Urban surface water quality enhancement

A case study on a floating treatment system

M.R. van Dieren

Master of Science Thesis

Delft, July 2011

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Preface and Acknowledgement

This thesis presents the research project conducted to complete the Master's program Water Management at the faculty of Civil Engineering and Geosciences of the Delft University of Technology. The subject of this thesis is the enhancement of surface water quality in urban areas focusing on a floating treatment system developed by the Bright Water Company.

During my studies in Delft I got acquainted with many civil engineering subjects. The international aspects of water management even gave me the opportunity to carry out part of my studies abroad. My scientific curiosity roots nonetheless in the Netherlands, and I preferred to conclude my Master on a Dutch issue. The combination of environmental science and urban challenges triggered me to do my thesis research project on urban surface water quality enhancement in the Netherlands.

I would like to thank the members of my graduation committee for their support during my research and for their compliance with the struggles I confronted. Prof.dr.ir. Nick van de Giesen gave me the opportunity to explore this rather unconventional aspect of water management in Delft. Dr.ir. Frans van de Ven introduced me to water management in urban areas and provided me with relevant background information for this research. Drs. Bas van der Wal showed me the focus of water quality management and dr.ir. Marie-claire ten Veldhuis helped me with the start up and introduced me to Matthijs Siers. I am especially grateful for the opportunity Matthijs gave me to work with his company and the freedom he gave me studying the floating helophyte filter. Conducting measurements together on a small boat during a rainy day in Groningen might not provide the ideal circumstances but resulted in a great cooperation! The application of the filters in Groningen brought me in contact with ing. Greetje Kampinga of the water board Noorderzijlvest and ing. Anne Helbig of the municipality of Groningen. I would like to thank them for their information on the Floresvijver and on the applied water management policies. Furthermore, I would like to express my gratitude to ir. Tonny Schuit, who assisted me with my laboratory analysis. My special thanks goes to drs. Gerda Bolier, who provided me with advice throughout the whole process.

Finally, I would like to thank my family and friends who supported me and believed I could successfully accomplish what is now right in front of you.

Maarten van Dieren

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Abstract

Ongoing urbanisation and the subsequent extensive use of the urban water system can lead to degradation of its surface water quality. In the Netherlands, urban water bodies often suffer from the manifestations of eutrophication due to (historically) high nutrient loadings. This research focused on the enhancement of surface water quality of semi-confined urban water bodies with a case study on a floating treatment system.

Urban water bodies often function as amenities of the urban area. Their ornamental and ecological value depends on the state of their aquatic ecosystem. Excessive nutrient loading, leading to the collapse of the system's biodiversity, turns a water body into a turbid state without submerged macrophytes. Restoration of the clear water state through reduction of nutrient loadings alone is hindered by hysteresis caused by the ecosystem relations and could be supplemented with an approach focused on increasing the system's nutrient carrying capacity or an internal approach directly targeting the manifestations of eutrophication.

The Bright Water Company floating helophyte filter actively drains a filter bed with bog plants growing in it. The influent of this biofilter is provided by free inflow of surrounding surface water. Its water treatment ability depends predominantly on filtration and adsorption by specific nutrient absorbents. Additionally, its inner reservoir serves as a habitat for small aquatic organisms.

Insights on the functioning and applicability of the biofilter were gained through in situ measurements. Two biofilter were applied in the Floresvijver in Groningen and measurements were conducted on influent, effluent and surface water. Visual observations and laboratory analyses of the water samples showed effective filtration and daphnia flourishing in the inner reservoir of the biofilter. Accumulation of the residual solids as a sludge layer on top of the filter bed and formation of biogas inside the filter material proved to reduce the hydraulic capacity significantly. Nutrient removal efficiency could not be determined with the monthly measurements of the water board but for optimal functioning of the phosphorous absorbent the current filter bed design should be adjusted while effluent samples indicated leaching of absorbent components.

Application of the biofilter can contribute to the enhancement of urban surface water quality by increasing the nutrient carrying capacity of a water body. Especially in urban areas with various diffusive nutrient sources and physical constrains, the application of the biofilter can be efficient. Additionally, the biofilter functions as a habitat for zooplankton which are an important ecosystem element for the prevention of algae blooms. Furthermore, the effluent of the biofilter can provide a local increase in transparency and improve conditions for macrophyte development. The number of biofilters applied in a water body determine the significance of these contributions relative to the existing conditions.

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Chapter 1

Introduction

1.1 Introduction

In today's society, ongoing urbanization increasingly leads to the accumulation of people and their activities in urban areas. The subsequent extensive use of the water system and the often unconstrained degradation of its aquatic ecosystem can limit a prosperous and sustainable development. On the other hand, well functioning urban water systems can contribute greatly to the well being of people integrated with vital ecosystem services creating habitats for biodiversity in urban areas.

Historically the main function of Dutch urban water systems is to drain, store and discharge water to the surroundings to prevent water quantity nuisances like pluvial flooding and excessive groundwater levels. Since the nineteen seventies the attention for water quality management in urban areas gained interest intensely. The introduction of a law to reduce surface water pollution (i.e. Wet verontreiniging oppervlaktewateren) stimulated for example the construction of sewer systems to discharge sewerage from households and industries to water treatment plants. In this way major improvements were established, but the focus was predominantly on chemical parameters as indicators of the systems water quality (e.g. MTR-normen). In recent years the thinking in water quality management transformed into a more ecological based approach, with the introduction in 2000 of the European Water Framework Directive (Europese Kaderrichtlijn water) as a major stimulant for quality assessments based on ecological values.

Even in urban water systems with its intensive human influence, the state of the aquatic ecosystem plays an important role in its appraisal nowadays. In part, this is because surface water bodies in the urban environment are amenities and the improvement of their aquatic ecosystem increases their added value for the residents (Van Noort et al., 1998; VROM, 2005). This new approach, however, also generates new problems. In fact, to improve an aquatic ecosystem, a single solution or measure is often not available or sufficient.

Figure 1.1 Water as an amenity of the urban area.

Disturbance by excessive nutrient loading (eutrophication) is one of the main problems for Dutch urban water systems. It can cause water bodies to turn into a monotonous pool of algae, even with the chance of creating a human health risk. If a water system suffers from eutrophication, an approach focused on the removal of the excessive nutrient source(s) alone is often not feasible and is even not directly going to improve the state of the aquatic ecosystem due to a process called hysteresis. Additional measures are therefore needed to restore the system and its biodiversity (Scheffer, 1998).

In urban areas, space is limited and the water system is often confined to strict boundaries. This limits the possible measures which can be applied. On top of that the function of a water body as a spatial or even architectural element limits the possible alterations of its physical structure (e.g. no swamps in the canal system of the city centre).

1.2 Extent of the research

The urban surface water system typically includes man-made water bodies that are principally constructed to provide services such as the drainage of urban areas. In Dutch cities, the artificial surface water system usually includes numerous (small) ponds. They are constructed as part of the drainage system (or even the sewer system) to create the required storage capacity within the system. On the other hand, these ponds are often technically ‘over-dimensioned’ to function as blue/green spatial elements being an amenity for the residents sometimes as part of an urban park. This thesis focuses on these semi-confined ponds, their particular conditions and the possibility to improve their ecological status.

The floating helophyte filter of the Bright Water Company has an comprehensive approach for the enhancement of the water quality of urban surface waters. The significance and effectiveness of the different contributions to this approach are unknown. This research will examine this floating treatment system and focus on its application in a semi-confined pond in the city of Groningen.

1.3 Research objective

The main objective of this research is to:

Create insight in the possible contribution of the Bright Water Company floating helophyte filter to the improvement of the (ecological) water quality of semi-confined urban surface water bodies.

1.4 Research question

The main objective can be translated into the research question:

What can be the possible contribution of the Bright Water Company floating helophyte filter to the improvement of the (ecological) water quality of semi-confined urban surface water bodies?

This research question can be split in several sub questions:

- Which (ecological) water quality aspects are relevant for semi-confined urban surface waters?
- What is the current approach in urban water quality management?
- What is the functioning of the floating helophyte filter?
    - What is the filtration efficiency of the filter?
    - What is the nutrient removal capacity of the filter?
    - What can the application, as a habitat, contribute to the biodiversity of a surface water body?

1.5 Research approach

To answer the research questions and achieve the research objectives, the initial focus is on principle processes and management of the urban water system. A literature study is conducted on these subjects and forms the foundation for the rest of the research. Subsequently, a survey is done on existing measures and applications for improving the water quality of urban water systems. And consequently, the functioning of the Bright Water Company floating helophyte filter is examined. Knowledge on the significance and effectiveness of all processes and aspects of this biofilter is gained. The latter is done in a case study including in situ measurements.

1.6 Outline of the report

This report is an overview of the conducted research. Chapter 2 describes the most important water quality aspects in urban surface waters and their interactions. Chapter 3 deals with the management issues including the laws and regulations applicable to urban water quality management. This chapter also give the historic development of the management approach and explains the current state of water management of urban surface water quality in the Netherlands. Chapter 4 gives an overview of the available types of measures and their application, elaborating on the application of floating treatment systems. Chapter 5 deals with the case study on the Bright Water Company floating helophyte filter and its application in a semi-confined pond in the city of Groningen. In chapter 6 the final conclusions and recommendations of this research are presented.

Chapter 2

Urban surface water quality

2.1 Introduction

A water system can be defined as the coherent entity of surface water, groundwater, aquatic soils, banks and technical infrastructure, including the community of organisms present in it and all accompanying physical, chemical and biological characteristics and processes. The water system in a particular urban area forms an urban water system. This urban water system is part of a larger catchment area but it diversifies from its surroundings by several aspects:

- It is physically separated and water quantity (i.e. the water levels) and water quality within the created sub shed is more or less regulated.
- The increased building density and the accommodating infrastructure create large areas of impermeable surface causing increased stormwater runoff, altered runoff patterns and often deterioration of runoff quality.
- The drainage is further influenced by the presence of sewer systems which also deal with significant volumes of wastewater resulting from the increased population density.

Urban surface water is a term which includes various types of surface water bodies all situated in the build-up area. Although this is not a classical distinguished category of water bodies, they have communalities and specific characteristics due to the anthropogenic 'element' which also influence the relevance of the different water quality processes.

The individual waters in Dutch urban water systems are usually man-made or at least canalized and have an artificial morphology often with banks of wooden boards or other constructions to avoid erosion. This, together with strictly controlled water levels, leads to monotonous riparian zones with very limited transition zones between water and land (important to the development of various ecological relations). Maintenance to retain the quantitative functioning regularly disturb the water bodies and can prevent for example the succession of the flora or the natural silting up by sedimentation. Combined with the (historical) loading of the water system with significant amounts of nutrients and contaminants, the aquatic ecosystem is often degenerated and not very robust (i.e. vulnerable to further deterioration by perturbations). Tolerant species will dominate while sensitive species diminish or disappear entirely (Walsh et al., 2005).

Semi-confined urban ponds are especially susceptible to (incidental) problematic manifestations of the degradation. These stagnant water bodies with limited water depth can for example warm up rapidly during spring or summer. With no supply of clear and fresh water for flushing, the oxygen concentrations can radically drop during algae or duck weed blooms resulting in all kinds of nuisances (e.g. bad odour, dead fish). Furthermore low transparency or low biodiversity reduces the pond's value to humans and nature. Blue-green algae blooms or outbreaks of botulism can even be a threat to their well being.

Figure 2.1 In a canal, flushing can remove manifested quality problems (e.g. blooms of duck weed).

The following sections describe the key aspects in the deterioration of aquatic ecosystem quality relevant to semi-confined ponds. Turbidity is thereby the prime indicator of aquatic ecosystem development, with nutrients as the main initiator of the degradation itself. The process of eutrophication is described and the relations between the elements of the ecosystem which obscure or promote the change of its status are explained.

2.2 Turbidity: under water optics

By using the energy from the sun to convert carbon into organic compounds, the process of photosynthesis is the source of virtual all biomass on earth and thus essential for all ecosystems. The transparency of a water body determines the incidence of photosynthesis and is therefore an important determinant of its condition and productivity. Consequently, under-water optics are crucial for understanding the functioning of surface water bodies.

Figure 2.2 Limited turbidity allows sunlight to penetrate the water column.

When light (i.e. a photon) enters surface water, it will be either scattered or absorbed. This causes the extinction of light over depth; vertical light attenuation. Although the water itself contributes to absorption and scattering of light, the optical properties of surface water depends largely on suspended particles and dissolved substances (Scheffer, 1998).

Absorption is due to all coloured matter. In urban water bodies light is absorbed by particulate matter such as phytoplankton, detritus and suspended sediment particles but dissolved organic substances can contribute to this phenomenon as well; primarily in peaty catchments, humic substances give the water an explicit brown stain.

The relative contribution of different materials to scattering and absorption varies strongly. Suspended clay particles for example, predominantly cause scattering while phytoplankton contributes to both scattering and absorption. Overall, the inherent optical properties of absorption and scattering are well defined and measurable for all individual contributors (Kirk, 1994). However, the relation between the sum of all these inherent properties and the apparent optical properties of a surface water body is not straight forward. This is due to the scattering. Scattering does not remove light like absorption but does increase the path length of a photon and thereby the chance of it being absorbed.

The apparent optical properties of a surface water body (its visual transparency) are often measured by means of a black and white disk called a Secchi disk (after its inventor). The depth at which it is visible again when lifting it after it was completely lowered into the water, is called the Secchi-depth (Sd). The method and materials used are very simple and explain its widespread use among scientific researchers. However, there are a number of disadvantages to this method. First of all, it can only be applied in situ and in systems with a water depth is larger than Secchi-depth. Furthermore, Secchi-depth is much more effected by scattering than the vertical light attenuation is. So this measure of visual transparency cannot easily be translated into light attenuation. Additionally, the intensity of the incoming light has a profound influence on the Secchi-depth, unlike the vertical light attenuation coefficient (Kd) which is relative to the irradiance just under the water surface. The capacity and experience of the observer can also contribute to the accuracy of the measurements while executing them (e.g. repetitive consistency, eyesight capacity). All this results in an approximate accuracy of the Secchi of 10^{-1} m.

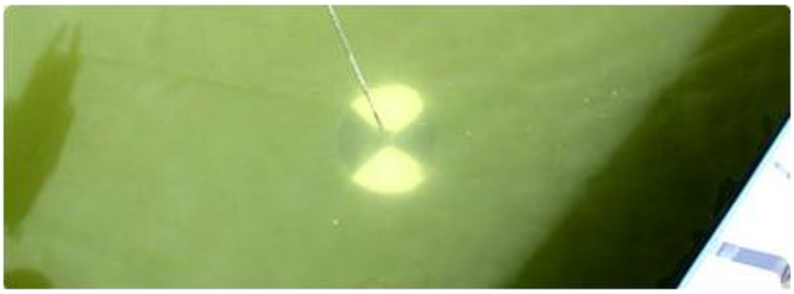

Figure 2.3 Measuring Secchi depth.

The preferred characteristic of the optical properties of a surface water body depends on its purpose. Light attenuation is useful when considering photosynthesis by means of the inverse proportional euphotic depth (Z_{eu}= 3,7/K_{min} where K_{min} is the vertical attenuation coefficient for the most penetrative waveband). Euphotic depth is the depth beyond which the light falls below 1% of the surface irradiation and is considered the boundary for positive net photosynthesis of an algal cell (Moss, 1998).

Secchi depth is more important when the impression of clarity is relevant. Visually hunting fish (e.g. roach and pike) and the appreciation by humans largely depend on it.

In laboratory research, turbidity is often expressed in nephelometric turbidity units (NTU). This value is determined by sending a light beam through a glass cylinder with the water sample and measuring the amount of scattered light at an angle perpendicular to the beam. Obviously, the nephelometric turbidity corresponds closely to the scattering coefficient so the relation to Secchi depth and vertical light attenuation is weak.

2.3 Nutrients

Nutrients are substances that are essential for life. Some nutrients are required in relative large quantities, the so-called macronutrients. Carbon is the most important, making up half of the dry weight of most organisms. Other macronutrients like nitrogen and phosphorus occur in biomass only in concentrations of a few percent or even less. That is however still far more than the concentrations of micronutrients. These trace elements like iron and zinc occur in concentrations less than 10 parts per million. The ratio of the required nutrients to each other is called the stoichiometry.

It is well known from literature that in fresh water ecosystems, phosphorus and nitrogen occur in concentrations that make them the main limiting nutrients of (algal) growth. That's why the availability of these elements in surface water is used as an indicator of the productivity of its aquatic ecosystem. The stoichiometry of carbon, nitrogen and phosphorus at balanced growth of phytoplankton is referred to as the Redfield ratio (Redfield, 1958).

Carbon

The carbon cycle is the basis of all biogenic processes. Inorganic carbon is abundant in the atmosphere, mainly as carbon dioxide (CO_2). Dissolved in water, it persists in various forms depending on the pH. Autotrophic organisms like plants and algae can fixate carbon in organic material (primary production). In this way it enters the food web where it can be assimilated by heterotrophs. The growth of autotrophic organisms like phytoplankton or macrophytes is rarely limited by the availability of carbon and therefore the carbon cycle is not addressed in detail in this research.

Nitrogen

The worlds atmosphere consists for 78% of nitrogen gas (N_2). Water generally contains N_2 as a dissolved gas. For organisms however, the N_2 molecule is very difficult to use directly (Dodds, 2002). Two other forms of dissolved inorganic nitrogen are ammonium (NH_4^+) and nitrate (NO_3^-). Ammonium in water is in equilibrium with ammonia gas (NH_3) which can be exchanged to the atmosphere.

