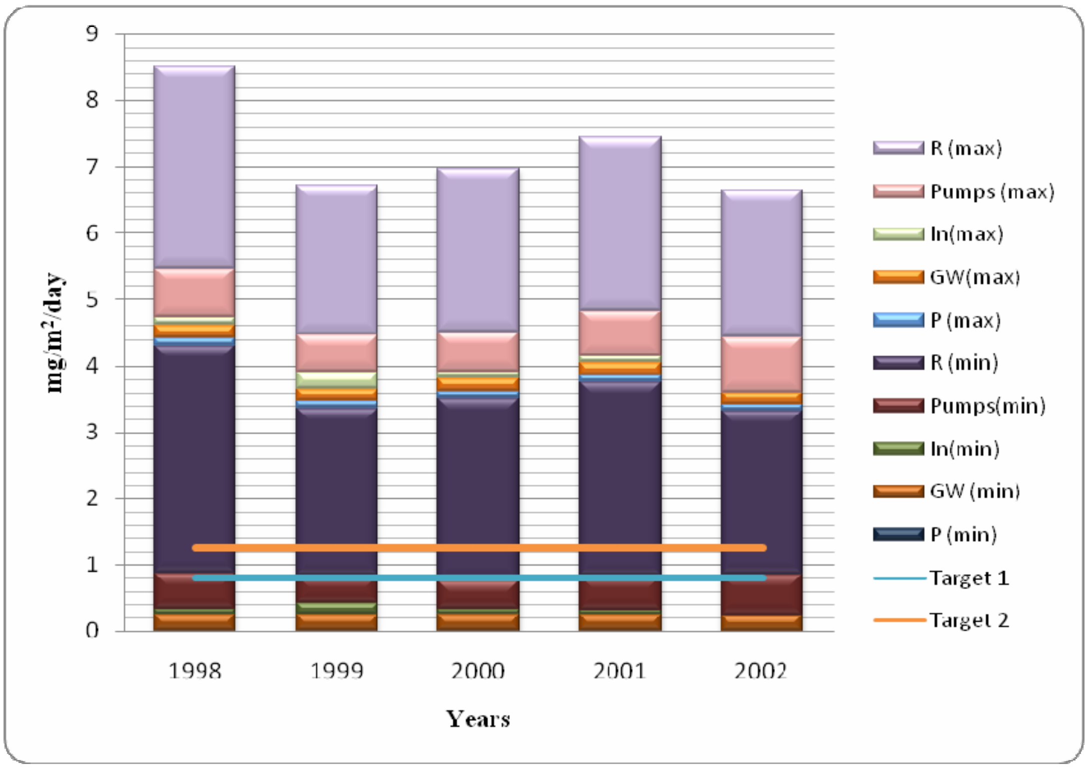

Figure 4.10 Lake External Phosphorus loads with combined scenario.

However the achieved TP reduction of the external loads to the Lake by the combined scenario is still far away from the target condition, due to the following reasons:

- Although the removal efficiency of the treatment of the surface runoff flow rate is 90%, the remaining 10% which is not treated is still representing about 3% (about 0.5 mg/m^2/day) of the total average external load.
- The subsurface runoff and base flow which comprise 50% of the total runoff are very difficult to be avoided due to the fact that the SBP is a peat soil which causes enrichment of nutrient content (average TP concentration in the peat soil is 0.34 mg/l). The average load of subsurface runoff and base flow are representing about 15% (about 2.2 mg/m^2/day) of the total average external load.

- Even though the removal efficiency of the treatment of the pumps and inlet flow rates is 90%, the remaining 10% which is not treated still represent about 5% (about 0.7 $mg/m^2/day$) of the total average external load, due to their high concentrations. So upstream measures are needed to reduce the source of nutrients in these flow rates.

- The average load contribution of the ground water seepage and the precipitation to the Lake is 2% (about 0.3 $mg/m^2/day$) of the total average external load, which cannot be avoided.

The aforementioned unavoidable external loads to the Lake could be considered as natural background loads.

The predicted Lake TP concentration from a Vollenweider type model after treatment of external flows to the Lake was 185.96 μg/L, while the observed Lake TP concentration was 273.23 μg/L for the Year 2000, which mean that the reduction in the Lake TP concentration after treatment is 32%. This is due to extensive sediment TP internal loading (about 6.8 $mg/m^2/day$).

The mass balance models do not predict the long-term response of Lake TP to input reduction. If the lake has not yet reached equilibrium to the new reduced loading, they may under-predict Lake TP (Havens and James, 1997). Long-term response can be predicted by including a mass balance on sediment P (Chapra and Canale, 1991). However, such predictions have not yet been verified (Cooke et al., 2005).