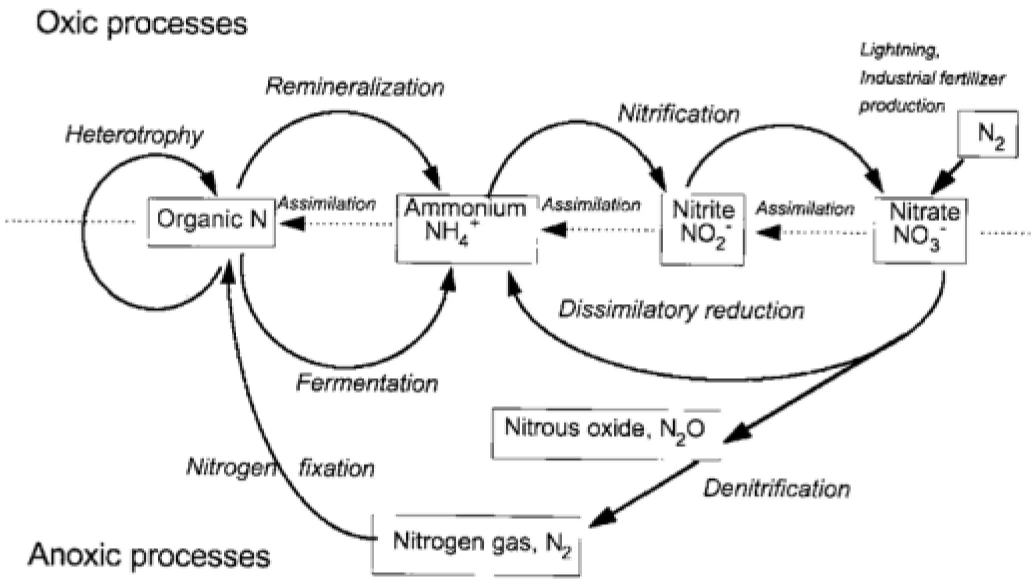

Figure 2.2.4 Conceptual diagram of the nitrogen cycle (Dodds, 2002)

Ammonium can be transformed microbially to nitrate under aerobic conditions. This process is called nitrification and consists of two oxidation steps carried out by two different groups of bacteria. One group (e.g. Nitrosomosas) is responsible for the oxidation of ammonium into nitrite (NO_2^-), and the other (e.g.

Nitrobacter) for the oxidation of nitrite into nitrate. The latter process works relatively fast so in general nitrite concentrations stay low.

Most bacteria and primary producers in fresh water prefer to consume nitrogen as ammonium but often have to use nitrate because of availability. Some bacteria, like some cyanobacteria (also called blue-green algae), have the capacity to directly assimilate N_2. This capacity is called nitrogen fixation and can be an important contribution to the nutrient balance of fresh water systems during anaerobic conditions (Moss, 1998).

Ammonium itself is released by microbial decomposition of organic material. This mineralization of detritus is called ammonification. Furthermore, aquatic organisms primarily excrete excessive nitrogen in the form of ammonium. Other creatures often excrete nitrogen in the form of urea in their urine.

When anaerobic conditions occur, nitrate can be reduced by the microbial process of dissimilatory nitrate reduction. Some bacteria reduce nitrate to ammonium but more commonly, nitrate is reduced through a series of intermediate gaseous nitrogen oxide products to nitrogen gas. This denitrification process requires anaerobic conditions and at the same time the availability of nitrate which is produced under aerobic conditions. That is why denitrification predominantly occurs in the surface layer of (underwater) soils where these conditions co-occur or alternate in time.

Phosphorous

Phosphorus occurs in surface waters in various forms. First of all, a distinction can be made between particulate and soluble phosphorus. The particulate fraction consists of phosphorus absorbed to soil particles or precipitated as inorganic complexes formed with metals like iron, magnesium or aluminium. The phosphorus in tissue of organisms like algae, zooplankton or fish are part of the particulate fraction too.

The dominant inorganic dissolved form of phosphorus in surface water is phosphate (PO_4^{3-}). The basic form is also called orthophosphate and is directly available for algae growth. For the estimation of this immediately available phosphorus, chemical methods are used to determine soluble reactive phosphate (SRP). Other dissolved forms of phosphorus are polyphosphate and organic dissolved phosphate from lysis of dead organisms.

- Particulate P
    - Absorbed to soil particles (e.g. clay or peat)
    - Absorbed to metals (e.g. iron, aluminium or calcium)
    - Bound in tissue of (dead) organisms (e.g. algae, zooplankton or fish)
- Soluble P
    - Orthophosphate (estimated by SRP)
    - Polyphosphate (polymers)
    - Dissolved organic phosphorus (by lysing of particulate organic phosphorus)

The bioavailable fraction of phosphorus is surface water is generally very small but has a high turnover rate (minutes) so it can be easily released from, for example, the bottom sediment.

2.4 Eutrophication

Nutrient levels provide a basis to classify surface water bodies. They determine the trophic state of a water body, ranging from oligotrophic, with low levels of nutrients, via mesotrophic to eutrophic in case of highly loaded waters. In strict sense eutrophication is the transition from the oligotrophic state towards the eutrophic state by accumulation of nutrients.

In fresh surface waters, phosphorous is often the prime limiting nutrient and therefore a suitable indicator for the tropic state. During the nineteen seventies a positive and causal correlation between phosphorous levels and algal densities was observed in various deep upland lakes (see Figure 2.5) Based upon the “Vollenweider-model” and the according classification of tropic states (OECD, 1982), the perception was established that ecological water quality problems (i.e. eutrophication problems) were caused by eutrophication in strict sense (i.e. phosphorous loading).

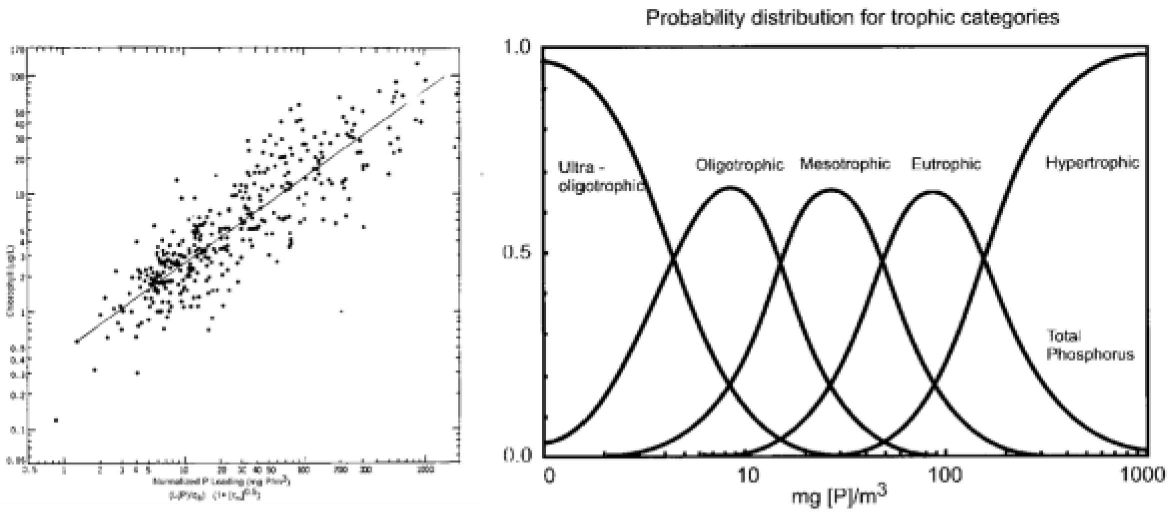

Figure 2.5 Vollenweiders correlation (Vollenweider et al., 1980) and tropic classification(right).

Although phosphorous is a suitable rough indicator for the trophic status of a water body, this does not imply that it is a reliable (i.e. causal) predictor of the manifest eutrophication problems (Reynolds, 1992). The Vollenweider-model already showed a broad variation in the relationship, and in practice mesotrophic waters can appear eutrophicated while eutrophic waters do not. So it is necessary to look beyond nutrients to the many other variables that influence the manifestation of eutrophication problems.

When an oligotrophic water body is loaded with nutrients, the nutrients are at first used by the aquatic plants (macrophytes). Algal growth is limited, also due to the simultaneous increase of zooplankton which feed on them. The zooplankton itself is protected against predation by fish through the refuge of the macrophytes. Nevertheless, fish stock increases with the abundance of available food. Submerged macrophytes continue to expand until they are shading themselves. At that point, algae can profit from the surplus of nutrients and increase their abundance and thereby decreasing the light attenuation (see also §2.2). This will lead to an increase of turbidity to such extend that the macrophytes can no longer persist. At that point zooplankton loses its shelter which results in a strong reduction of their numbers by predation. As a consequence the algal community can flourish and succession of species can lead to dominance of blue-green algae (cyanobacteria). The fish community alters too. Bethivorous species (e.g. bream and carp) become dominating and most of the piscivores (e.g. pike and perch) disappear.

Figure 2.6 Blue-green algae blooms in urban ponds.

This eutrophicated situation shows a totally different aquatic ecosystem. And reduction of the nutrient loading will not lead to a similar transition in reverse order. To the contrary, it turns out that several factors hinder the restoration of the initial clear water state. Benthivorous fish stir up the bottom sediment keeping turbidity high, giving submerged macrophytes no chance to grow. Blue-green algae tend to cope with the available phosphorous very efficiently and for long periods after external loading internal buffers can deliver nutrients (e.g. from phosphorous storage in the bottom sediment). This new situation turns out to be very resilient and can be assigned as a new stable state. The occurrence of two alternative equilibrium states of an ecosystem under similar nutrient conditions is called the hysteresis effect. And with it, the state of the aquatic ecosystem is indicated by the level of its turbidity (see Figure 2.7).

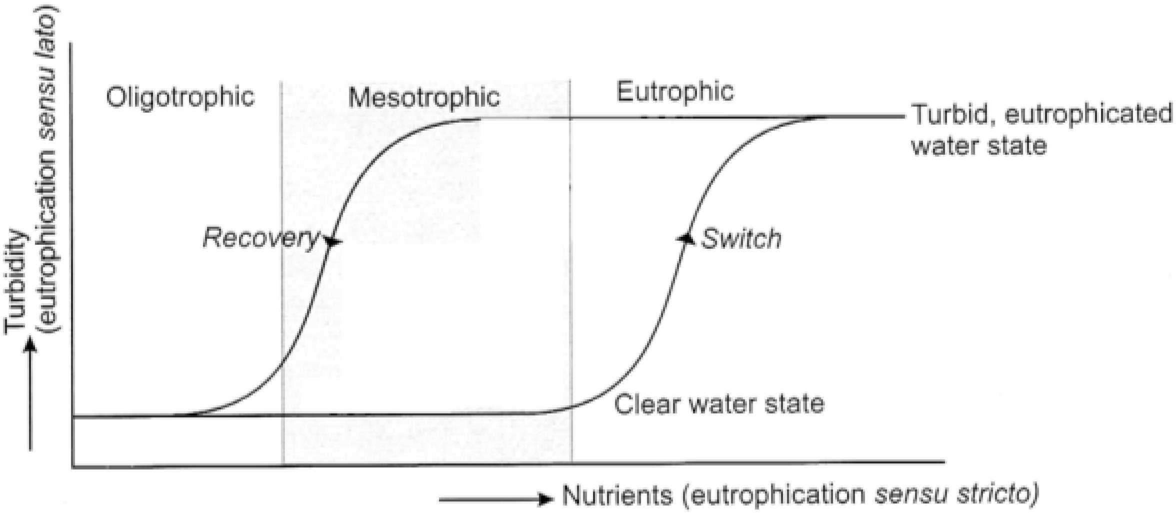

Figure 2.7 Hysteresis effect (redrawn from Scheffer et al., 1993 and Scholten et al., 2005).

2.5 Ecosystem relations

The occurrence of hysteresis in the response of water bodies to changes in nutrient loading is caused by the interdependencies of the aquatic ecosystem elements. In the previous section some of the relations are already described. Figure 2.8 shows the interdependencies of some of the mayor factors in a schematic way. In practice the relations between the different elements are not always as straight forward but the general tendency demonstrates the persistence of the two alternative equilibrium states (i.e. the clear water state vs. the turbid state).

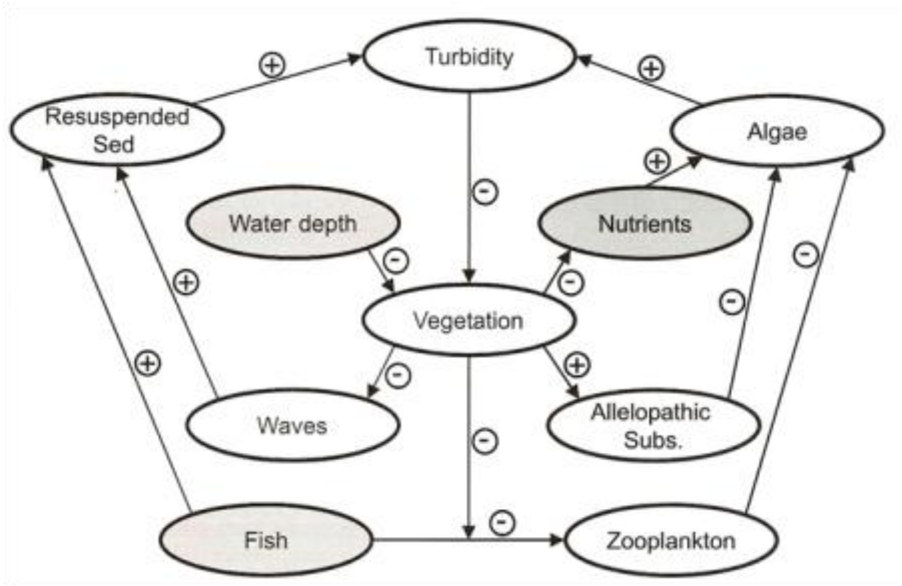

Figure 2.8 Feedback patterns between main ecosystem elements (Scheffer et al., 1993).

At the base of the food web the primary producers take up available nutrients. Algae and macrophytes compete over the existing resources. Where algae respond opportunistically to changes in availability, macrophytes have a more consistent growth during the year as they can store nutrients and acquire them from the sediment with their roots. Algae and macrophytes also compete over the penetrating sunlight. Shading of macrophytes may inhibit algal growth but when algae increase turbidity levels to such extend that light cannot reach through the water to the sediment, the development of macrophyte sprouts is hindered. Some macrophytes can also suppress growth of specific algal species by allelopathy (i.e. releasing of allelochemicals) (Jasser, 1995).

The most important primary consumers of algae are the zooplankton groups (e.g. cladocerans, rotifers and copepods). Cladoceran water fleas (Daphnia) are the most effective algae grazers due to their relative large size and indiscriminate feeding habit (Scholten et al., 2005). Combined with their high reproduction rates, they have a substantial impact on algae density. Benthic filter feeding bivalves (e.g. zebra mussels) can be significant too but the lack of hard substratum in urban ponds usually prevent substantial colonisation. Adaption of their reproductive rate is also limited compared to algae population dynamics.

Daphnia have the potential to graze down algal biomass to very low levels. Early spring blooms of algae often lead to expansion of its population resulting in the well documented spring clear-water phase. The reduction of algal biomass eventually causes the daphnia population to collapse and consequently giving the algal community an opportunity to recover. This classic predator-prey cycle could oscillate sustainably (Pratt, 1943). Planktivorous fish can, however, obscure this pattern by predation of the zooplankton. Especially young-of-the-year fish can prohibit daphnia population expansions after its spring peak.

These relations within the food web are also known as tropic cascades while the impact cascades down the tropic levels of the food chain. There is, however, both top-down as bottom-up control and there are various indirect relations as well. This makes the aquatic ecosystem a interdependent assembly of elements. Especially the role of submerged macrophytes is divers.

In order to avoid predation, zooplankton seek refuge among macrophytes. The macrophytes give shelter in two ways. First of all the physical structure of the stems and leafs provide cover against planktivorous fish (e.g. roach and small perch). Secondly, in a more indirect way, piscivorous fish (e.g. pike and large perch) hide between the macrophytes to ambush their prey so small fish cannot graze unrestricted.

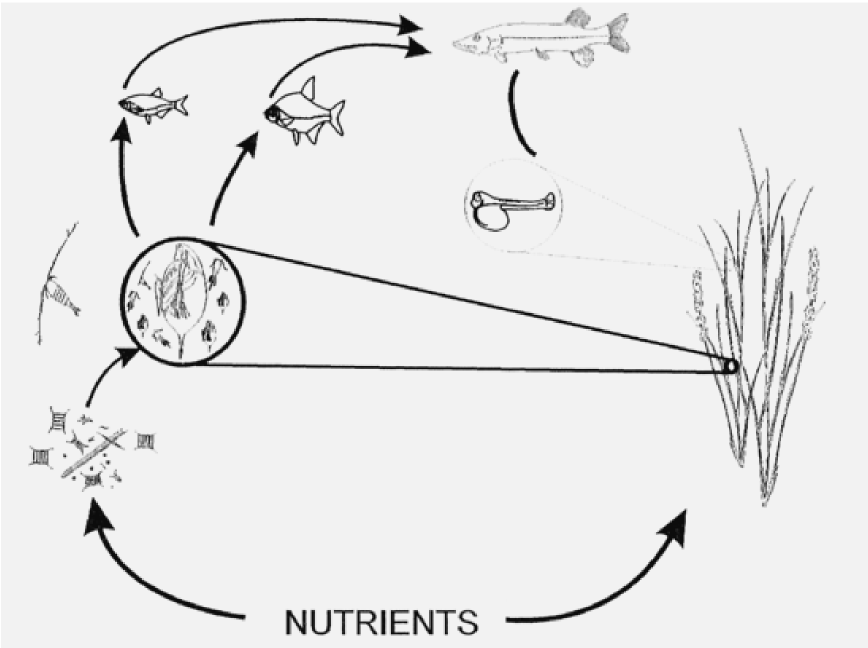

Figure 2.9 The position of important ecosystem elements in the food web (Scholten et al., 2005).

Macrophytes give shelter to juvenile piscivores too while they are very susceptible to cannibalism. It is very important to provide a habitat for the development of these predators because they have the potential to control the number of planktivorous fish and bring about a well balanced fish stock. The presence of vegetation is one of the main factors in structuring the fish community (Scheffer, 1998).

Macrophytes also effect sedimentation and resuspension. In the Netherlands, urban ponds often have soft sediment soils of clay and decaying organic matter. Without rooted macrophytes this sediment is very susceptible to wave resuspension. Furthermore the feeding of bream and roach on benthic organisms (e.g, chironomidae) is strongly hampered and turns them to the aquatic macroinvertebrates abundant among the plants. This leads to less perturbation of the sediment and thus to a decrease of overall turbidity.

Restoration of the clear water state thus needs a restoration of the aquatic ecosystem community with a central role for submerged macrophytes and a leading role for daphnia in algae growth control. The resilience of the ecosystem to cope with nutrient loadings is however not unrestricted and limitation of nutrient loadings are therefore a prerequisite.

Figure 2.10 Pike (Esox lucius) lying in ambush.

Chapter 3
Urban water quality management

3.1 Introduction

In the Netherlands, the management of the urban water systems is rather complex. Not only do the various water bodies often have multiple (conflicting) functions, its management is also carried out by various authorities with joint responsibilities. This has led to numerous laws and regulations on different organizational levels and makes the management of urban waters a composite matter. In the following section (§3.2), the different stakeholders and their policies are portrayed, starting at the highest management level all the way down to the local level which includes the operational management.

Water quality management in urban areas has not always been a priority to the different stakeholders and to provide some context on the current assignment of the urban water managers (§3.4), several mayor developments in water quality management are described (§ 3.3).

3.2 Stakeholders, laws and regulations

Urban water management in the Netherlands is primarily a task of municipalities and water boards. They take into account specific regional and local conditions for their strategic and operational management. Urban water systems are integrated elements of larger regional water systems or even international catchment basins, so management on national or even European level is important too.

Each water authority has its own specific obligations and responsibilities in controlling and improving the functionality of water systems. The obligations for operational management by local authorities, however, often originate from policies set by higher management levels. It is therefore very important to attune these obligations and responsibilities to one another for optimal implementation of all regulations. A wide range of other parties can be involved in policymaking as well and a participatory approach is often used to include all stakes.

In the following sections, all management authorities, their laws and regulations relevant in urban water management will be discussed.

3.2.1 European Union

European policy is one of the leading principles in water management nowadays. With the implementation of the European Water Framework Directive (Directive 2000/60/EC; **WFD**), the various former Water Directives were consolidated into a single scheme. This integral approach harmonised the water management strategies of the EU Member States, resulting in an uniform European-wide response to policy questions in the domain of ground- and surface waters.

The geographical coordination of the WFD management is based on river basins. Each river basin district requires a management authority and a river basin management plan (stroomgebiedbeheerplan; **SGBP**) including a program with measures. Of all river basins in Europe, four are partially covering Dutch territory (see Figure 3.1).

Figure 3.1 River basins in the Netherlands (retrieved May 2011 from www.kaderrichtlijn.nl).

The WFD provides a framework for the protection of inland surface waters, transitional waters, coastal waters and groundwater. The Directive (Art. 1) intends to ensure that:

- Aquatic ecosystems are preserved from further deterioration;
- The aquatic environment is improved e.g. through substantial reduction in discharges and emissions;
- Sustainable use of water is promoted;
- Groundwater pollution is reduced considerably;
- The effects of floods and drought are mitigated.

The WFD (Art. 4) has set environmental objectives for surface waters, for groundwater and for protected areas. The objectives for surface waters come down to preventing any decrease in the status of all surface waters, achieving good surface water status by 2015 and progressively reducing or phasing out pollutions.

‘Surface water status’ is defined (WFD, Art. 2) as ‘the general expression of the status of a body of surface water, determined by the poorer of its ecological status and its chemical status’. ‘Good surface water status’ is the status achieved by a surface water body when both its ecological status and its chemical status are at least ‘good’.

The chemical status is determined by substances included in the list of priority (hazardous) substances (Decision No. 2455/2001/EC). Directive 2008/105/EC sets environmental quality standards for these substances. If a surface water complies with these standards, the chemical status is assessed ‘good’.

The ecological status of surface water bodies is defined in terms of biological quality elements (phytoplankton, fish fauna, etc.), hydromorphological quality elements (hydrological regime, riparian zone structure, etc.) and physico-chemical quality elements (nutrients, oxygen levels, etc.).

‘Good ecological status’ is achieved if the biological quality elements of a surface water body deviate only slightly from those normally associated with that particular surface water body type under undisturbed conditions. Hydromorphological and physico-chemical quality elements provide as boundary conditions for this biological quality achievement.

The WFD designates surface water bodies as either ‘natural’, ‘heavily modified’ or ‘artificial’. It defines an ‘artificial water body’ as ‘a body of surface water created by human activity’ and a ‘heavily modified water body’ as ‘a body of surface water which as a result of physical alterations by human activity is substantially changed in character’. To designate a water body as artificial or heavily modified additional requirements have to be met. The effects of the hydromorphological alterations needed to achieve a good ecological status must be significantly negative for other functions. Furthermore, no alternative measures are possible to create a better situation from an environmental point of view or their costs are disproportionately high.

The designation as natural, artificial or heavily modified is important for the assessment, because the artificial and heavily modified water body will be subject to an adjusted ecological objective. Natural surface waters have the objective of a good ecological status (GES), whereas artificial and heavily modified waters are required to reach good ecological potential (GEP). The GEP is thereby an ecological objective that takes account of hydromorphological alterations that have led to the designation of the water body as artificial or heavily modified.

Another important step in the implementation of the WFD is the identification of surface water bodies. The WFD defines a surface water body in article 2 as a ‘distinct, sizeable body of surface water’. The phrase ‘sizeable’ means, in the case of stagnant waters such as lakes and reservoirs, a minimum size of half a square kilometre (50 ha) (see WFD Annex II, §1.2.2).

This identification of surface water bodies has turned out to be essential in the implementation, as the designated water bodies are explicitly subject to the specific ecological objectives mentioned before and the argumentation to designated these water bodies as artificial or heavily modified has to be included in the SGBP’s. The objectives for the chemical status are defined by the environmental quality standards and they apply to all surface waters, just like the objectives for progressively reducing priority substances and phasing out priority hazardous substances.

Although the WFD integrated various water related directives, there are still numerous other directives which are relevant to (urban) water management. Examples are the urban wastewater treatment directive (Directive 91/271/EEC), the directive on maximum residue levels of pesticides (Directive 86/363/EEC) and the nitrates directive (Directive 91/676/EEC). It is however beyond the scope of this research to deal with these directives in detail.

3.2.2 National government

The management of the national waters (rijkswateren) is the responsibility of the Directorate-General for Public Works and Water Management (Rijkswaterstaat), an executive service of the Ministry of Infrastructure and the Environment. National waters only include the major rivers and canals, the North Sea, the Delta, the Wadden Sea and lake IJssel.

Although the role of the national government for the operational management is limited to these specific water bodies, its strategic management, set with laws and regulations, determines the vision of the Dutch water management and lays down the guidelines for the management on lower governmental levels.

The Dutch national water policy is drawn up in the National Water Plan (Nationaal Waterplan; **NWP**) based on the Water Act (**Waterwet**). The NWP for the period of 2009-2015 (V&W, 2009) succeeds the Fourth Policy Document on Water Management (Vierde Nota Waterhuishouding; **NWH4**) which focused on integrated water management with a water system approach and stimulated interest for water management in urban areas. The NW4 also set water quality standards (Maximaal Toelaatbaar Risico; **MTR**) and target values (**streefwaarden**) which the NWP adopted.

Furthermore, the NWP builds on the National Water Management Agreement of 2003 and 2008 (Nationaal Bestuursakkoord Water(-actueel); **NWB**) which emphasizes the joined responsibility of all water authorities to achieve a good functioning of the national water system by 2015 and indicates how they should deal with the increasing threats and challenges in water management in the 21st century (e.g. climate change, land subsidence). This policy (Waterbeheer in de 21st eeuw; **WB21**) created a paradigm shift in water management; from fighting against water towards living with water (see also § 3.3).

The NWP continues with these strategies and increases the adaptive approach to deal with climate change. Additionally it focuses on integrating water management with other policies (e.g. spatial or economical) to create the best output with minimal costs. The NWP also draws up a specific target vision (streefbeeld; NWP § 5.9) for the water systems of urban areas. Its aims at creating a sustainable, robust, healthy system with large natural purification capacity contributing to the quality of living in the city.

The water quality objectives of the WFD are put into Dutch practice by an order in council (Besluit kwaliteitseisen monitoring water 2009; **Bkmw).** The standards for the chemical and ecological status of water systems are thereby laid down on the basis of Chapter 5 of the Environmental Management Act (**Wet Milieubeheer**). The Dutch government decided to included the four SGBP's as addenda to the NWP. The collateral regional measures programmes are only summarized in the SGBP's and not included as separate documents. The detailed descriptions of these measures are laid down in the water management plans of the regional management authorities.

As mentioned before, the ecological water quality standards only apply to water bodies as defined by the WFD. Most of the surface waters in urban area do not comply to this definition (predominantly because they are much smaller). Although the legal obligation might not affect these waters, they are still relevant to them. First of all the quality of these water bodies can influence the status of the WFD water bodies so that the responsible water manager can implement measures based on the WFD in their water plan. Secondly the NWP states that the standards established in the Bkmw should be used as reference when making assessments for the non WFD waters.

3.2.3 Regional authorities

The elements of the water system which do not belong to the national waters are called regional waters (regionale wateren). And although the policies of the national government act as the main guideline, most of the strategic and operational management is conducted by the regional water authorities; the provinces and, above all, the water boards.

Provinces

The 12 Dutch provinces have an significant role in the strategic management of the regional water systems. They set up provincial water plans which apply the NWP and the WFD transforming them into a regional approach. Additionally, they set up the spatial planning vision of the province in structure plans and assemble provincial environmental plans. These plans are closely related to strategic water management and often include it as an integral part.

Another important competence of the provinces is the regulation of the structure and obligations of the water boards. They can for example, in order to create coordinated and efficient regional water management, prescribe the preparation, adoption, amendment and content of water management plans by the water boards (Water Act Art. 3.11). The provinces even have the authority to establish and abolish (merge) entire water boards but the current arrangement of 25 water boards (see Figure 3.2) is already slimmed down a lot from over 2500 mid-20th century so that competence is hardly used any more.

Figure 3.2 The 25 water boards (Rioned, 2009a).

Water boards

In the Dutch water management practice, water boards have an essential role. They are obliged to take care of the management of the regional water system, most of the flood defence structures and the treatment of domestic wastewater.

Since 1992, with the introduction of the water board act (Waterschapswet), water boards are officially responsible for the strategic and operational water management of the primary surface water system in urban areas. The municipalities are responsible for the secundary surface water system used for the drainage of stormwater runoff of public area (ontwatering van de openbare ruimte) and the groundwater management. The following water bodies are part of the primary surface water system:

- Water bodies used for discharging surplus drainage water
- Water bodies used for water supply and flushing
- Water bodies functioning as storage buffers
- Water bodies which receive combined sewer overflows

The differentiation between water bodies used for discharge or used for drainage is made based on a reference discharge: the discharge occurring during a rain event with a certain probability (e.g. once a year). In practice this means that the differentiation can be made by the total surface area which runs off towards a water body.

In order to lay down their vision on the management of water systems in their district, water boards draw up a water management plan (**waterbeheerplan**). This document sets the strategic and operational management objectives and measures for a period of six years (in line with the WFD planning scheme). It describes the strategy to deal with the most recent developments and challenges (e.g. climate change, modifications of legislation) and establishes the financial foundation of it all. In the end the water management plans of the water boards give a local interpretation of the WFD, the NWP and the provincial water management plans.

In addition to their water management plans, water boards often initiate a specific memorandum with their vision on urban water management (e.g. Groot Salland, 2007; Hunze en Aa, 2003; Roer en Overmaas, 2007).

Intermezzo: Transition of responsibilities

In urban areas the management of the water system is a shared responsibility of water boards and municipalities. Due to historical development, the water boards focused on the rural area and municipalities took care of the management in urban areas. With the legal designation of water boards as responsible managers of urban surface water systems, the transition of the management of the accompanying banks and sediment beds turned out to be complex.

The banks are often part of the public space which is owned by the municipality and thus the responsible operational manager. Water boards are not purchasing all land surrounding the water system so arrangements had to be made on the division of the operational management.

The main problem was however the overdue maintenance on the sediment beds. In 2004 the association of regional water authorities (Unie van Waterschappen; **UvW**) and the association of Dutch municipalities (Vereniging van Nederlandse gemeenten; **VNG**) made a policy document (Uitgangspuntennotitie waterbodem in bebouwd gebied) which stated that the costs of the required dredging operations needed to be split 50/50 after reduction of the contribution by the national government. The transition of responsibilities is still an ongoing process as a consequence of the complexity of the responsibility and maintenance issues (e.g. Rijnland, 2006; Groot Salland, 2010; Verdellen et al. 2010).

Figure 3.3 Surplus sediment removal.

3.2.4 Municipalities

Municipalities are the local management authorities in the Dutch governmental system. Currently there are a little over 400 municipalities (mid-20th century there were still 1000). The legal responsibility of the municipalities is limited to the collection and transport of wastewater, the drainage of stormwater runoff (ontwatering) and groundwater management but they are often the operational managers of the morphological features of the urban surface water systems as well. Water boards outsource this duty while municipalities can include it in their operational management of the public space and are thereby the most cost efficient executors of these tasks.

The starting point of the municipalities urban water management is the management of the sewer systems. Since 2008 (Wet gemeentelijke watertaken), the responsibility of the municipalities extended with the management of stormwater runoff and the groundwater management. They now have to provide a management plan on how they will cope with these responsibilities (verbreed gemeentlijk rioleringsplan; **vGRP**).

Besides the extension of the legal responsibility on runoff management, there are numerous other aspect on urban water management which have to be addressed by the municipalities (e.g. spatial implementation of water quantity measures based on WB21 or environmental measures to comply to the WFD). That's why an increasing number of municipalities creates an urban water plan (**stedelijk waterplan**). These urban water plans draw up an integrated vision on the water management of a specific urban area and are created in cooperation with the other (management) stakeholders. The plans usually contain the division of responsibilities over the stakeholders, the objectives on the various issues, a set of concrete measures and the financial accountability of it all.

The municipal water plans are not obliged by the Water Act like the National or Provincial Water Plan. It is the NWB that states that the municipalities and the water boards have to disclose the urban water assignment (stedelijke wateropgave) based on the framework of WB21 and the WFD and, if they both desire it, draw up an urban water plan. Because of the lack of a legal commitment there are no official guidelines for these plans. The VNG and the UvW did lay down guidelines in an assistance document (Van der Meide et al., 2004).

3.3 Quality management development

3.3.1 Management development

In 1962 Rachel Carson published Silent Spring, a book on the (environmental) effects of uncontrolled pesticide use. Around the same time the Club of Rome was founded and publish its report Limits to Growth in 1972. These events were part of a greater movement of increasing environmental awareness. At that time the water quality of the river Rhine was so much degraded that aquatic life was limited to an absolute minimum and at even formed a human health risk. The Rhine was called the sewer of Europe.

The first framework to address the problems with surface water quality was the Pollution of Surface Waters Act of 1970 (Wet verontreiniging oppervlaktewateren; **WVO**). This act regulated the discharge of pollutants on surface waters and resulted in the construction of sewer networks and effective treatment plants to limit the pollution by domestic and industrial sewerage. A special committee to guide the implementation of the act was formed (Commissie Uitvoering WVO; **CUWVO**).

During the following decades, developments in social expectations, scientific knowledge and public welfare lead to a expansion of the approach on water management issues. This is well illustrated by the transformation in 1995 of the CUWVO into a committee on integrated water management (Commissie Integraal Waterbeheer; **CIW**). Nowadays the CIW is merged into the national water consultation (Nationaal Water Overleg; **NWO**) which, for example, counsels on the implementation of the WFD.

When zooming in to the local scale of urban water management, where policies and scientific knowledge have to be implemented by concrete measures, a mayor phase lack becomes apparent. Until recent, urban water management and especially urban water quality management was an insignificant management aspect. The underlying cause can partially be found in the division of administrative responsibilities over the governmental authorities and their respective focus of interests. As mentioned before, the water boards historically focused on the rural area and the municipalities integrated the management of the urban water system into their management of the public space. On top of that, the primary function of the urban water system is the regulation of (ground) water levels in the city to provide for optimal living conditions (i.e. no pluvial flooding or groundwater nuisance at dwellings). This consequently lead in most Dutch cities to little attention for water quality and to a situation with very monotonous aquatic ecosystems.

Fortunately in the last two decades a shift has come about. It more or less started in the early ninety's when the CUWVO initiated the beginning of the reduction of combined sewer overflows. Their recommendations were based on European policy (part of Directive 91/271/EEC) to reduce pollutant emissions to the North Sea by 50 %. In practice this launched a mayor cutback of (nutrient) loads on urban water bodies while they often functioned as extensions of the storage capacity of combined sewer systems in case of severe rain events. This management approach is known as the 'basisinspanning' and had an initial objective of 1998. Later, this deadline was extended to 2005 but only in recent years a majority of municipalities has accomplished its goals (V&W, 2008).

Figure 3.4 Combined sewer overflow (left) and construction of a CSO settling tank.

The transition of the operational management responsibility (see intermezzo §3.2.3) turned out to be another stimulant for urban water quality enhancement. This transition of responsibilities from the municipalities towards the water boards initiated a lot of dredging of (polluted) sediments beds (overdue maintenance). The national government contributed significantly to this by providing subsidies (Regeling eenmalige uitkering baggerwerkzaamheden bebouwd gebied; **SUBBIED**).

With the proclamation of the NWH4 in 1998, urban water management was, for the first time, mentioned explicitly as an important aspect of regional water management. It stressed that the water boards and municipalities had to develop an integrated and comprehensive vision on their urban water management. It also stated that the potential ecological value of the urban water system was underestimated and at that time not utilized. The usage of sustainable building materials, the infiltration of stormwater and the ecological design of riparian zones were other important management issues which were addressed.

Some serious threats of river flooding mid-nineties created the (public) demand for a thorough assessment on the status of the Dutch water system and its capability to cope with the challenges of the 21st century. The committee on water management in the 21st century (**Commissie wb21**) came up with a strategy to deal with these challenges and asked for an anticipatory approach which does not shifts its problems to others/other places using the following principles:

- water as leading principle in spatial planning
- provide room for water
- retaining, storing and draining (on dealing with stormwater)
- keeping it clean, separated and treat it (on quality management)

These principles, later established in the NBW, created a paradigm shift in water management: from fighting against water towards living with water. The Dutch government also launched a public communication campaign in 2003 to stress this; Nederland leeft met water (see Figure 3.5).

Figure 3.5 Advertorial to raise public interest in water management (retrieved June 2011 from www.nederlandleeftmetwater.nl).

By now, at the beginning of the second decade of the 21st century, urban water quality management is a hot topic. Concrete measures are being implemented and the reduction of point sources like combined sewer overflows decreased loadings significantly. The WFD urges the water managers to achieve its objectives by 2015. The NWP has dedicated a section to urban areas which underlines the value of a robust, health water system that functions as an amenity for residents .

The next steps in urban water quality enhancement are however increasingly challenging. The NWP already states that water quality is no longer the limiting factor for ecosystem development due to the progress in reduction of pollutant loads. This results in an approach focused on measures like redesigning the riparian zone or stimulation of water level fluctuations. The effects of these kind of measures are however not as straightforward as the reduction of pollutants by eliminating point sources. So a more local, integrated management approach is needed to enhance the aquatic ecosystems in urban areas.