The goal in diversion or treatment is to reduce Lake P concentration and case histories show that the mean concentration can be expected to decrease nearly in proportion to the reduction in inflow concentration on an annual basis. However, lake quality (algal biomass and transparency) is a function of the absolute equilibrium lake P concentration, not the proportional change in concentration. If P does not reach a limiting level, and the European lake evaluation suggested that until the summer epilimnion concentration of soluble reactive P (SRP) fell below a mean of 10 μg/L, algae would not be P limited, and even though lake TP declined, algal biomass would not respond (unless due to N reduction), then one cannot expect quality improvements following diversion or treatment.

Current understanding of internal loading and lake response to diversion or treatment suggests that sediment P release will decline, so one option following diversion/treatment is to wait and see if the trend in P release and lake quality satisfies lake users. If the choice is to ensure lake quality improvement, and Lake P is not expected to reach an algal-

limiting level, assuming that the release rate stays constant, then an in-lake treatment to control sediment release should be instituted soon after external controls are in place.

4.4.4 In-Lake Measures

The most common reason why lakes or reservoirs do not respond to controls on external inputs of nutrient is due to excessive internal loading or recycling of nutrients from bottom sediments, particularly P.

Cooke et al., (2005) have stated that in that case, one can proceed to the right side of Figure 4.11. If sediments are the source of internal loading and the bulk of nutrients are located in the top 0.3 to 0.5 m of a sediment core, then removal of that layer by dredging should provide the most reliable and permanent solution, although it will be the most costly. If sediments are rich in nutrients below that depth, then dredging would result in only exposing more sediment with the same high nutrient content providing little or no expected decrease in internal loading.

In that case, there are six measures that could be considered. These are arranged in sequence of their reliability and expected longevity for control of the nutrient source itself (Cooke et al., 1993).The six measures are: dilution/flushing, artificial circulation, and biomanipulation, nutrient inactivation, hypolimnetic withdrawal, and hypolimnetic aeration

Dilution/flushing, artificial circulation, and biomanipulation are, aimed at control of algal biomass or nutrient concentration and are not expected to control the source of loading.

Riplox, or sediment oxidation, is included with nutrient inactivation and although there has been limited demonstration, its major goal is the restoration of the upper sediment layer and therefore should provide an even longer-term solution than alum. Alum, on the other hand, simply covers the sediment with a floc layer and while its reliability at interrupting sediment P release has been excellent, the layer has been observed to sink through the sediment presumably exposing newly deposited, P-rich sediment that is available for release (Cooke et al., 2005).

Although the record for hypolimnetic withdrawal as a control for internal loading has not been as dramatic as that for alum addition, it has been demonstrated to be reasonably reliable and has the potential to deplete the sediment of nutrients (Kortmann et al., 1983; Nürnberg et al., 1987).

Hypolimnetic aeration has not been as effective as alum or sediment oxidation in controlling sediment P release, although it provides direct and effective re-aeration and coupled with iron addition has been effective at P control in some cases (Lean et al., 1986).

Alum addition is the least and dredging the most costly (Cooke et al., 2005).