3.3.2 Water quality assessment development

Water quality management is about the anthropogenic influences on abiotic and biotic elements which interact and form an interdependent ecosystem. To determine the effects of these influences, the state of a water body have to be determined with a water quality assessment.

The first forms of water quality assessments focused on the classification of the impact of organic pollution in running waters. The relations between aquatic organisms and these loads of organic matter was established early 20th century (Kolkwitz et al., 1902) and led to the development of the so called saprobity system. This system was increasingly used during the sixties and seventies and used indicator organisms to represent the state of a surface water body. It is based on the assumption that specific organisms, the **indicators**, could act as 'pars pro toto' to reveal the overall water quality conditions (STOWA, 2006).

Early scientific research on the influence of nutrients on ecosystem communities focused on natural situations and did not include the effects pollution (e.g. Thienemann, 1912). Furthermore it was primarily conducted at standing waters and as a consequence examined predominantly primary producers like algae (STOWA, 2006).

After the introduction of the WVO, sewage treatment plants decreased the discharge of organic matter rapidly during the seventies. Consequently, the interest of the public and the water managers gradually shifted from organic pollution to eutrophication (Klapwijk et al. 1994). During the eighties this gradually lead to the introduction of biological water quality assessment methods in the Netherlands (Tolkamp & Gardeniers, 1988). The national policy documents (i.e. Indicatief Meerjarenprogramma Water; IMP Water) started to set ecological objectives distinguishing three levels: basic, intermediate and natural. They also initiated a differentiation of the objectives for various water types. There was however still a lack of references to assess the status of the water bodies (V&W, 1990).

In 1985 the Dutch Foundation for Applied Water Research (Stichting Toegepast Waterbeheer; **STOWA**) initiated a national program to come to ecological assessment methods for Dutch inland waters. Since 1992 STOWA started to publish ecological assessment methods (Ecologische beoordelingssystemen; **Ebeo-systemen**) for the main Dutch water types:

- Running water (Stromende wateren; STOWA report no. 92-07 + 92-08)
- Ditches (Sloten; STOWA report no. 93-14 + 93-15)
- Shallow Lakes (Meren en plassen; STOWA report no. 93-16 + 93-17)
- Canals (Kanalen; STOWA report no. 94-01 + 94-02)
- Sand, gravel and clay pits (Zand-, grind- en kleigaten; STOWA report no. 94-18 + 94-19)

In 2001, with the raise in attention for urban water systems, STOWA devised a specific ecological assessment method for urban waters (STOWA report no. 01-18). Although the individual water bodies of an urban water system are typological the same as elsewhere, the urban waters have one particular aspect that makes them stand out: man as element of their ecosystem. This does not only imply that the aquatic ecosystem is influenced by anthropogenic factors but the presents of a water body also influences the appreciation of humans for their urban environment. That is why the ecological assessment method for urban waters takes into account the 'amount of appreciation' (**belevingswaarde**) as a quantity in the assessment. The aesthetics are determined by the morphology of the riparian zone, the endemic fauna (ornamental species), the total species variety (flora and fauna), the occurrence of debris, turbidity and various other factors.

The ecological assessment methods of STOWA do not only represent a manner to review the water quality, they also include the next management step by giving suggestions of measures to improve the status of a water body.

Intermezzo: The MTR standards (based on Van Liere, 2002)

The MTR standards first set by the Fourth National Water Management Policy Document are based on 'desired ecological quality', described as 100 µg chlorophyll *a* l^{-1}. This value was chosen to reach a level of water clarity allowing enough light energy to penetrate the water without limiting the growth of submerged macrophytes. A critical summer-average concentration of 0.15 mg P l^{-1} was derived from a data set of some 80 lakes, most of them dominated by green algae. In 1987 a database of some 120 lakes studied showed blue-green algae to dominate in many of these lakes. The critical summer averaged concentration of 0.07 mg P l^{-1} then found has not led to any policy change; the MTR standards remained 0.15 mg P l^{-1}. And additionally, this value was set, without any scientific argument, for other water types too. As for nitrogen, an analogous relation in shallow lakes was derived, leading to a MTR of 2.2 mg N l^{-1} (summer average).

The present MTR of phosphorus does not at all guarantee recovery from eutrophication in lakes dominated by blue-green algae (see also §2.4). The chance of recovery increases with decreasing concentration. Field- and model studies showed that most lakes remain clear when phosphorus concentration is 0.05 mg P l^{-1}. This became the target concentration for lakes in the fight against eutrophication. For nitrogen the target concentration became 1 mg N l^{-1}.

The introduction of the WFD and its obligations to determine the ecological status of water bodies, set objectives with GEP through MEP, implement measures and monitor the results, has led to a vast increase in the application of ecological water quality assessment methods and models (e.g. QBWat, KRW-Verkenner, ARCADIS-methode). Due to the simultaneous developments in the required components of the assessment systems (e.g. the references, yardsticks and models to predict the effects of the measures), the whole process of ecological water quality assessments for the first generation SGBP's turned out to be rather chaotic (ARCADIS, 2009). For the next generation of SGBP's (2015-2020) the developments continue and experience with these assessments will enhance them.

As mentioned before, the implementation of the WFD in urban water management lags a little behind due to the absence of the obligation to report on them. Urban water managers can now take advantage of the experience already gained with ecological assessment methods and might use it to take water quality assessment for urban waters to the next level.

3.4 Current management assignment

Based on the laws and regulations described in §3.2, the water managers have an extensive assignment ahead. Especially the ambition to reach conditions which provide an adequate chance of liveability to the ecological community depending on the aquatic environment is challenging in the urban environment.

The overall objective is to reach good ecological potential for all waters. This also means that the assessments to determine the water bodies status focus on other parameters than before. In practise this for example implies for nutrients that it is no longer about reaching certain standard concentrations but more about mass balances and critical loading, taking into account the system's capacity to deal with the loads.

Although the assessment of the ecological status can look like a comprehensive effort, picking the right indicators can turned out far more (economical) efficient than 'randomly' measure all kinds of (chemical) parameters.

When the needs of the water system are determined, the water manager have to take into account the specific boundary conditions of the urban area before appropriate measures can be selected:

- Hydraulic constrains: The prime function of the urban water system is to prevent water quantity problems so the hydraulic capacity of a water body cannot be disregarded.
- Spatial constrains: In urban areas, space is often limited by the high density of buildings and infrastructure.
- Architectural constrains: Water bodies are often integrated elements of the urban spatial plan. This can restrict the possible adjustments to their profile (e.g. canals in the old city centre).

On top of these constrains, the measures should also be integrated with the management programs of other policy fields to be efficient and implemented in the most economical way. This often leads to partnerships of the various responsible authorities and a rather practical approach in water quality management.

Chapter 4
Measures

4.1 Introduction

Water managers can establish the objectives of strategic water management plans in various ways. They have for example the ability to enforce, prohibit or control certain actions by license requirements or they can adjust their own operational water management approach. It is also possible to implement specific measures for more fundamental changes on the water system itself.

For the improvement of the (ecological) water quality of water bodies and to reduce eutrophication and its manifestations, various types of measures can be distinguished. It is important to realise that the restoration of the vegetated clear water state requires far more extensive measures than the general enhancement of water bodies which are already clear. Which consequently makes the latter an important aspect as well to efficiently prevent systems from switching to the turbid state.

Besides measures focussing on the prevention of eutrophication and its manifestations, there are numerous measures which focus on other aspects of water quality enhancement and ecosystem development. These measures often contribute indirectly to the reduction of eutrophication problems as well because they both relate to the overall ecological functioning of urban water bodies.

The following sections describe the different types of measures for the prevention of eutrophication and its manifestations and give some examples to illustrate the differences. Section 4.4 addresses the specific characteristics of floating treatment systems and its applications as system measure.

Figure 4.1 Implementing measures

4.2 Types of measures

Since excessive nutrient loading is the major reason why water bodies turn turbid and lose their appreciated vegetation-dominated clear water state, reduction of the nutrient loading is the principal approach in eutrophication management. Although a source approach is preferred while its impact will be the most sustainable, it is not always feasible. Some sources are unavoidable or so diffusive that measures to limit those are economically or practically unrealistic (e.g. groundwater seepage, bird droppings). Additionally, water managers are often not directly authorized to control the sources (e.g. agricultural use of fertilizers).

Another option is to implement measures which increase the capacity of a water body to cope with nutrient loading. The nutrient carrying capacity of a water body in a clear water state can be extensive (Moss, 1998). The extent of this ability is limited to a certain critical loading level. This level can be elevated by increasing the robustness of a water body and its aquatic ecosystem. The robustness originates from several buffer mechanisms within the system. Ecosystem relations have a central role (see §2.4) but are not easily influenced directly. The measures are therefore focussed on creating the optimal conditions for these relations to be established. With this system approach, the critical nutrient loading level at which a system restores from a turbid to a clear water state can be raised as well.

For the actual transformation to the clear water state, the implementation of a 'catastrophic shift' can be necessary. Measures which drastically alter the food web and direct influence the manifestations of eutrophication need to be considered. In order for this internal approach to have a prolonged effect on the state of the water body, the nutrient loading conditions should be within the boundaries of the critical loading levels.

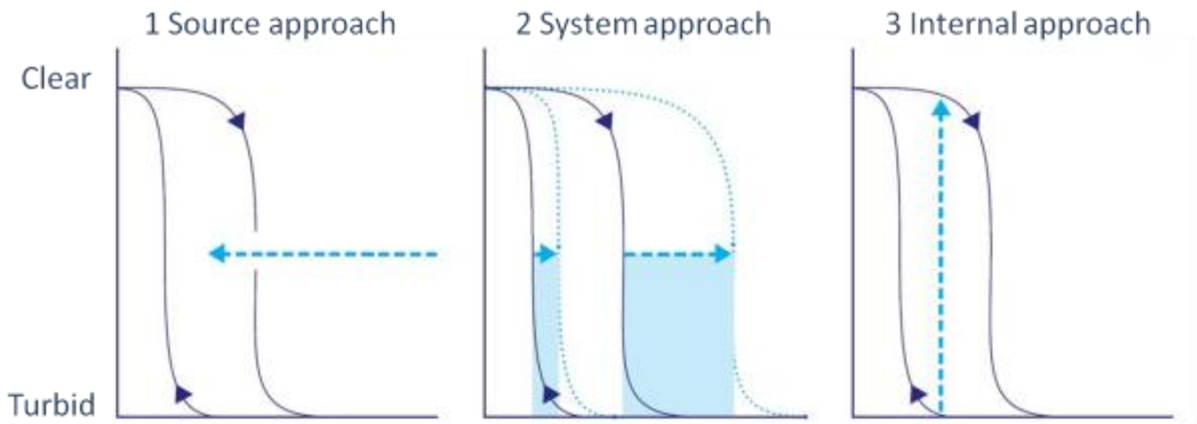

Figure 4.2 Different approaches in eutrophication management (redrawn from STOWA, 2008).

The measures to diminish eutrophication and its manifestations thus have different types of approach (see Figure 4.2). The three main types of measures are:

1. Source measures decreasing the nutrient load. Measures to reduce the nutrient loading to the carrying capacity of a water body.
2. System measures increasing the nutrient carrying capacity. Measures to enhance the robustness of the system by raising the critical levels of nutrient loading.
3. Internal measures altering the state of a water body. Measures to manipulate the food web and manifestations of eutrophication.

4.3 Various examples

4.3.1 Source measures

Measures which can be applied in urban surface water systems to reduce nutrient loading.

Water intake limitation

Water of the surrounding water system is required to minimise water level fluctuations in the urban water system. In some cases water level control is very strict and water supply and discharge alternate regularly (order of days). If the external water source (e.g. a river or a canal) contains nutrient-rich water, the allowing of water level fluctuations can reduce the amount of intake and thereby the nutrient loading. In order to implement this measure in the urban environment, the accompanying groundwater level fluctuations should be within any boundary conditions set by other functions (e.g. foundation).

Sludge removal

Dredging the upper sediment layers of nutrient-rich sludge removes the internal nutrient source. This measure is successful if nutrients accumulated due to historical loading and the sediment layers underneath the sludge are consolidated or nutrient poor. This measure can be combined with covering the sediment bottom with sand or chemicals which permanently lock the nutrients (e.g. Phoslock).

Introducing pet waste management.

Animal waste can be a significant nutrient source in urban areas. Based on a survey of MarketResponse conducted for the *Nederlandse Voedingsindustrie Gezelschapsdieren* in 2009 there are over 2 million dogs in the Netherlands. Their faeces can deteriorate stormwater runoff quality and contribute to the nutrient loading of the urban water system via stormwater sewer systems, subsurface leaching and direct overland wash off. Encouraging or enforcing residents to clean up after their pets or the assignment of special disposal sites can reduce this nutrient load. This kind of regulation as a source measure can only be implemented by the municipality which is authorized to enforce local ordinances (Algemene plaatselijke verordeningen; APV's).

Figure 4.3 Pet waste management.

4.3.2 System measures

Measures to create a more robust water systems and optimal conditions for ecosystem development.

Creation of shallow areas

Limiting the water depth enhances the conditions for submerged macrophytes growth. The penetration of light increases (relatively) which stimulates the development of macrophytes. The prevalence of macrophytes has a central role in the robustness of a water body (see §2.4) and this measure will thereby increase the critical loading level. If the development of macrophytes is hampered, shallow areas can promote resuspension of sediment by wind action.

Restoration of riparian buffer zones

Creating gradual transition zones without 'hard' artificial banks generate the conditions for riparian ecosystems to develop. The biodiversity of a water system can be increased significantly and all kinds of retention and removal mechanisms (e.g. denitrification, sedimentation, biomass accumulation, etc.) are introduced. The effectiveness of this measure is very site dependent and nutrient retention can be negative as well.

Figure 4.4 Restored riparian buffer zone (Charmantendreef, Utrecht)

Application of floating wetlands

Rafts with hydroponic growing plants can increase the habitat diversity of a water body and thereby its system robustness. Additionally the plants and the biofilm on its roots take up nutrients and trap suspended solids. The floats themselves also reduce wave action to increase sedimentation and create shading which prevents algae growth. The application of this measure will be discussed further in the next section.

4.3.3 Internal measures

Measures to directly influence the manifested state of a water body.

Fish stock management

A drastic reduction (>75%) of the fish stock generally leads to a clearing up of the water. Less sediment resuspension by benthivorous fish directly lowers turbidity and the reduction of predation pressure on zooplankton leads to top-down control of phytoplankton. Sustainable effects depend, among many other factors, on the development of macrophytes in combination with the recovery of the fish community.

Complete draw-down

The extreme water level management measure of a complete draw-down can be applied for aquatic ecosystem control (eradicating most of the plant and fish community). Additionally, in unvegetated turbid water bodies with sediment resuspension problems, prolonged draw-down could consolidate the sediment and colonization by terrestrial vegetation could prevent initial resuspension when the water body is filled u again and give macrophytes a chance to sprout.

Hydrogen peroxide adding

Adding hydrogen peroxide to a water body can eradicate blue-green algae blooms (Hazenoot, 2010). This novel internal measure targets a specific manifestation of eutrophication and sustainable effects on the state of the water body have not yet been determined.

Figure 4.5 Eradicating blue-green algae by hydrogen peroxide injection.

4.4 Floating treatment systems

An unconventional application of a system measure to increase the robustness of a water body is a floating wetland. The urban water quality assessment method of STOWA (see § 3.4) recommends this measure for water bodies with artificial banks in order to enhance its ecological and aesthetical value. The actual application of this measure in the Netherlands is limited. Only a few case studies are known and they function with varying results. Throughout the world, a number of commercially systems is available that use various techniques for the creation of floating wetland systems to improve water quality (Headley et al., 2006). Some focus on overall quality enhancement others have more specific objectives like treating glycol laden stormwater runoff at Heathrow Airport (Revitt et al., 2001).

The following section describes some of the varieties that exist. Starting with the most basic form of plants growing on rafts followed by more engineered systems with active water circulation to batches systems that operate as small autonomous water treatment plants.

4.4.1 Basic systems

The basic design of a floating treatment wetland comprises rooted, emergent plants (bog plants) growing as a floating mat on the water surface. The plants are supported by a constructed raft and rooted in a matrix or soil media, or (as in many natural floating marshes) self-supported on intertwined mats of their own buoyant roots and rhizomes, and accumulated plant litter and organic matter. The plants acquire their nutrition directly from the water column in which their roots are suspended. The extensive roots system provides a large surface area for the growth of microorganisms. This biofilm forms under shaded conditions and with low dissolved oxygen levels giving rise to specific bacteria rather than periphyton and algae.

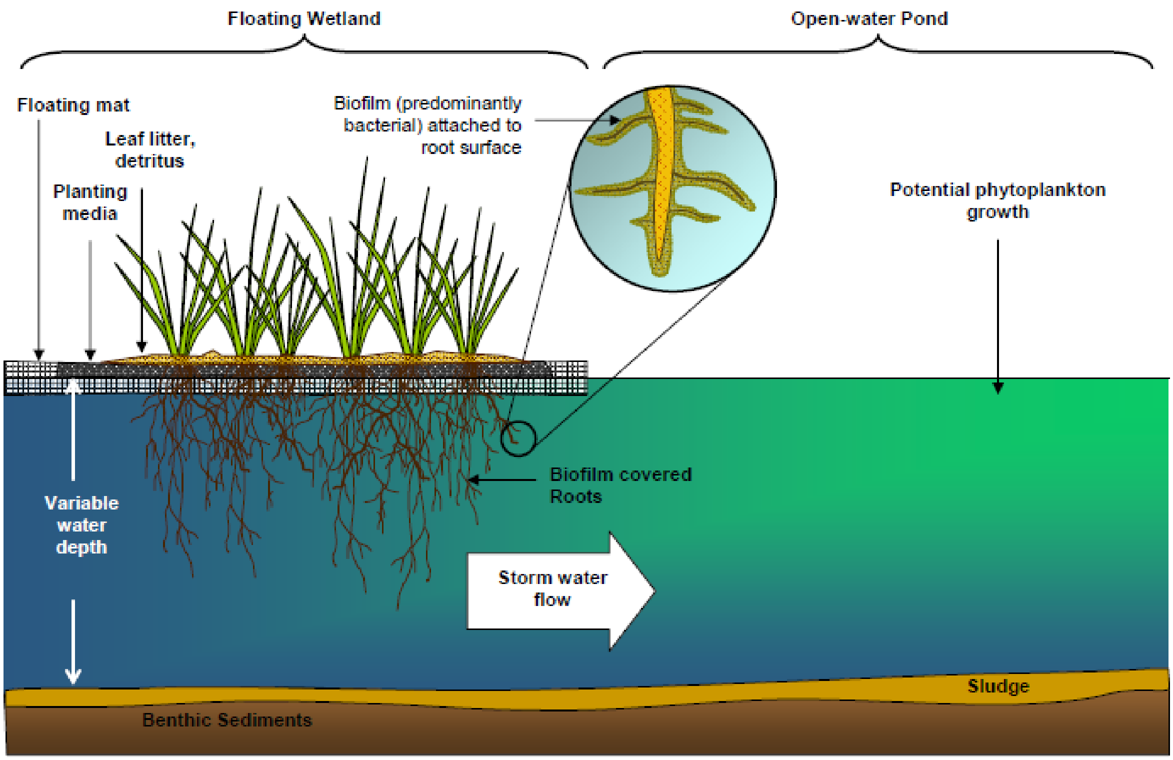

Figure 4.6 Cross-section of a basic floating treatment wetland (Headley et al., 2006).

The main water treatment processes are the biochemical processes among the root system, the entrapment of fine particulate matter and subsequent flocculation and sedimentation. Other processes are the uptake of nutrients by the plants and the promotion of sedimentation in the water column below the raft.

The application of these basic systems is predominantly in heavily loaded ponds like combined sewer overflow basins (e.g. Van Acker et al., 2005) or mine drainage ponds (e.g. Smith et al., 2000). In the Netherlands, floatlands were applied in the historic canals of Amsterdam but significant effects on water quality could not be found (AquaSense, 2000). These floatland also suffer from degradation by pollution and waterfowl.

4.4.2 Engineered systems

In order to enhance the treatment capacity of the floating systems, active water circulation can be introduced. Examples are a floating restorer of Todd Ecological Design for sewage treatment (see Figure 4.7) and the floating helophyte filter of the Bright Water Company for urban surface water quality enhancement. The next chapter will discuss the latter in detail.

Figure 4.7 Canal restorer in Fuzhou, China

4.4.3 Autonomous systems

The most comprehensive floating treatment system is comprised of several microcosms constructed in containers connected in series to provide treatment of a stream that flows through it. John Todd created a number of these Advanced Ecologically Engineered Systems (or 'Living Machines') for various water treatment applications (see Figure 4.8).

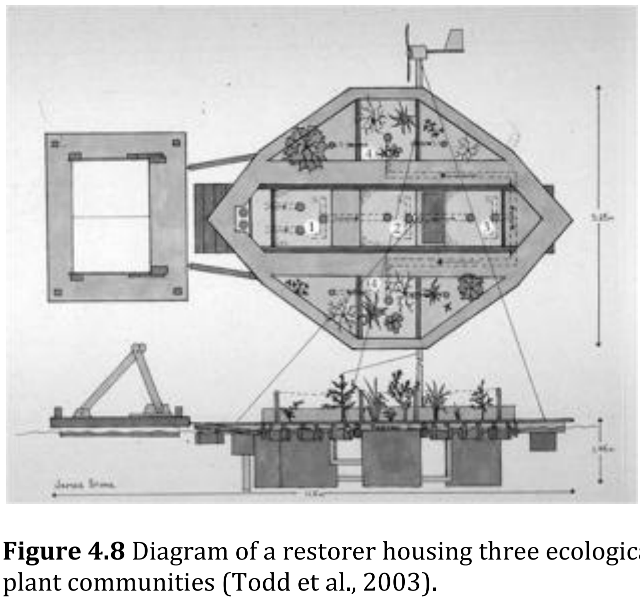

Figure 4.8 Diagram of a restorer housing three ecologically fluidized beds and six cells containing wetland plant communities (Todd et al., 2003).

Chapter 5
Case study

5.1 Introduction

The increasing interest in the qualitative aspects of sustainable urban water systems has led to a need for innovative and unconventional measures on stormwater runoff management, diffusive source control and aquatic ecosystem enhancement. The application of floating treatment systems might contribute to the approach of urban surface water quality enhancement which increases the system's capacity to deal with nutrients and limits the manifestations of eutrophication.

The Bright Water Company developed a modular floating helophyte filter to purify fresh surface water and to increase habitat diversity of semi-confined urban surface water bodies. This chapter describes the design and the main treatment functions of this filter. It examines the filter capacity through in situ measurements conducted in Groningen and finishes with the insights gained during this case study.

5.2 Description of the application

5.2.1 Filter principle

The concept of the modular floating helophyte filter of the Bright Water Company is described in detail in European patent 2028160A2 (Siers, 2009). See Figure 5.1 for an impression of the biofilter. This floating treatment system is a combination of a vertical infiltration wetland treatment system and a regular filtration system with active drainage. Its main purpose is to purify surface water of urban surface water bodies in order to improve overall water quality conditions and to prevent the occurrence of harmful algae blooms.

The system has a multiple approach for achieving its goals. It removes nutrients from the water to reduce concentrations in the water body. It filters the water in order to remove suspended solids and improve transparency. And it can contribute to the habitat diversity of a water body in order to create a more robust aquatic ecosystem. Furthermore the biofilter can contribute in raising awareness among residents about water quality conditions and the efforts to improve them. This aspect can also help to improve water quality (e.g. decreasing the amount of food (bread) being provided by residents to water birds) but is beyond the scope of this research.

Figure 5.1 The floating helophyte filter of the Bright Water Company.

5.2.2 Principle components

The biofilter consist of several components (see Figure 5.2);

- a polyester **housing** (55) forming the containment of the filter bed
- six separate **air chambers** (4) in the housing surrounding the inner reservoir for buoyancy (including water tight air valves)
- the inner reservoir is filled with multiple layers of **substrate** (20-22)
- **bog plants** (60) growing in the substrate
- two water **inlets** (50, 51) in the housing
- a **drainage system** consisting of a drainage tube (40), a pump house (80) with a drainage pump (81) and an effluent hose (82)
- an **electrical circuit** (70) for the power supply
- four **mooring rings** at the corners of the housing

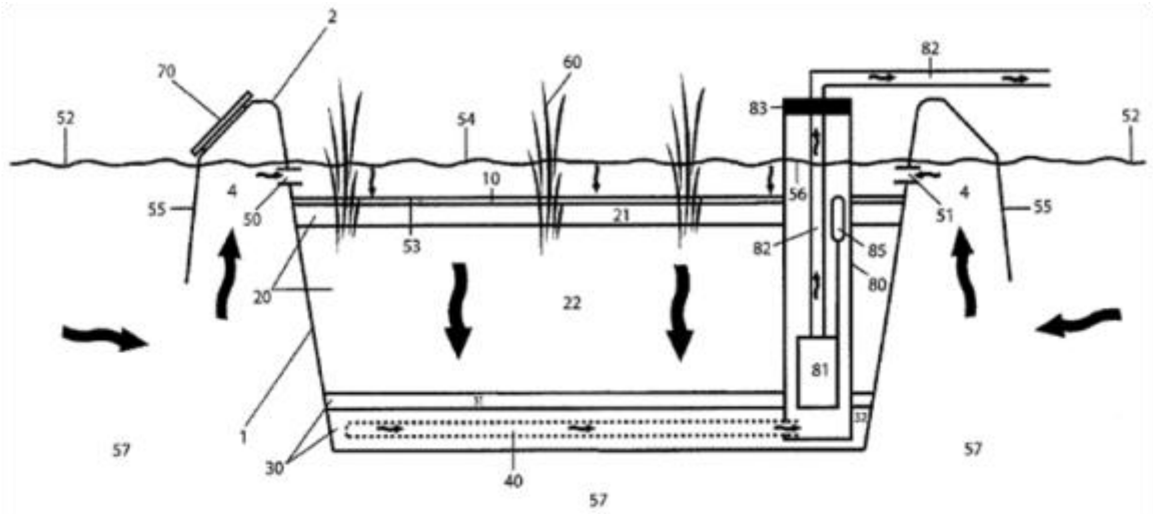

Figure 5.2 Schematic model of the biofilter (Siers, 2009).

The design of the biofilter is in such a way that modifications are quite easy to introduce. The only more or less constant is the housing which uses a mould to be fabricated.

The biofilter examined in this research was planted with common reed (Phragmites Australis) but others bog plants are suitable as well. The filter bed surface can even be constructed with a gradient to create more divers conditions for these plants to grow. Consequently the substrate thickness can vary and the composition of the substrate layers is variable too. There composition can be adjusted to specific circumstances or objectives. The current filter bed composition is discussed further on in this chapter.

The drainage system is modeled at the expected discharge and the current design uses a small, adjustable, 80 W drainage pump with a maximum hydraulic head raise of 3,1 m.

An optional future is a venturi tube at the effluent hose. This apparatus helps to increase oxygen levels in the effluent but at the same time increase the resistance of the drainage system.

The power to operate can be extracted by a solar panel (if constant flows are desired, a battery can be included as well). But often a direct power supply from the mains is available (via a fountain if present) and preferred. To manage the power system and provide safety during maintenance a water tight box with a breaker panel is mounted at the inside of the housing. In this way the biofilter has its own electrical circuit.

Finally the effluent can be discharge at a specific location by means of a effluent discharge float attached to the effluent hose. When positioned near the embankment of the water body, the effluent can be easy monitored and no boat is needed.

5.2.3 Filter bed composition

The volume of the inner reservoir of the biofilter is approximately 3 m3 including the pump house. Most of it is filled with mineral wool which has a fine fiber mesh and a very porous structure (porosity (n) is 90%). Other substrate layers consist of specific nutrient adsorbents. The current application uses 40 kg JBL PhosExPond filter® grains (a commercial product from JBL GmbH & Co. KG) with a diameter of 5 mm consisting of 40% iron for phosphorous fixation. And 550 kg zeolite gravel (a natural mineral) with a 5 mm cross-section is used for nitrogen fixation. See Figure 5.3 for the exact dimensions and composition of the various layers in the inner reservoir including the water layer on top of the filter bed.

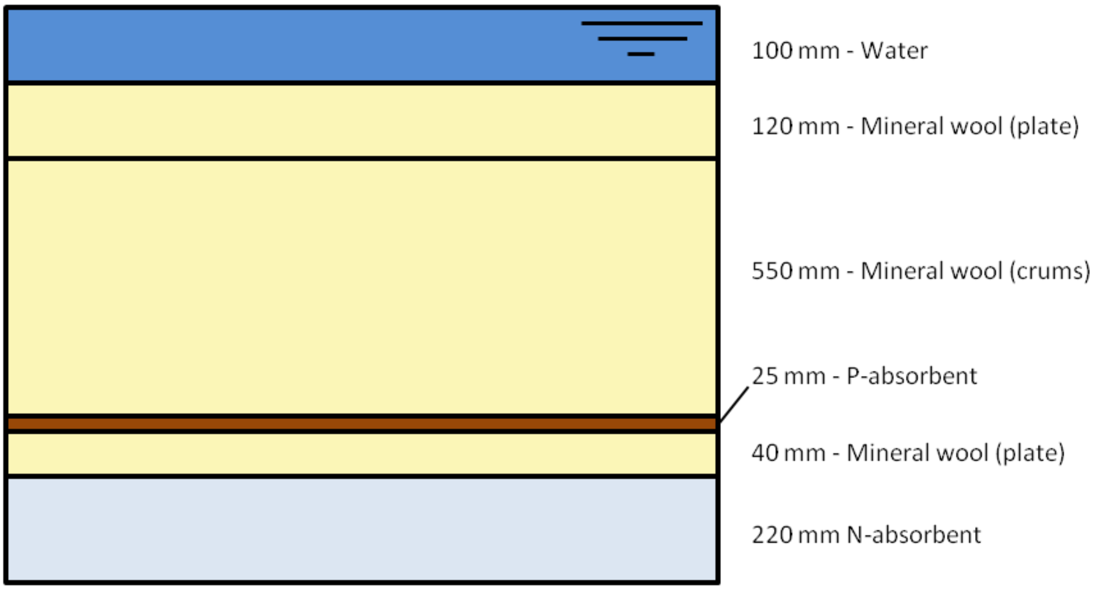

Figure 5.3 Composition of the filter bed.

The calculation of the total volume of water (V_{wtot}) in the biofilter is based on the volume of water in the different substrate layers (resulting from their porosities) and of the layer of water on top it (see Table 5-1). In this calculation the surface area (A) of each layer is not including the area the pump house occupies. The space which the bog plants and their roots take in at the upper layers is incorporated in the value for the porosity of these layers.

Table 5-1 Water content of the filter bed.

surface (m2)	depth (mm)	substrate	porosity (-)	water content (m3)
3,31	100	water	0,9	0,30
3,31	120	mineral wool (plate)	0,8	0,32
3,31	550	mineral wool (crums)	0,8	1,46
3,31	25	P-binder	0,35	0,03
3,31	40	mineral wool (plate)	0,9	0,12
3,31	220	N-binder	0,55	0,40
		Total		**2,62**

5.2.4 Hydraulics

After placing the biofilter in a surface water body, the air chambers will be filled with water until the filter is horizontally aligned and the two 150 mm inlets are partially below surface water level. In this way the influent is naturally provided from the surrounding surface water body. Floating debris like leaves, branches or plastic bottles are prevented from entering while the inlets are located behind the outer wall of the housing only allowing water from deeper layers to enter (see Figure 5.2).

The hydraulics of the design are similar to a wetland treatment system with vertical subsurface flow or a slow sand filter. The hydraulic head difference between the top and the bottom of the substrate is created by the drainage system and forces the water to flow vertically through the filter (see Figure 5.4).

With the assumption of uniform vertical flow (i.e. no preferential flow paths), the relation between head difference (ΔH), filter bed depth (ΔL), hydraulic conductivity of the filter bed(K) and discharge (Q) is determined by Darcy's law; $\frac{Q}{A} = K\frac{\Delta H}{\Delta L}$ including the horizontal surface area of the filter bed (A).

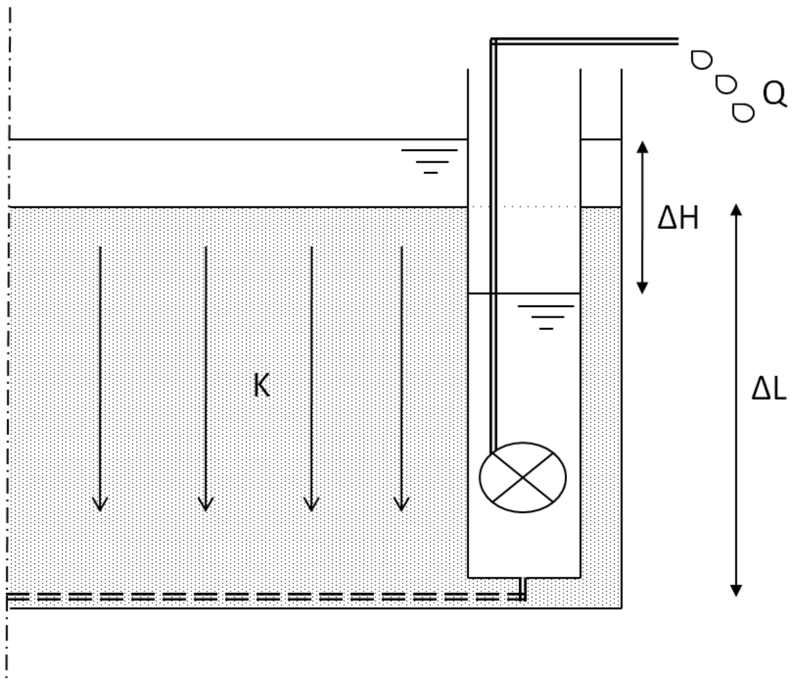

Figure 5.4 Schematization of the hydraulics.

An important condition for the calculation of the hydraulic capacity of the biofilter is the open pump house. It makes the pump unable to create suction in the drainage tube. So the maximum amount of head difference (ΔH_{max}) is the filter bed depth plus the depth of the water layer on top of the bed. When the maximum head difference is reached, the hydraulic capacity of the biofilter depends on the characteristics of the substrate layers (including bog plants). The capacity of the drainage system itself depends on the pump capacity, the resistance of the drainage and effluent hose and the head difference between the openings of these hoses.

The residence time (T) of the water in the filter is $T = \frac{V_w}{Q}$ where V_w is the water content and Q the filter discharge.

The residence time can be calculated for the entire filter or for specific parts (i.e. total filter bed or specific substrate layers) by choosing the corresponding water content.

5.2.5 Water quality processes

Various processes alter the quality of the water which flows through the biofilter when it operates. Three types of processes can be distinguished.

1. The physical process of **filtration** retains suspended solids from the water. Particles which are larger than the filter pores are sieved out of the water and smaller particles can be physically adsorbed to these particles and the substrate.
   During the filtration process, two phenomena can occur: top filtration and deep filtration.
   Top filtration occurs when the major part of the particles are retained at the top layer of the filter. A dense layer of residue will be formed at the surface of the filter bed. This 'crust' causes a relative effective filtration of suspended solids from the water but also results in a significant increase of the filter bed resistance.
   With deep filtration suspended solids will be retained more evenly over the depth of the filter bed. The substrate clogs less and the filter bed resistance increases less radically during operation. The penetration of particles with this form of filtration is larger and at the time the filter material is saturated, the efficiency drops considerably.
   The occurrence of the these different saturation mechanisms depends on the design of the filter bed and the characteristics of the solids load.
   Besides the porosity of the substrates and its saturation mechanism the overall efficiency of the filtration process also depends on the filtration rate and possible preferential flow paths.

2. The process of a**dsorption** binds dissolved chemical substances from the water to the substrate. The biofilter uses different layers of specific adsorbents for mineral nutrient fixation. The effectiveness of this fixation depends on contact time, chemical concentrations in the influent water and various other aspects.
   In order to contribute to the purification capacity of the biofilter, the adsorption has to be sustainable. The adsorbed substances should therefore remain attached to the adsorbent under all prevailing natural circumstances.

3. All kinds of **biological activity** take place in the filter and the biochemical processes accompanied with these activities alter the nutrient quantities of the water. Incorporation of nutrients in biomass occurs as bog plants grow in the substrate and periphyton grows on the wet surfaces of the biofilter.
   Biochemical transformations by specialized bacteria (e.g. mineralization and denitrification) take place in the aerobic and anaerobic subsurface layers of the filter. These biochemical processes can cause nutrients to be taken up, transformed, and removed from the water but under certain conditions emissions can take place as well.