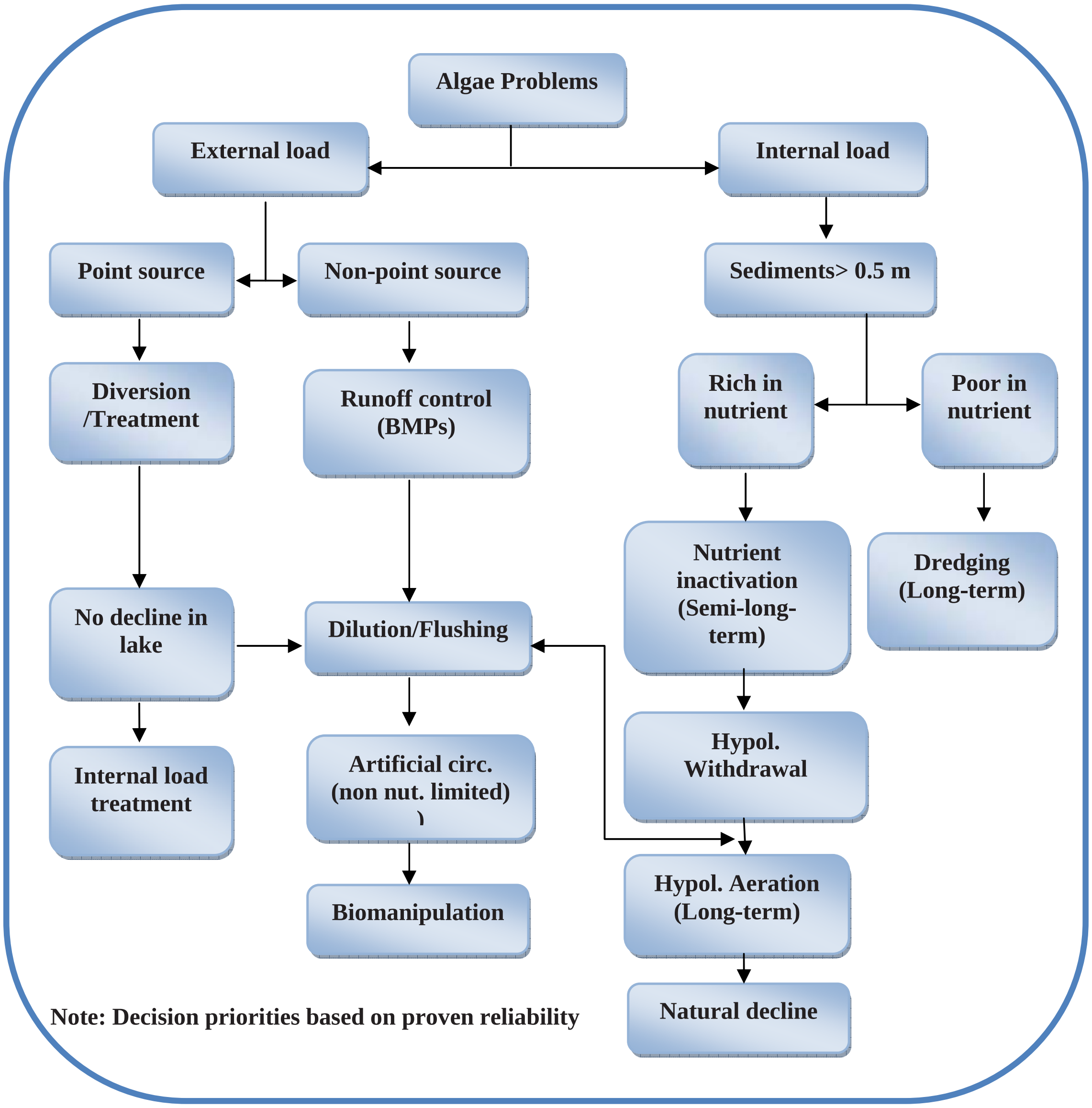

Figure 4.11 Decision tree for choice of best restoration procedures for control of algae problems. [*Source*: Cooke et al., 2005]

5 Conclusions, Recommendations, and Limitations

5.1 Conclusions

On the basis of the results brought about from this research, the following conclusions could be made:

i. The Trophic State Index for the Sloterplas indicates that the lake is a highly eutrophic lake. Despite nitrogen limited growth rate during summer, fixation of atmospheric N by blue-green algae should have supplied enough N to complement the available P supply. N fixation is not a rapid process; maximum rates of cell N replacement and growth of about 5% and 10%/d have been reported (Horne and Goldman, 1972; Horne and Viner, 1971)., the research focused on the phosphorus sources reduction because phosphorus can be made to limit algal growth if its concentration is sufficiently reduced.

ii. The TP steady state mass balance model used in this research to calculate the inflow flow rates to the Sloterplas Lake showed that about 65% of the total runoff, and 85% of the total inlet and pumps flow rates to the SBP are entering the Lake respectively.

iii. Both TP internal (32% of the total load) and external loads (68% of the total load) to the lake are contributing to the lake eutrophication problem, so attention has to be paid for both TP internal and external loads in order to restore the Lake. The runoff, pumps, and inlet are the most important external loads contributors, where they represent about 99% of the total external loads.

iv. Lake restoration could be achieved by external load control measures that can eliminate stress loadings, and internal load control through in-lake measures to accelerate the Lake return to earlier (more natural) conditions

v. The diversion of pumps flow rates from the lake would achieve about 41% load reduction of the external loads, which accounts for about 28% of the total loads (external and internal).

vi. The treatment of the pumps and inlets flow rates to the Lake could achieve external load reduction of about 43%, which accounts for about 30% of the total loads.

vii. Treatment will be better option to reduce the external load of pumps and inlet flow rates, because diversion is just problem shifting from one water course to another one without solving the problem of high TP load.

viii. Treatment of surface runoff could be achieved either by ecological engineering (BMPs), or by conventional engineering (alum treatment).Treatment of surface runoff flow rates by BMPs could achieve external load reduction of about 32% (about 21% of the total loads), while treatment by alum could achieve almost the same reduction (31% of external loads) as BMPs.