5.2.6 Treatment quantities

All three types of quality processes contribute to the removal of nutrients from the water. To create more insight in the functioning of the biofilter and in the significance of each process, an estimation on the order of magnitude of these contributions is made. For the quantification of this estimation several assumptions had to be made. Some of them are based on personal experience, others on literature.

It has to be mentioned that most of the values in Table 5-2 are at the top of their range. This means that the calculation based on these values is very optimistic about the nutrient removal of these processes. The rate of biomass increase of reed for example, only occurs under optimal conditions and can be far less when conditions are not.

Table 5-2 Assumptions on nutrient removal rates.

parameter	value	dimension	based on
General			
Filter discharge	500	l/h	Personal field observation (September 2010)
Influent quality			
Suspended solids	30	mg/l	Educated guess
P-concentration	0,4	mg/l	Educated guess (well above MTR value)
N-concentration	5	mg/l	Educated guess (well above MTR value)
1:Filtration			
Filtration efficiency	75	%	Experiments in Best (Van Eeker et al. 2009)
2:Adsorption			
P-binder capacity	50	g/kg	Product information JBL GmbH & Co. KG
N-binder capacity	24	g/kg	Product information Zeolite Products Arnhem
3:Biochemical transformation			
Biomass increase (dw)			
Reed	3	kg/m2/y	Van Diepen et al. 2002, Mouissie et al. 2008, Vyzamal 2010
Periphyton	0,05	kg/m2/y	Meliefste 2008
P-content			
Reed	3	g/kg	Van Diepen et al. 2002, Mouissie et al. 2008, Vyzamal 2010
Periphyton	20	g/kg	Meliefste 2008
N-content			
Reed	20	g/kg	Van Diepen et al. 2002, Mouissie et al. 2008, Vyzamal 2010
Periphyton	80	g/kg	Meliefste 2008
Denitrification	1	g/m2/d	Stowa 2007, Stowa 2008

With these assumptions, the nutrient removal per year is calculated (see Annex A for the quantitative calculation and Table 5-3 for the results):

1. The amount of suspended solids retained in the filter is a straight forward calculation of the concentration of suspend solids in the influent multiplied by the total discharge through the biofilter and the filtration efficiency of the filter bed.

   The exact content of the remaining residue is much harder to estimate while the suspended solids constitution vary significantly with local conditions. The fraction of phosphorous and nitrogen is assumed to be 1/150 and 1/200 respectively.

2. The adsorption of nutrients to the specific substrates is calculated as the maximum potential adsorption per year. The capacity of the absorbent multiplied by the total amount of absorbent present in the biofilter gives an estimate of this potential nutrient removal from the water.

   With the current hydraulic capacity and the estimated influent concentration of phosphorous, the absorbent cannot be fully loaded in one year. With an absorption efficiency of 100%, phosphorous removal will be 1752 g at most.

3. The removal of nutrients via biomass is derived from the amount of biomass formed each year. This is the potential growth over the available surface area in the biofilter. The actual removal happens when bog plants are consequently mowed or periphyton is predated by other organisms (e.g. snails).

   The nitrogen transformation and removal by denitrification is calculated with the daily rate per m2. Multiplication by the available surface area and 365 days gives the total potential removal per year.

Table 5-3 Nutrient removal quantities

process	amount (g/y)	constituent
Phosphorous		
1:Filtration	98550	Total Suspended Solids
	657	bound P
2:Adsorption	2000	PO4
3:Biochemical		
Reed	40	PO4
Periphyton	10	PO4
Nitrogen		
1:Filtration	98550	suspended solids
	493	bound N
2:Adsorption	13200	NH4
3:Biochemical		
Reed	240	NO3/NH4
Periphyton	40	NO3/NH4
Denitrification	1460	NO3

5.2.7 The biofilter as a microcosm

The current filter design provides a constant water layer of approximately 0,1 m on top of the filter bed. This layer is in open contact with the surrounding surface water through the inlets. Small aquatic organisms like zooplankton and little fish might use these openings to enter the filter in search for food, shelter or breeding ground. The constant renewal of the water limits temperature increase during warm and sunny days and provide for a continuous supply of food sources (e.g. algae). Organisms which have the ability to move through the water, can also return to the surrounding surface water body while the velocity of the water through the inlets is very low (less than 10^{-2} m/s). In this way, the biofilter can become part of their habitat and provide additional means for their survival and reproduction.

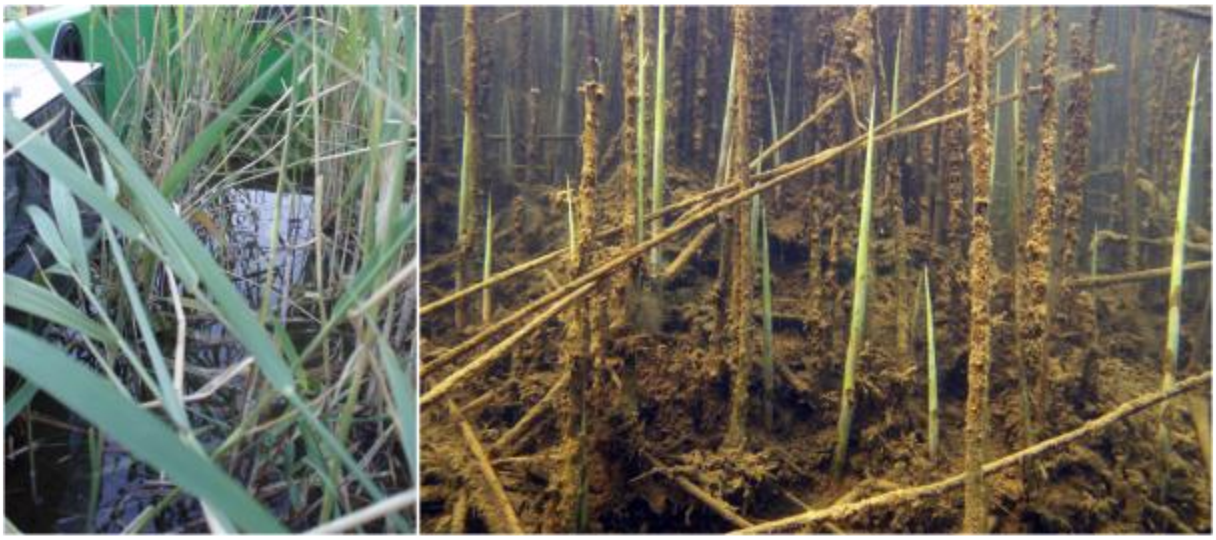

Figure 5.5 Water on top of the filter bed and an impression of the underwater microcosm (right).

5.3 Application in Groningen

In the northern part of the Netherlands, in the city of Groningen, two biofilters are applied in a surface water body called the Floresvijver (see Figure 5.6). The filters were placed on September 28th 2010 and functioned until November when winter set in. The filters have stayed in the pond but pumps were shut down when the water in the inlets froze. The filters started to function again around February 2011 and are supposed to run continuously from there on. The following sections describe the details of the conditions of the location and the role of the filters.

Figure 5.6 Location of the Floresvijver.

5.3.1 Properties of the Floresvijver

The Floresvijver is a surface water pond in a residential area north of the city centre (Korrewegwijk). It has an oval shape and is approximately 110 meters in length and 50 meters wide. The average depth is 1 meter with very little variation along its traverse (0,8 - 1,2 m). A fountain is present and increases the oxygen concentration of the water and enhances the ponds ornamental status. Furthermore, a considerable bird house floats in the middle of the pond and accommodates nesting water birds (predominantly ducks). The pond is confined by a concrete barrier all along its perimeter. The embankment itself is 15 meters wide (or more) with a small upward gradient towards the surrounding roads. A well maintained lawn covers the slopes. A couple of trees and some bushes grow on these banks as well. A sidewalk runs along the eastern side and a couple of benches are situated along this path.

Figure 5.7 Impression of the Floresvijver.

In 2010 the pond has been dredged to remove accumulated sediment and to re-establish the morphological profile. The pond is connected with the combined sewer system from which it receives overflows. During low sewer discharge, surplus water from the pond flows back into the sewer system. There is no connection to any other water body in the surrounding so the pond functions predominantly as a storage basin for the sewer system.

Overall, the pond is a blue/green element in its build-up surroundings. Residents use it to walk their dogs, to feed the ducks or to fish. The quality of the aquatic ecosystem is however very limited, with turbid water, total absence of aquatic vegetation and scarcely any habitat diversity. The quality of its supplied water is poor as well. The supply of stormwater runoff is limited and its quality is deteriorated by the banks which are heavily loaded with dog and bird faeces. Additionally, its functioning as storage capacity extension of the sewer system adds to its loading with contaminants.

5.3.2 Urban water management in Groningen

In urban areas several stakeholders are responsible for the water (quality) management of surface water bodies (see also §3.2). In Groningen the municipality joined the water boards Hunze en Aa's and Noorderzijlvest and the drinking water company Waterbedrijf Groningen to set up a partnership called Waterwerk. This cooperative body made a combined water and sewer plan for the municipality of Groningen (Gemeentelijk Water- en Rioleringsplan (GWRP)).

The city of Groningen has an extensive surface water system and its water bodies have various appearances. In order to adjust the management to its diversity, each water body is assigned an aspired function (see Figure 5.8).

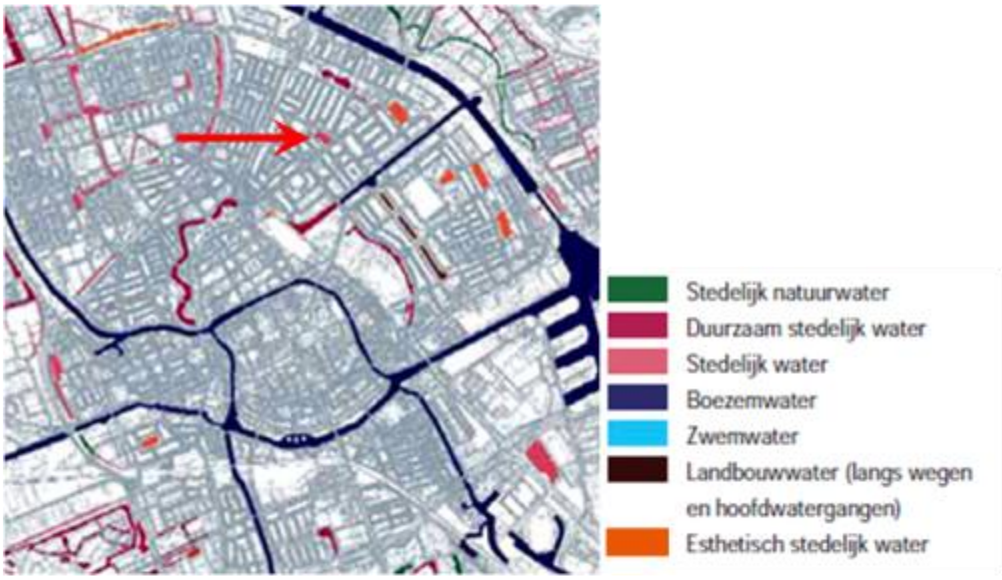

Figure 5.8 Map with aspired functions (Waterwerk, 2008) with the arrow pointing at the Floresvijver.

The GWRP describes six different water functions (in Dutch):

- Stedelijk natuurwater; water bodies which are part of the urban ecological structure
- Duurzaam stedelijk water; water bodies with a sustainable good water quality
- Stedelijk water; a left over category with water bodies primarily for water storage and drainage
- Landbouwwater; ditches along roads and some main canals, all located outside the main city
- Boezemwater; water bodies for water supply, water discharge and transport
- Zwemwater; water bodies for swimming with special attention to its quality during summer

Additionally, there is the function of 'Esthetisch stedelijk water' which covers water bodies with a specific urban appearance. This means that the water body is an integrated part of the esthetics of its surroundings.

Each function is coupled to a particular target vision (streefbeeld). This vision determines the operational management approach and includes the preferred morphological characteristics of the water bodies.

To compare the state of the water system with the target objectives (streefdoelen), an 'ecoscan' was carried out (Tauw, 2005). The results formed the basis to propose measures and set priorities to accomplish the objectives. The selected measures are now part of the GWRP 2009-2013.

The Floresvijver was not included in the ecoscan and no particular measures were included in the GWRP. The ponds function of 'stedelijk water' does not lead to any priority for water quality enhancement. Its concrete confinement is preserved due to architectural motives and clear transition between the bank and the water body is preffered. The options to create more natural riparian zones is thereby limited and alternative measures were considered. The Bright Water floating helophyte filter was selected as a measure to increase the system's robustness by introducing habitat diversity and additional nutrient carrying capacity.

To create more insight on the water quality conditions of the Floresvijver and to monitor the results of this measure, the Water board Noorderzijlvest set up an elaborate monitoring program. During three years, monthly water samples of the surface water and the influent and the effluent of both biofilters will be analysed on a large number of parameters. Right before the application of the two biofilters, the first measurements were conducted (see Table 5-4).

Table 5-4 Parameters monitored by the water board Noorderzijlvest including the first measurement results (September 23rd 2010).

BZV5	16	mg/l	NO2	0,04	mg/l
CHLFa	290	ug/l	NO3	0,1	mg/l
Cl	33	mg/l	O2	204	%
FEOa	40	ug/l	O2	19,5	mg/l
GELDHD	43	mS/m	P	0,22	mg/l
N	3,2	mg/l	pH	9	DIMSLS
NH3	0,01	mg/l	PO4	0,01	mg/l
NH4	0,1	mg/l	T	17	oC
NKj	3,1	mg/l	ZICHT	0,2	m

5.3.3 Estimated loadings

In order to determine the nutrient conditions of the Floresvijver, an estimation of the loadings is made. First of all, the relevant nutrient sources are determined based on a personal quick scan of the pond. The main sources of nutrient loadings are assumed to be:

1. Combined sewer overflow (CSO)
2. Droppings of water birds
3. Animal food surplus
4. Washed off dog faeces (by hortonian overland flow)
5. Subsurface stormwater runoff
6. Organic (leaf) litter
7. Deposition (wet and dry)

Other factors like excessive feeding during recreational fishing or upward seepage could contribute as well but are not taken into account. Due to the recent dredging, the fully loaded sediment layers are suppose to be removed.

The quantification of the contributions of these sources is predominantly based on a loadings study for five different water types including an urban surface water pond (RIONED, 2009b). The assumptions of this study were checked and adjusted for the Floresvijver when necessary. The results in Table 5-5 show that the majority of the nutrient loading is due to faeces of dogs and birds.

Table 5-5 Estimation of the nutrient loadings on the Floresvijver.

source	P(kg/y)	N(kg/y)	Based on RIONED, 2009b with assumption:
1 CSO	1,9	7,9	average value
2 Droppings of water birds	3,2	17	equivalent to 40 ducks
3 Animal food surplus	1,2	4	standard value
4 Washed off dog feces	2,7	14	25% wash off
5 Subsurface stormwater runoff	0,2	6,4	double standard value
6 Organic (leaf) litter	0,08	0,7	equivalent to 2 trees
7 Deposition	0,05	8,1	equivalent to pond surface
Total	9,33	58,1	

5.4 Measurements

5.4.1 Introduction

The turbidity of a water body is an important indicator of the state of its aquatic ecosystem and its potential development (see §2.2). Algae, suspended sediment particles, detritus and dissolved organic substances are examples of material contributing to the scattering and absorption of light in water and thereby the turbidity.

Methods to determine the solid content of water make use of a differentiation between suspended solids and dissolved solids through filtration over a filter with a specific pore size. In reality the particle size distribution of solids is a continuous range (see Figure 5.9) which makes the choice of an exact 'cut off' size rather arbitrary. The Dutch standard for determining total suspended solids content prescribes filters with a pore size of 0,45 µm (NEN 6484, 2007) while American standards use filters with pore sizes ranging from 0,2 to 2,0 µm (Eaton et al., 1995; Morquecho, 2005; Stone, 2003) underlining the subjective character of the parameter definition.

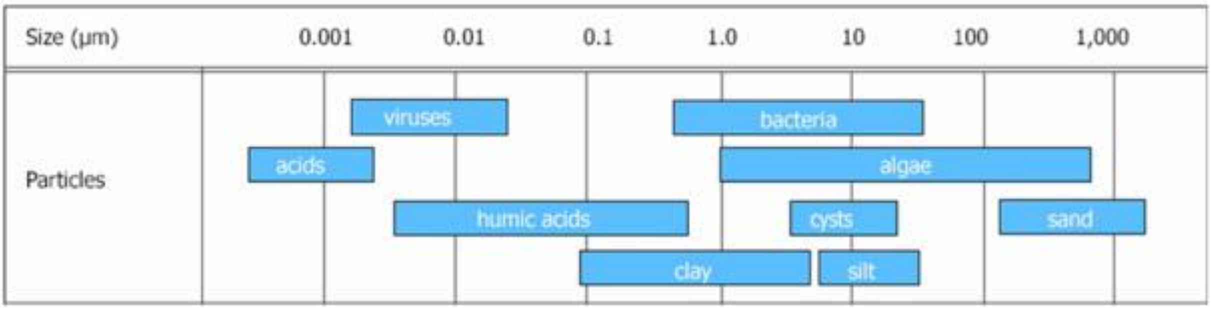

Figure 5.9 Particle size distribution (redrawn from De Moel et al., 2006).

5.4.2 Objectives

The objective of the measurements is to determine the physical filtration efficiency of the biofilter for suspended solids. In addition, the measurements combined with visual inspection can be used to gain insight on the saturation mechanism of the filtration and the functioning of the filter as a habitat.

5.4.3 Method en materials

In order to evaluate the ability of the biofilter to remove suspended solids, the total suspended solids content of the influent and of the effluent had to be determined. Therefore water samples of both influent and effluent have been taken on site and subsequently analyzed in the laboratory.

To gain insight on the saturation mechanism of the filtration process a thorough check on the state of the filter bed was needed. During a maintenance check on one of the two present biofilters, its inlets were closed and all water was drained from the inner reservoir. This gave the opportunity to check the filter bed.

Figure 5.10 Using a boat to reach the measurement locations.

Measurement scheme

To analyze the filter efficiency, the turbidity of the influent and the effluent must be compared and so the retention time of the water in the filter has to be taken into account. The retention time of the filter bed is estimated by:

$$T_{FB} = \frac{V_{wFB}}{Q}$$

V_{wFB} = 2,32 m3 (the water volume of the filter bed)

Q = 500 l/h (the total discharge through the filter)

T_{FB} = 4,6 h

So the water from the top layer entering the filter bed will be effluent approximately 4,6 hours later. If the water content of the entire inner reservoir of the biofilter (V_{wT} = 2,62 m3) is used in this calculation, the retention time is about 45 minutes longer. Because water samples are collected from the water layer on top of the filter bed (see further on), the retention time for sample comparison is assumed to be five hours.

The frequency of measuring depends on the expected variation of the suspended solids content over time. Major changes are predominantly the result of prolonged processes like stormwater running off and long-lasting substantial winds. Disturbance of the bottom sediment by fish or deposition of faeces by water birds can, in contrast, cause sudden changes. They, however, have a more local impact and the influence on the overall suspended solids content of the water body is more prolonged too. It is assumed that a measurement frequency of one hour is sufficient to observe these possible changes. This assumption is checked by increasing the measurement frequency between the measurements of 10:00h and 15:00h. The resulting measurement scheme is shown in Table 5-6.

Table 5-6 Measurement scheme.

tijd	influent	effluent
9:30	x	x
10:00	x	
10:30	x	
11:00	x	
12:00	x	
13:00	x	
14:00	x	x
14:30		x
15:00		x
15:30		x
16:00		x
17:00		x
18:00	x	x

Sampling

Water samples of the influent water are obtained from the water layer on top of the filter bed in the biofilter. The effluent is collected by simply tapping the effluent hose at the end.

The required volume of the samples depends on the expected suspended solids content of the water. To achieve sufficient amounts (> 10 mg) of residue during laboratory analysis, sample volume of influent water will be one litre. Containers of five litres are collected as effluent samples (see Figure 5.11).

After gathering, the samples are transported from the site in Groningen to the laboratory in Delft to be analyzed. All handling and preservation of the water samples is done according to the guidelines of NEN-EN-ISO 5667-3: 2004.

Figure 5.11 Effluent sampling.

Analysis

Total suspended solids content is determined in the TU Delft laboratory according to the guidelines of NEN 6484. This means that a measured volume of sample water is filtered over a pre-weighed filter with a specific pore size. The filter with residue is consequently dried and weighed. The gained weight is the measure of solids present in the sample volume. Additionally the volatile matter of the residue can be determined by burning the filter with residue and weighing the residual ashes. The weight loss is the measure of volatile matter. Water samples are analyzed in duplicate to increase the accuracy of the results.

Additional parameters

During the sampling, some basic water quality parameters are measured as well. Temperature, pH and transparency are measured with a digital thermometer, a digital pH meter and a Secchi disk respectively. The temperature and pH measurements are conducted on the surface water, the influent and the effluent. Secchi-depth is measured only for the surface water.

5.4.4 Results

On site findings

On April 5th and May 17th 2011 the measurements were conducted at the biofilters in the Floresvijver. At the beginning of April, spring just set in and the first shoots of reed appeared above the water surface of the inner reservoir. In May, after numerous days with temperatures well above normal, the plants were grown to almost a meter in height.

Figure 5.12 Six weeks of reed growth.

The discharge of the biofilters appeared to be reduced significantly compared to levels at start up (September 2010). The decrease in discharge was not caused by the pump capacity while the water level in the pump house was zero (the pump managed to discharge all water available to it). So it implies that the resistance of the filter bed caused the reduction.

Despite that the decreased discharge had a direct consequence for the residence time of the biofilter, the measurement scheme was still carried out as proposed in the previous section.

Visual inspection of the biofilter showed the formation of a sludge layer on top of the filter bed. This layer appeared to have a thickness of several millimetres to a few centimetres. The sludge itself was of very fine matter and could easily be removed with a vacuum cleaner when the filter bed was drained. The intrusion of the particles into the substrate was limited which could be determined by removal of a small portion of the upper substrate (see Figure 5.13).

Figure 5.13 Accumulation of sludge at the surface of the filter bed.

Another important finding was the accumulation of considerable amounts of gas inside the filter material. Poking the upper 10 to 20 centimetres released a pungent odour indicating the presence of methane.

The formation of the sludge layer on top of the filter bed and the development of large bubbles of gas underneath it, can be designated as the cause of the decreased discharge. This conclusion was confirmed when the discharge increased after removing most of the sludge and releasing the accumulated gas.

During the second measuring day the effluent hoses of the biofilters showed accumulation of a brownish flocculent matter which was released when disturbing the hose. Some of which was gathered and taken to the laboratory for further analysis.

The results of the basic water quality parameter measurements are listed in Table 5-7 and Table 5-8. The most remarkable findings are the very limited Secchi-depth and the relatively high pH value of the surface water (and influent) on April 5th. The pH level of the water dropped during filtration but was still more than neutral in the effluent. Meteorological data from a nearby measuring station (KNMI-station 280; Eelde) showed a considerable rain event 6 days before (see Annex B). Although an combined sewer overflow is not expected to have occurred, the pond conditions, including a high water level, could be attributed to this event.

The temperature of the (surface) water was often higher than the temperature of the air. This is probably due to the fact that the days the measurements were conducted were bleak compared to the days before.

Table 5-7 Results of the measurements of April 5th 2011.

	Temperature (˚c)				pH			Secchi depth (m)
time	influent	effluent	surf. water	air	influent	effluent	surf. water	surf. water
9:30	10,6	12,2	11,6	8,6	8,4	7,9	8,4	0,2
10:00								
10:30								
11:00			11,5					
11:30								
12:00	11,4		11,6					0,2
12:30								
13:00								
13:30		12,0	11,7	11,2	8,4	7,7	8,5	0,2
14:00								
14:30								
15:00								
15:30								
16:00	11,5	11,7	11,7	10,8				0,2

Table 5-8 Results of the measurements of May 17th 2011.

	Temperature (˚C)				pH			Secchi depth (m)
time	influent	effluent	air	surf. water	influent	effluent	surf. water	surf. water
9:30	14,8	15,0	12,7	14,7	7,2	7,0	7,3	0,2
10:00								
10:30								
11:00								
12:00								
13:00	15,1	15,0	15,5	15,2	7,1	7,0	7,2	0,2
14:00								
14:30								
15:00								
15:30								
16:00								0,2
16:30	15,6	15,0	17,1	15,4	7,2	7,0	7,1	

Laboratory analysis

Visually, the difference between influent and effluent water samples and between filter residue was clearly noticeable (see Figure 5.14). The water samples of influent were turbid and the effluent was quite clear. The residue of the influent had a dark green colour and the effluent residue was brownish. This indicates that the influent residue includes predominantly algae and the effluent might include ferric compounds. The latter could be caused by emissions of the phosphorous absorbent present in the biofilter which contains 40% iron.

Figure 5.14 Pronounced difference between influent and effluent water (left) and filter residue.

Another noticeable difference was the amount of organisms in the surface water samples compared to their presence in the influent samples. Especially the samples of May 17th showed large quantities of zooplankton and other small aquatic organisms (e.g. macroinvertebrates) in the influent (see Annex C for a impression) where surface water samples lacked most of them. Filtration of 200 ml of influent showed for example over a hundred daphnia (identified with a microscope).

The accumulated matter from the effluent hose (collected in water samples during the cleansing of the hose) turned out to be a very flocculent substance, precipitating easily when the water was left undisturbed. Examination with a microscope showed what appeared to be strands of micro flora (see Figure 5.15).

Figure 5.15 Accumulated matter from the effluent hose.

The determination of the suspended solids content of the water samples could not be carried out with the membrane filters with 0,45 µm pore size recommended by the guidelines of NEN 6484. The filters clogged before sufficient residue remained. To solve this problem for the first series of water samples (April 5th), membrane filters with a pore size of 1,2 µm were used[1]. The volume that could be filtered before clogging was 100 ml and the resulting mass of the residue was in the order of several mg. The results of the analysis are shown in Figure 5.16 including the average value of each duplicate. See also Annex D for all the exact values.

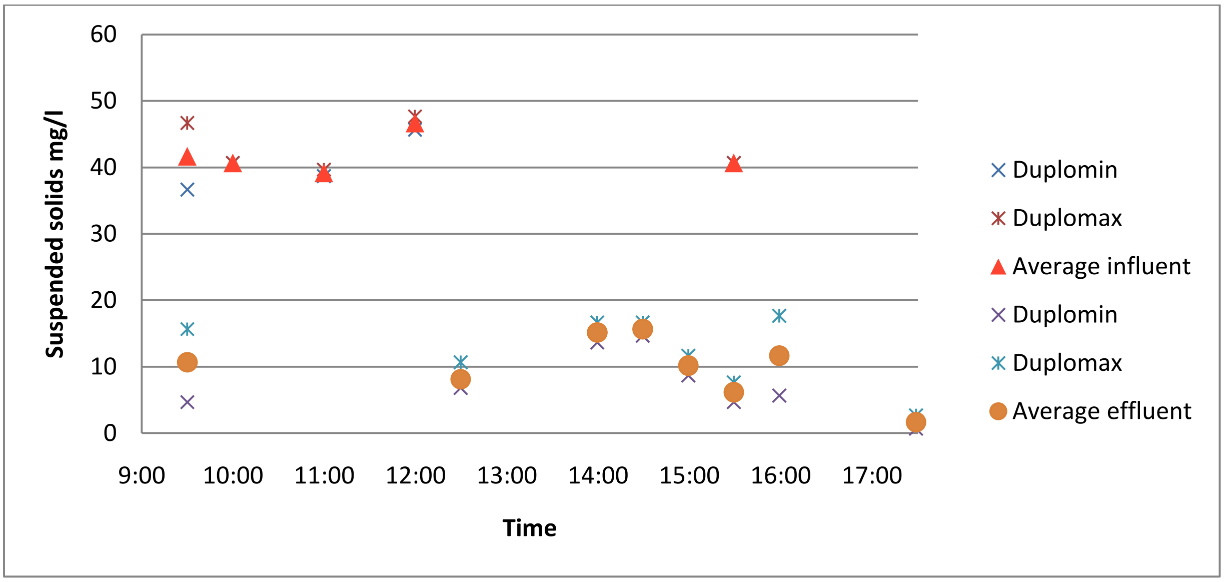

Figure 5.16 Results of the total suspended solids content analysis (April 5th)

The values of suspended solids content of the influent are quite constant with an average of 41 mg/l. The average of the effluent values is 11 mg/l but the results fluctuate considerably (2 – 16 mg/l). The standard deviation of the duplicates is 1,6 for the influent values and 2,3 for the effluent values. Especially for the effluent values this is rather a large deviation which might be due to the low absolute mass of the total residue (around 1 mg).

To determine the suspended solids content of the second series of water samples (May 17th), glass microfiber filters with a pore size of 1,2 µm were used[2]. These filters allowed more residue to be collected before clogging (6-153 mg) while still having a 98% retention of the specific particle size (product specifications). The results of the analysis are shown in Figure 5.17 including average values of the duplicates of the influent (see Annex D for all the exact values).

[1] Sartorius© gridded cellulose acetate membrane filters (pore: 1,2 µm, diameter: 47 mm)
[2] Whatman© binder free glass microfiber filters (Grade GF/C; pore: 1,2 µm, diameter: 70 mm)

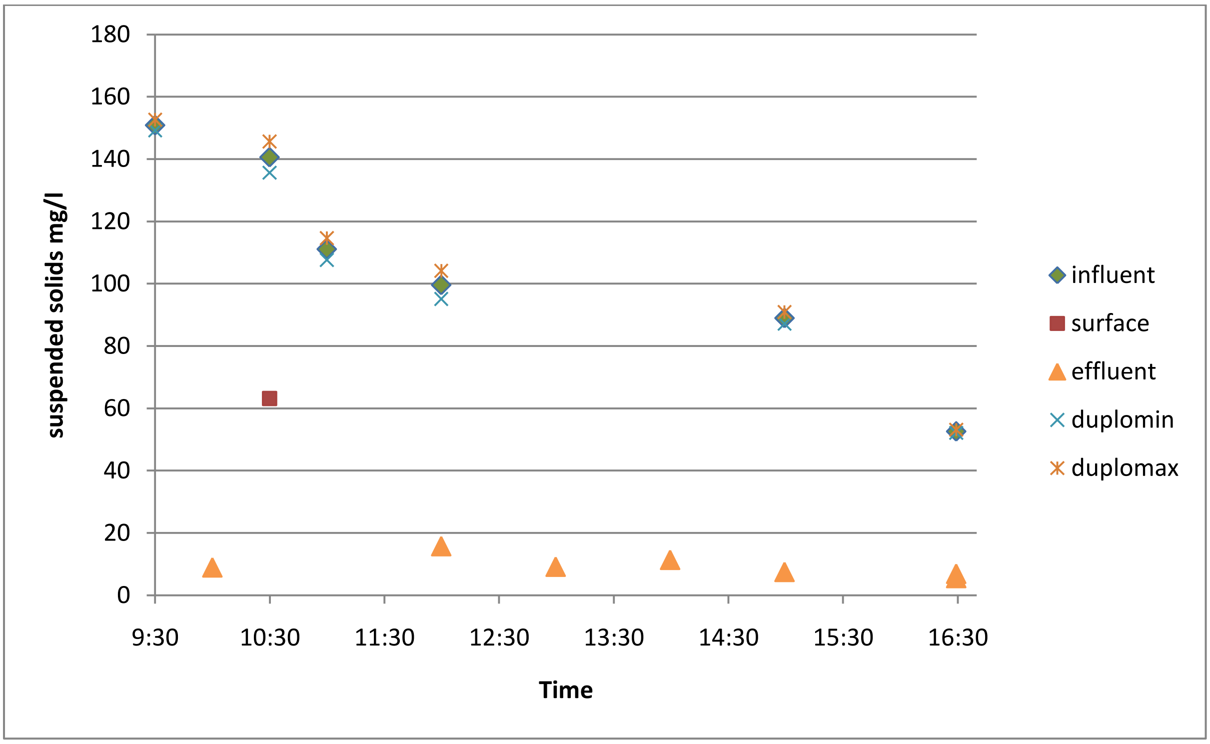

Figure 5.17 Results of the total suspended solids content analysis (May 17th)

The graph shows a clear decline of the values of suspended solids content of the influent over time. The elevated values in the morning could be due to water birds stirring up the inner reservoir content of the biofilters before the sampling started (ducks were spotted inside the biofilters at arrival on site). The afternoon values tend to be more similar with the suspended solid content of the surface water (63 mg/l).

The average value of the suspended solid content of the effluent is 10 mg/l and the individual values fluctuate moderately (5 – 15 mg/l).

The standard deviation of the duplicates of the influent values is 2,9. No standard deviation could be determined for the effluent values while those analyses were not all in duplicate.

To create more insight on the content of the suspended solids captured in the biofilters and discharged with the effluent, the volatile matter of the residue was determined. This could only be done for the second measurement series because of required residue quantities.

The results show a rather constant value of 0,40 for the volatile fraction of influent samples(see Figure 5.18). The volatile fraction of the effluent is more divers. It appears that there is a shift from values around 0,40 towards volatile fractions around 0,65.

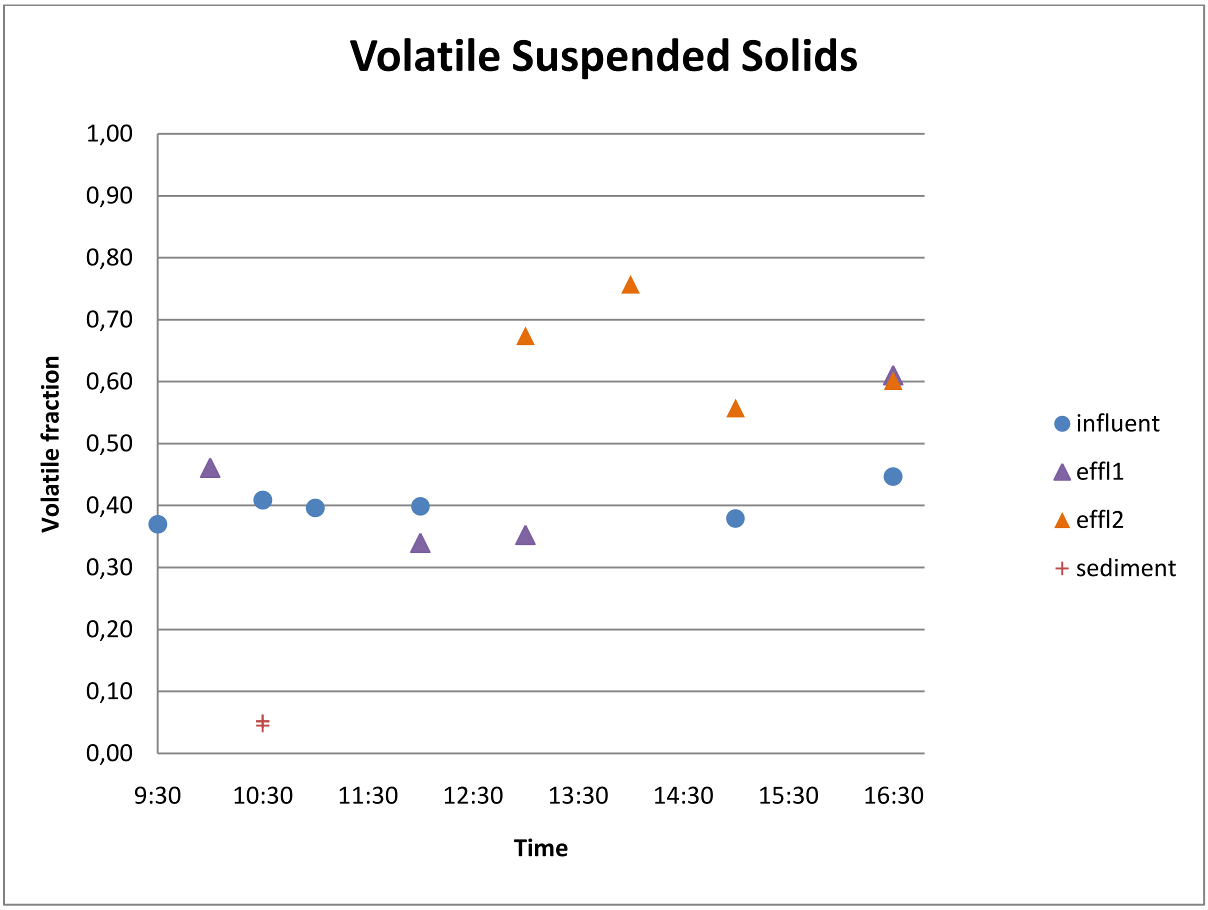

Figure 5.18 Results of the volatile suspended solids fraction analysis (May 17th)

This shift in the effluent results could be explained by the fact that during the day a transition was made from sampling one biofilter (no. 1) to the other (no. 2). The reason for this was the cleaning of the effluent hose (see also §5.4.1). In the morning the effluent hose of biofilter 2 was cleared of matter which had accumulated in it. During the day, the effluent hose of biofilter 1 also appeared to need a cleanup. A transition was made from sampling filter 1 to sampling (the cleared) number 2. A sample of effluent of filter 1 at the end of the day (16:30) showed a volatile fraction (0,61) similar to the values of filter 2 confirming this explanation.