ix. Reviewing ecological engineering (BMPs), and conventional engineering (alum treatment) to treat the surface runoff has shown that the construction cost of the ecological engineering method is quite less than the treatment by alum, but the ecological engineering method requires quite large land area; however it can be used for recreation when dry, or serves as green space, supporting wet prairie functions and wildlife habitat. In addition ecological engineering is better from the environmental point of view because it is natural system, while the use of alum will introduce strange substances to the system (chemicals) and also the sludge resulted from the use of alum method is difficult to be handled and it could be environmentally harmful.

x. Neither treatment of pumps and inlets flow rates, nor treatment of runoff will be enough to reduce the external loads, so both of them should be used together in order to achieve the maximum external loads reduction. External loads reduction achieved by the combined measures was about 75% (about 49% of the total loads).

xi. Although the achieved TP reduction of the external loads to the Lake by the combined scenario is quite high, the lake will still be far away from the target condition. This is due to the fact that the lake will still receive unavoidable external loads (untreated portion of surface runoff, inlets and pumps, and subsurface runoff, base flow, and precipitation), which could be considered as natural background loads.

xii. The predicted Lake TP concentration from a Vollenweider type model after treatment of external flows to the Lake was 185.96 μg/L, while the observed Lake TP concentration was 273.23 μg/L for the Year 2000, which means that the reduction in the Lake TP concentration after treatment is 32%. This is due to extensive sediment TP internal loading (estimated to be about 6.8 $mg/m^2/day$).

xiii. A significant reduction in external nutrient loading is an essential, but not necessarily sufficient step towards reducing lake P concentrations. Internal loading from aerobic and anaerobic sediments, groundwater seepage, sediment resuspension, and organism activities might add more nutrients to the lake than external loading during some times of the year.

xiv. Lake quality (algal biomass and transparency) is a function of the absolute equilibrium lake P concentration, not the proportional change in concentration. If P does not reach a limiting level - and the European lake evaluation suggested that until the summer epilimnion concentration of soluble reactive P (SRP) fell below a mean of 10 μg/L, algae would not be P limited - algal biomass would not respond, thus one cannot expect quality improvements following diversion or treatment.

xv. So one option following diversion/treatment is to wait and see if the trend in P release and lake quality satisfies lake users. If the choice is to ensure lake quality improvement, and Lake P is not expected to reach an algal-limiting level, assuming that the release rate stays constant, then an in-lake treatment to control sediment release should be instituted soon after external controls are in place.

5.2 Recommendations

The following issues are recommended for future research:

i. Lake quality, or trophic state, is a direct result of their location within the landscape and nutrients and sediment that enter them from their watersheds. Thus, a thorough understanding of the watershed's characteristics (soils, slope, vegetation, tributaries, unique non-point nutrient sources, etc.) is necessary to explain the condition of the lake.

ii. Construction of an accurate water budget is the first step in diagnosing a lake's problem(s), because the substances that determine quality, or trophic state, originally are transported by water from the watershed. So continuous gauge recording is recommended to determine flow in major tributaries.

iii. A twice-monthly sampling frequency during May through September and monthly for the remainder of the year is recommended for temperate waters. Monthly measurements during summer may miss algal blooms completely and result in underestimated means for trophic state indices. Twice monthly sampling is also recommended for nutrient budgets.

iv. Whole-lake mean concentrations (sum of the products of depth-interval volumes and concentrations) or epilimnetic water column means are useful for assessing long-term change and the nutrient budget and models. Profile plots of TP, SRP, DO, and temperature for several dates in the summer may also be instructive to illustrate the effects of stratification and DO depletion on sediment P release. Volume weighted hypolimnetic TP plotted against time can be used to calculate a release rate from sediments.

v. More detailed study of the possible measures that could improve the lake water quality, containing not only the environmental demand but also the economic, social, and political demands should be done.

vi. Using pilot systems could help in assessing the possible measures that could improve the lake water quality.

5.3 Limitations

The following are the main limitations of this research:

i. Inflow Data used in this research are for the period from 1998 to 2002; data from most recent years were not available.

ii. The inflows to the polder may be not precise, due to the fact that pump flow rates are not precisely calibrated, groundwater seepage flow rate was assumed, and inlet flow rate was calculated from the water balance model used by Waternet.

iii. The inflows to the lake calculated by using the steady state mass balance model are not entirely precise, due to the uncertainties in determining the sedimentation rate coefficient.

iv. The internal load used in this research was obtained from a study done in 1996, and based on recent observations.

v. The water quality measurements were taken once every month, and even some months have no data.

6 References

AES (Applied Ecological Services), 2006. Alternative Stormwater Best Management Practices Guidelines. Public Works and Utilities Department Watershed Management Division City of Lincoln

Asaeda Takashi, Jagath Manatunge, Tilak Priyadarshana and Bae Kyung Park, (2004), PROBLEMS, RESTORATION, AND CONSERVATION OF LAKES AND RIVERS, in Oceans and Aquatic Ecosystems, [Ed. Eric Wolanski], in Encyclopedia of Life Support Systems (EOLSS), Developed under the Auspices of the UNESCO, Eolss Publishers, Oxford ,UK, [http://www.eolss.net] [Retrieved August 11, 2006].

Bass, B., and Baskaran, B., 2003. Evaluating Rooftop and Vertical Gardens as an Adaptation Strategy for Urban Areas. *Institute for Research and Construction*, NRCC-46737, Project no. A020, CCAF Report B1046. Ottawa, Canada: National Research Council.

Brezonik, P. L. & Stadelmann, T. H., 2002. Analysis and predictive models of stormwater runoff volumes, loads, and pollutant concentrations from watersheds in the Twin Cities metropolitan area, Minnesota, USA, Water Research, 36, pp. 1743–1757.

Brown, W. and T. Schueler. 1997b. The Economics of Storm Water BMPs in the Mid-Atlantic Region. Center for Watershed Protection. Ellicott City, MD.

Carlson, R.E. 1977. A trophic state index for lakes. Limnol. Oceanogr. 22: 361–368.

Carlson, R.E. 1981. Using trophic state indices to examine the dynamics of eutrophication. p. 218-221. In: Proceedings of the International Symposium on Inland Waters and Lake Restoration. U.S. Environmental Protection Agency. EPA 440/5-81-010.

Carlson, R.E. 1983. Discussion on “Using differences among Carlson’s trophic state index values in regional water quality assessment”, by Richard A. Osgood. Water Resources Bulletin. 19:307-309.

Chapra, S.C. and R.P. Canale, 1991. Long-term phenomenological model of phosphorus and oxygen for stratified lakes. Water Res. 25: 707–715.

Claytor, R. and T. Schueler. 1996. Design of Stormwater Filtering Systems. Prepared for the Chesapeake Research Consortium. Center for Watershed Protection. Ellicott City, MD.

Cooke, G.D. and R.E. Carlson. 1986. Water quality management in a drinking water reservoir. Lake and Reservoir Manage. 2: 363–371.

Cooke, G.D., Welch, E.B., Peterson, S.A., Newroth, P.R., 1993.Restoration and management of lakes and reservoirs, Lewis Pub-lishers, Boca Raton, FL.

Cooke.G. Dennis, Eugene B. Welch, Spencer A. Peterson, Stanley A. Nichols, 2005. Restoration and management of lakes and reservoirs, 3rd ed. CRC Press

CWP Center for Watershed Protection, 2006 Stormwater Manager's Resource Center (SMRC) Website www.stormwatercenter.net

Davis, A., M. Shokouhian, H. Sharma, and C. Henderson. 1998. Optimization of Bioretention Design for Water Quality and Hydrologic Characteristics. Department of Civil Engineering, University of Maryland, College Park.

Debo, Thomas N. and Reese, Andrew J., 1994Municipal Stormwater Management, CRC

Dillon, P.J. and F.H. Rigler. 1974a. A test of simple nutrient budget model predicting the phosphorus concentration in lakewater. J. Fish. Res. Bd. Can. 31: 1771–1778.

Edmondson, W.T. and J.R. Lehman. 1981. The effect of changes in the nutrient income on the condition of Lake Washington. Limnol. Oceanogr. 26: 1–28.

EUWFD, 2005. TOWARDS A GUIDANCE DOCUMENT ON EUTROPHICATION ASSESSMENT IN THE CONTEXT OF EUROPEAN WATER POLICIES

FHWA (USA Federal Highway Administration), 2005. Stormwater Best Management Practices Manual.

Galli, F. 1990. Peat-Sand Filters: A Proposed Stormwater Management Practice for Urban Areas. Metropolitan Washington Council of Governments. Washington, DC.

Gulati, R.D., E.H.R.R. Lammens, M.-L. Meijer and E.Van Donk. 1990. Biomanipulation: Tool for Water Management. Kluwer Academic, Dordrecht. The Netherlands.

Hadjinicolaou,J., 2003. Ecological Engineering in the Urban Environment, in *Sustainable Built Environment*, [Eds. Haghighat F., and Kim J.,J.], in *Encyclopedia of Life Support Systems (EOLSS)*, Developed under the Auspices of the UNESCO, Eolss Publishers, Oxford ,UK, [http://www.eolss.net] [Retrieved August 9, 2006]

Harlow C. Landphair, Jett A. McFalls, David Thompson, 2000. Design Methods, Selection, and Cost-Effectiveness of Stormwater Quality Structures

Harper, H.H. 1990. Long-Term Performance Evaluation of the Alum Stormwater Treatment System at Lake Ella, Florida. Project WM 339. Florida Department of Environmental Regulation, Tallahassee, FL.

Harper, H.H., M.P. Wanielista and Y.A. Yousef. 1983. Restoration of Lake Eola. In: Lake Restoration, Protection, and Management. USEPA 440/5-83-001. pp. 13–22.

Havens, K.E. and R.T. James. 1997. A critical evaluation of phosphorous management goals for Lake Okeechobee, Florida. Lake and Reserv. Manage. 13: 292–301.

Horne, A.J. and A.D. Viner. 1971. Nitrogen fixation and its significance in tropical Lake George, Uganda. Nature 232: 417–418.

Horne, A.J. and C.R. Goldman. 1972. Nitrogen fixation in Clear Lake, California. I. Seasonal variation and the role of heterocysts. Limnol. Oceanogr. 17: 678–692.

Hrbacek, J., M. Dvorakova, V. Korinek and L. Prochazkova. 1961. Demonstration of the effect of the fish stock on the species composition of zooplankton and the intensity of metabolism of the whole plankton assemblage. Verh. Int. Verein. Limnol. 14: 192–195.

Ierotheos, Z., Dimitriou, E. & Koussouris, T., 2003. Estimating Land Cover Changes and Associated Environmental Impacts on Wetlands by Coupling Remote Sensing and Hydrological Modelling, pp. 247–254 (Wallingford, UK: IAHS-AISH).

Johnston, W.H. and D.A. Sherwood. 1996. *Water Resources of Monroe County, New York, Water Years 1984-1988, With Emphasis on Water Quality in the Irondequoit Creek Basin. Part 2: Atmospheric Deposition, Ground Water, Stream Flow, Trends in Water Quality, and Chemical Loads to Irondequoit Bay*. Water-Resources Investigations Report 96-4054; Ithaca, New York: U.S. Geological Survey.

Jørgensen S.E. and Vollenweider R.A, 1988.Guidelines of Lake Management.Volume 1.Principles of Lake Management. United Nations Environment Programs.

Jorgensen, S. E. 1996. The application of ecosystem theory in limnology Verhandlungen der Internationalen Vereinigung fur Teoretische und Angevandle Limnologie, 26: 181-192

Kennedy, R.H. 1999. Basin-wide considerations for water quality management: importance of phosphorus retention by reservoirs. Int. Rev. Hydrobiol. 84: 557–566.

Kratzer, C.R. and P.L. Brezonik. 1981. A Carlson-type trophic state index for nitrogen in Florida lakes. Water. Res. Bull. 17: 713-715.

Line, D.E., N.W. White, D.L. Osmond, G.D. Jennings, and C.B. Mojonnier. 2002. Pollutant Export from Various Land Uses in the Upper Neuse River Basin. *Water Environment Research*, 74 (1): 100-108.

Livingston, E., E. Shaver, J. Skupien and R. Horner. 1997. Operation, Maintenance and Management of Storm Water Management Systems. Watershed Management Institute.Ingleside, MD.

Loga, M., E. Hollan, U. Lemmin, H. Baumert, C. Guilbaud, T. Frisk,D. Fontvieille,B. Wahl,H. Güde,L.May,L. Carvalho,K. Duwe, (2004) D38: *Key Processes in Integrated Water Resources Management for Important Deep European Lakes and their Catchment Areas*. Warsaw, Poland, EUROLAKES Project.

Lower Colorado River Authority (LCRA). 1998. Non-point Source Pollution Control Technical Manual, 3rd edition. Austin, Texas: LCRA.

Mallin, M.; Ensign, S.; Wheeler, T.; and Mayes; D. "Pollutant Removal Efficacy of Three Wet Detention Ponds." Journal of Environmental Quality 31: 654-660 (2002).

MDEQ (Michigan Department of Environmental Quality), 1999. STORMWATER MANAGEMENT GUIDEBOOK

Metropolitan Washington Council of Governments. 1987. *Controlling Urban Runoff: A Practical Manual for Planning and Designing Urban BMPs*.

Mitsch W.J., Jørgensen S.E., 1989. *Ecological Engineering-An Introduction to Ecotechnology*. John Wiley and Sons, New York.

Mitsch, W. J. 1993. Ecological Engineering - a co-operative role with planetary life-support system. Environmental Sciences & Technology 27: 438-445.

Moran, A.C., 2004. *A North Carolina Field Study to Evaluate Greenroof Runoff Quantity, Runoff Quality, and Plant Growth*. McS thesis, North Carolina State University.

MU (Middlesex University), 2003. Review of the Use of stormwater BMPs in Europe. Report 5.1. Project under EU RTD 5th Framework Programme

National Research Council, 1992. *Restoration of Aquatic Ecosystems: Science, Technology, and Public Policy*. Washington, D.C., National Academy Press.

NCDENR, North Carolina Department of Environment and Natural Resources Division of Water Quality,2005.*Manual Updated Draft Manual of Stormwater Best of Stormwater Best Management Practices*.

NCDERN (North Carolina Department of Environment and Natural Resources Division of Water Quality), 2005. Manual of Stormwater Best Management Practices.

Northeastern Illinois Planning Commission (NIPC). 1993. Urban Stormwater Best Management Practices for Northeastern Illinois.

Northern Virginia Planning District Commission (NVPDC). 1980. Guidebook for Screening Urban Nonpoint Pollution Management Strategies. Northern Virginia Planning District Commission. Falls Church, VA.

Nürnberg, G.K. 1984. The prediction of internal phosphorus loads in lakes with anoxic hypolimnia. Limnol.Oceanogr. 29: 111–124.

Nürnberg, G.K. 1984. The prediction of internal phosphorus loads in lakes with anoxic hypolimnia. Limnol. Oceanogr. 29: 111–124.

Nürnberg, G.K. 1987. A comparison of internal phosphorus loads in lakes with anoxic hypolimnia: Laboratory incubation versus in situ hypolimnetic phosphorus accumulation. Limnol. Oceanogr. 22: 1160–1164.

Nürnberg, G.K. 1988. Prediction of phosphorus release rates from total and reductant soluble phosphorus in anoxic sediments. Can. J. Fish. Aquatic Sci. 45: 453–462.

Odum E.P., 1983. Basic Ecology. New York: Saunders College Publishing.

Odum, H. T., 1962. *Man in the Ecosystem*. In Proceedings Lockwood Conference on the Suburban Forest and Ecology. Bull Conn. Agr. Station 652. Storrs, CT, pp.57-75.

Odum, H.T. (1971). *Environment, Power and Society*. Wiley, New York, 331 pp

Ostroumov S.A., 2006. *Biomachinery for maintaining water quality and natural water self-purification in marine and estuarine systems: elements of a qualitative theory*.

International Journal of Oceans and Oceanography. Vol.1, No.1 (2006), pp. 111-118

Palmarsson, S. O., 2000. *Littoral Water Currents in a Shallow Lake*. *Civil and Environmental Environmental Engineering.* Davis, California, USA, University of California.

PCSL (Pennsylvania Clean Streams), 2006. Pennsylvania Stormwater Best Management Practices Manual.
Press, Inc.

Reckhow, K.H. and S.C. Chapra. 1983. Engineering Approaches for Lake Management: Vol. I. Data Analysis and Empirical Modeling. Butterworths, Boston, MA.

Reckhow, K.H. and S.C. Chapra. 1983. Engineering Approaches for Lake Management: Vol. I. Data Analysis and Empirical Modeling. Butterworths, Boston, MA.

Robarge, W.P. and W.W. Cure, 1999. *Quantification of Atmospheric Nitrogen Deposition in the vicinity of a Large Scale Swine Production Facility Located in the Neuse River Basin*—Final Report to the North Carolina Division of Air Quality, December 31, 1998. *Status Report on Emissions and Deposition of Atmospheric Nitrogen: Compounds from Animal Production in North Carolina.* Raleigh, North Carolina: NCDENR, Division of Air Quality.

Robertson, B., R. Pitt, A. Ayyoubi, and R. Field. 1995. A Multi-Chambered Stormwater Treatment Train. In Proceedings of the Engineering Foundation Conference: Stormwater NPDES-Related Monitoring Needs, Mt. Crested Butte, Colorado, August 7–12, 1994, American Society of Civil Engineers, New York, NY.

Scheffer, M. (1998). Ecology of shallow lakes. Chapman & Hall.

Schueler, T., 1987. *Controlling Urban Runoff: A Practical Manual for Planning and Designing Urban BMPs*. Metropolitan Washington Council of Governments. Washington, DC.

Shapiro, J., V. LaMarra and M. Lynch. 1975. Biomanipulation: An ecosystem approach to lake restoration. In: P.L. Brezonik and J.L. Fox (Eds.), Proceedings of a Symposium on Water Quality Management and Biological Control. University of Florida, Gainesville, FL. pp. 85–96.

Shuster, J.I., E.B. Welch, R.R. Horner and D.E. Spyridakis. 1986. Response of Lake Sammamish to urban runoff control. Lake and Reservoir Manage. 2: 229–234.

Slone, D.K. and D.E. Evans, 2003. *Integrating Green Roofs and Low Impact Design into Municipal Storm Water Regulations*. In *Proc. Greening Rooftops for Sustainable Communities: Chicago 2003*: May 29-30, 2003; Chicago, Illinois.

Southeastern Wisconsin Regional Planning Commission (SWRPC). 1991. Costs of Urban Nonpoint Source Water Pollution Control Measures. Waukesha, WI.

Straskraba, M. (1985). *Simulation Models as Tools in Ecotechnology Systems*. Analysis and Simulation, Vol.II. Academic Verlag, Berlin.

SWITCH, 2007. http://www.swithchurbanwater.eu, accessed on January 2007.

U.S. EPA, 1999. National Management Measures to Control Nonpoint Source Pollution from Urban Areas. EPA-841-B-05-004

U.S. EPA, 1999. Preliminary Data Summary of Urban Storm Water Best Management Practices, EPA-821-R-99-012

U.S. EPA. 1994. *National Water Quality Inventory: 1992 Report to Congress*. EPA 841-R-94-001. Washington, D.C.: U.S. Environmental Protection Agency.

U.S. EPA. 2002a. *National Water Quality Inventory: 2000 Report*. EPA-841-R-02 001. Washington, D.C.: U.S. Environmental Protection Agency.

Urbonas, B. and Stahre, P. (1993). “Stormwater-Best management practices and detention for water quality, drainage, and CSO management.” PTR, Prentice Hall, Englewood Cliffs, New Jersey.

Urbonas, B., and Ruzzo, W.P. 1986. Standardization of Detention Pond Design for Phosphorus Removal. In Procs. NATO Advanced Research Workshop, France, Aug. 1985. NATO ASI Series, Vol. G10.

Uttormark, P.D. and M.L. Hutchins. 1980. Input/output models as decision aids for lake restoration. Water Res. Bull. 16: 494–500.

Van der Veen, C., A. Graveland and W. Kats. 1987. Coagulation of two different kinds of surface water before inlet into lakes to improve the self-purification process. Water Sci. Technol. 19: 803–812.

Vollenweider, R.A. 1969a. A Manual on Methods for Measuring Primary Production in Aquatic Environments. IBP Handbook, No. 12. Blackwell Scientific Publishers, Oxford.

Vollenweider, R.A. and P.J. Dillon. 1974. The application of the phosphorus-loading concept to eutrophication research. National Research Council, Canada, Tech. Rept. 13690.

Wiegand, C., Schueler, T., Chittenden, W., and Jellick, D. 1986. "Cost of Urban Runoff Controls," Urban Runoff Quality Impact and Quality Enhancement Technology, Proceedings of an Engineering Foundation Conference, June 23-27, 1986, Henniker, NH, B. Urbonas and LA Roesner, ed., , New York, New York: American Society of Civil Engineers

Wiegand, C., T. Schueler, W. Chittenden and D. Jellick. 1986. "Cost of Urban Runoff Controls." Urban Runoff Quality Impact and Quality Enhancement Technology. Proceedings of an Engineering Foundation Conference, June 23-27, 1986, Henniker, NH. B. Urbonas and L.A. Roesner, ed. American Society of Civil Engineers. New York, NY.

Winer, R., 2000. *National Pollutant Removal Performance Database for Stormwater Treatment Practices*. 2nd Edition. Center for Watershed Protection. Ellicott City, MD.

Witteven Bos, 1996. *Eutrofiëring Westelijke Polders.*Gemeente Amsterdam,dienst RWA.

Wu, J.S., C.J. Allan, W.L. Saunders, and J.B. Evett. 1998. *Characterization and Pollutant Loading Estimation for Highway Runoff. Journal of Environmental Engineering*, 124 (7): 584-592.

Young et. al., 1996. Evaluation and Management of Highway Runoff Water Quality (Water Quality Synthesis).

Yu, S.L., S.L. Barnes, and V.W. Gerde. 1993. Testing of Best Management Practices for Controlling Highway Runoff. Virginia Transportation Research Council, Charlottesville, VA.