Water board Noorderzijlvest

The results of the monthly measurements of the water board NZV are hard to interpret. The frequency and duration of the measurement campaign does not allow detailed analysis of the performance of the biofilter or the effects on the Floresvijver. With just a few data points no trends can be extrapolated. The variation between the values is very large too so even general statements are difficult to make. On top of that, there is a large difference between the results of each biofilter. As an example, the results of the NO3-measurements of the Floresvijver (NO3_Fl) and the two biofilters are shown in Figure 5.19.

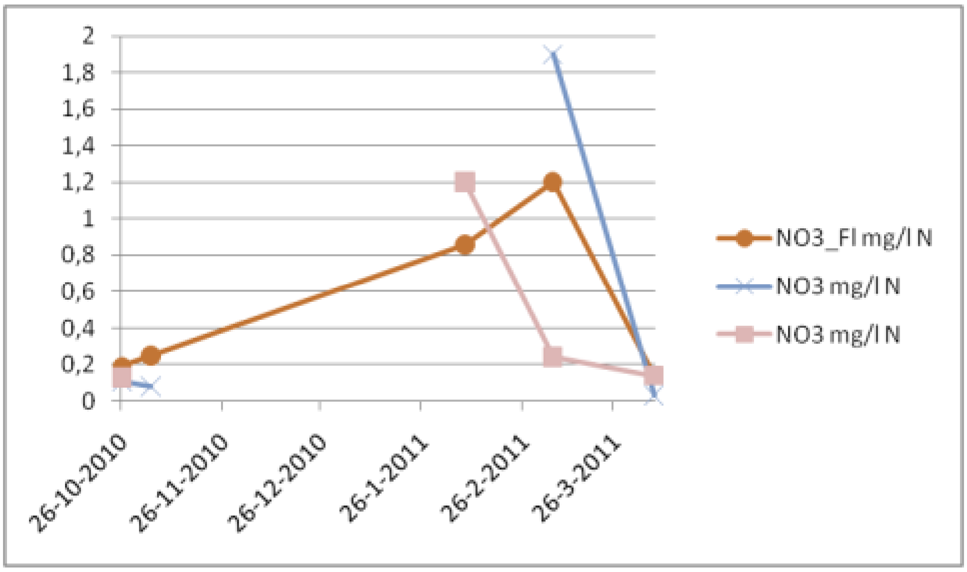

Figure 5.19 Results of the NO3 measurements by water board Noorderzijlvest.

5.4.5 Conclusions

Based on the results of the measurements a number of conclusions can be drawn:

- The biofilter lowers the suspended solids content of influent water to approximately 10 mg/l
- The physical filtration process occurs predominantly at the surface of the filter bed
- The formation of a sludge layer on top of the filter bed combined with the accumulation of gas underneath this layer decreases the discharge of the biofilter considerably.
- In the effluent hose of the biofilter accumulates (organic) matter.
- The upper water layer of the inner reservoir is a microcosm suitable for zooplankton (especially Daphnia) and other micro organisms to flourish.

5.5 Insights

5.5.1 Functioning

Filtration

The current design of the filter bed realises an effective filtration of suspended solids. This directly leads to an improvement of the transparency of the treated water. The accumulation of the solids on top of the filter bed, however, increases the hydraulic resistance and thereby reduces the hydraulic capacity of the biofilter. In order to sustain the hydraulic performance, the formation of the sludge layer should be targeted.

During the first measurement day, the accumulated solids were removed by means of a vacuum cleaner. This maintenance was rather effective but might be cumbersome in normal practise. Especially when the bog plants have developed during the growth season, the filter bed might be difficult to clean. There are several ways to improve this.

The design of the top layer of filter material can be adjusted in order to prevent the formation of a impervious layer. Introducing a gradual transition zone of course to fine filter material will vertically spread the solids, increasing the surface area to accumulate before clogging. Removal of solids can then only be accomplished in a destructive way. So a better option might be to create a vertical sawtooth pattern in the filter material. In this way, the solids can accumulate at the indentations thereby keeping the vertical edges unclogged and available for intrusion of influent. If the deeper parts of the pattern are lined with geotextile, the removal of sludge during regular maintenance at the end or the beginning of the growth season will be further assisted while no bog plants will grow in these trenches.

Another option is to connect multiple filters in series and use the first filter to remove the majority of the solids. This filter can be constructed without plants rooted in the filter bed so unhindered solids removal is possible.

Another aspect which increases the resistance is the formation and entrapment of biogas within the filter bed. Small layers of gravel could be placed between the rock wool layers to gather the gasses and ventilation tubes could release the gasses to the surface. See Figure 5.20 for an impression of the filter bed pattern and ventilation tubes.

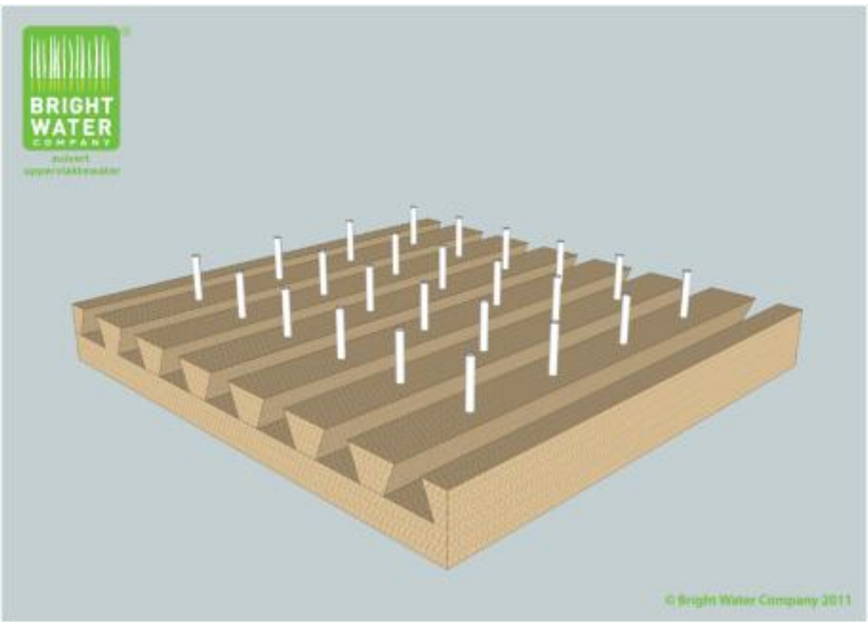

Figure 5.20 Alternative design of the filter bed surface including ventilation tubes.

Nutrients

The effective nutrient reduction of the biofilter could not be determined. Based on the calculated treatment quantities of §5.2.6, the relative importance of the different processes is shown. Adsorption by the specific substrates is the most significant (by far). The sustainability of this process is difficult to assess and depends largely on (nutrient) conditions on site. The zeolites which adsorb the nitrogen can be regenerated by flushing with a solution of sodium chloride (salt). While the zeolites are at the bottom of the biofilter, the flushing can be executed (via the drainage system) without disturbing the rest of the filter bed.

The regeneration of the phosphorous adsorbent is not possible in the current filter bed design. There is even no option to replace the absorbent without destructing the filter bed. Moreover, the effluent water samples of the measurements indicated leaching of the absorbent (see §5.4.4). The ferric components of the absorbent might dissolve during conditions of low redox potential (e.g. anaerobic conditions). These conditions are likely to occur as persistent amounts of organic matter accumulate on top of the filter bed and start decaying. The formation of biogas indicated this as well.

The current application of absorbents should thus be altered for optimal phosphorous removal and for non destructive renewal of fully loaded absorbents. A more practical application of phosphorous absorbents can be established by placing a column of absorbents in the pump house on top of the drainage pump (see Figure 5.21). The water can flow through this column before being discharged to the surrounding surface water. This set up simultaneously reduces the chance of preferential flow paths which can diminish contact time (the layer of phosphorous absorbent is only 25 mm in the current filter design). Previous aeration by means of a venturi could prevent the occurrence of reduction of ferric components of the absorbents.

Figure 5.21 View inside the pump house: Available space for an absorbent column.

Microcosm

Zooplankton and macroinvertebrates flourish inside the biofilter. A significant concentration of daphnia was observed in the water of the inner reservoir (approximately 10^3 l^{-1}). Daphnia are known to gather in dense swarms during daytime and actively spread to the open water during the night in order to forage on algae. Although there is a constant supply of food sources inside the biofilter through the influent, competition will eventually encourage horizontal migration of the zooplankton. This could be stimulated further through intermitted operation of the drainage system during night time.

The interaction with the adjacent surface water could be increased further by enlarging the inlets. Additionally, a fine grid could prevent larger (predatory) organisms from entering the biofilter.

The effective habitat could be increased by restructuring the filter bed with a gradient or simply increasing the depth of the upper water layer.

5.5.2 Application in Groningen

The direct influence of the two biofilters on the water quality of the Floresvijver was not determined with this research. In order to indicate the possible contribution to the enhancement of the water quality, the treatment capacities have to be considered relative to the conditions of the pond.

Suspended solids

The transparency of the Floresvijver is extremely low (± 0,2 m). The biofilters have the ability to filter particles from the water. Their capacity depends on the volume of filtered water and the suspended solids content of the influent. Section 5.2.6 shows an estimated solids retention of nearly 100 kg/biofilter/year. Resuspension of sediments by bream (Abramis brama) alone can be considerably more. Levels of 5 gram sediment/day/gram biomass of bream are known (Breukelaar et al., 1994). Even with a modest fish stock of 50 kg/ha (STOWA, 2008), fish could actively resuspend 125 kg sediments per day at the Floresvijver.

For an effective application of the biofilters, the filtered water should not be discharge back into the pond randomly. The filtered water can prove valuable if distributed and discharged at a specific location where its supply directly improves macrophyte development. With the monotonous underwater characteristics of the Floresvijver, such a location is currently not available.

Nutrients

The maximum nutrient removal capacity of the two biofilters is approximately 4 kg/y P and 30 kg/y N. These amounts are achieved if the adsorbents are fully loaded with nutrients in one year (which is currently not achieved due to hydraulic limitations). Compared to the estimated nutrient loadings on the Floresvijver (see Table 5-5), this is respectively 40% and 50% of the total loading amount. The effective loading on the Floresvijver can then be calculated as:

$P_{effective} = (P_{loading} - P_{filter})/(lxb)/365 \times 10^6 = 3$ mg/m2/d

$N_{effective} = (N_{loading} - N_{filter})/(lxb)/365 \times 10^6 = 14$ mg/m2/d

Considering phosphorous, this is still a major input, presumably not nearly at the critical levels causing a switch in the state of the system. An existing clear water state might be sustained with these loading levels (Janse, 2005). When looking at the relative contributions of the phosphorous load, dogs faeces and bird droppings stand out (see Figure 5.22). Targeting these diffusive sources is difficult, especially while residents have to be convinced to alter some of their customs (feeding ducks and walking their dogs along the pond). Increasing the amount of applied biofilters (e.g. doubling to four) could therefore be an favourable option.

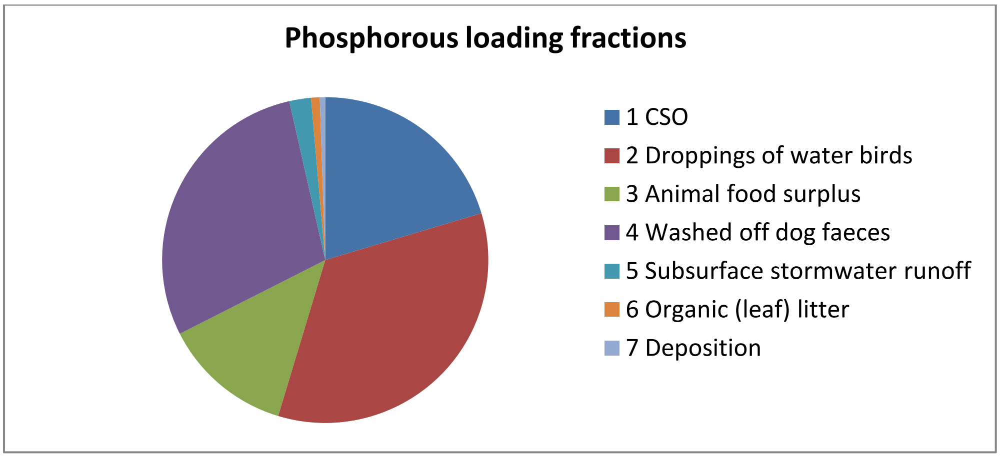

Figure 5.22 Relative contributions to the phosphorous loading of the Floresvijver.

Ecosystem development

The microcosms formed by the biofilters are two small oases for macroinvertebrates and zooplankton in a further turbid and monotonous pond. The biofilters can be considered drops in the ocean while they only represent 1,6 ‰ of the total pond surface. So the introduction of the two biofilters directly enhanced the aquatic ecosystem of the Floresvijver but the extend is very limited.

Further development of the aquatic ecosystem is not expected with the application of the two biofilters alone. Especially the very limited transparency combined with the lack of shallow and stable (i.e. consisting of consolidated sediments) riparian zones hinders the development of submerged macrophytes. The analysis of the volatile fraction of the suspended solids shows that inorganic components contribute significantly to turbidity in spring (see Figure 5.18). Sediment resuspension by fish, like bream and carp, foraging on benthic invertebrates is likely to be the dominant factor of this turbidity. The effect of wind is probably restricted to storm situations while fetch length is limited and the pond is surrounded by trees and buildings (see Figure 5.7).

Measures for overall enhancement of the Floresvijver should focus on the creation of favourable conditions for (submerged) macrophytes to sprout. Therefore, light needs to be able to penetrate through the water column and reach the bottom sediment. This can be achieved by decreasing turbidity with, for example, fish stock management (i.e. removal of benthivorous fish) or decreasing water depth by creation of shallow zones in the pond (preferably with sandy soils). The biofilters can contribute to the reduction of turbidity too but this only has a local impact. A combination of biofilters and a shallow zone might establish the favourable conditions for macrophytes to develop (see Figure 5.23).

Figure 5.23 A possible combination of biofilters and a shallow zone in the Floresvijver.

Chapter 6

Conclusions and recommendations

6.1 Conclusions

During this research, insight was gained on the functioning of the Bright Water Company floating helophyte filter and its ability to contribute to urban surface water quality enhancement. The conclusions on these two subjects are described separately while optimal functioning is required for possible enhancements.

6.1.1 Functioning

The biofilter is a floating treatment system that actively drains a filter bed with bog plants growing in it. The influent is provided by free inflow of surrounding surface water. Its water treatment ability depends on various processes including filtration, adsorption and biochemical transformations. Additionally, its inner reservoir serves as a habitat for small aquatic organisms like zooplankton and macroinvertebrates.

The physical filtration of the biofilter reduces the suspended solids content of the effluent to approximately 10 mg/l. The retained solids form a sludge layer on top of the filter bed and do not penetrate the filter material. The formation of the sludge layer increases the hydraulic resistance and thereby reduces the hydraulic capacity of the biofilter. Removal of this layer by means of a vacuum cleaner proved to be relatively easy. Biogas formation inside the filter bed turned out to be an another important factor in hydraulic capacity reduction. Entrapment of gas bubbles inside the filter material hinders the vertical water flow through the biofilter. Adjustments on the design of the (top layer of) filter bed material could provide sustainable hydraulic performance.

The nutrient removal capacity of the biofilter largely depends on specific nutrient absorbents which are integrated in the filter bed. The contributions of biological uptake and bacteriological processes are minor. This research did not determine the effective nutrient removal capacity of the biofilter but, based on several assumption, an estimation could be made. The optimal performance of the absorbents in the current design is reduced due to anaerobic reduction processes which occur in the filter bed. Placing an absorbent column in the pump house and let the water flow through it after aeration could improve this.

The inner reservoir of the biofilter functions as a microcosm. Zooplankton and macroinvertebrates flourish in the water layer on top of the filter bed. It provides shelter and breeding ground with a continuous supply of water from the surroundings which include food sources (e.g. algae) and which limits temperature increase during sunny days.

6.1.2 Urban surface water quality enhancement

Water managers can implement various measures to improve the water quality of urban water bodies that suffer from eutrophication and its manifestations (e.g. turbid water and algae blooms). The application of the biofilter increases the ability of a water body to cope with nutrient loadings and enhances its aquatic ecosystem.

The nutrient carrying capacity of the water body is increased through nutrient removal by the biofilter (if the biofilter is considered part of the water system). This system approach is efficient for situations with diffusive nutrient sources which are notoriously difficult to address directly (e.g. bird droppings). The significance of the nutrient removal by the biofilter depends on the nutrient loading conditions on site and (thereby) the biofilter's nutrient removal, and the number of biofilters applied.

The upper water layer of the inner reservoir of the biofilter is a suitable habitat for zooplankton and macroinvertebrates. The application of biofilters therefore adds to their habitat of a water body. The interaction of the organisms in the biofilter with the surroundings is not known but there are no restricting design factors for horizontal migration. Daphnia are key elements in the food web to help prevent the occurrence of algae blooms. The habitat of the biofilters proves to be a valuable addition to urban ponds while they often lack daphnia's hide outs (e.g. well developed riparian zones). The number of applied biofilters determine the surface area and volume of the additional habitat.

The reduced suspended solids concentration of the effluent increases the transparency of the treated water. The effluent can be used to create a shift in local transparency of the water body. This increased transparency can be applied at a location where macrophyte development profits from it. In this way, the biofilter can function as an internal measure too.

6.2 Recommendations

During this research, several interesting observations were made which can lead to further improvement of insight, functioning and application of the biofilters.

The nutrient removal capacity of the biofilters is an important aspect of its functioning. Determination of this capacity by means of an appropriate measurement campaign, taking into account the various influences and their time scales, is recommended. The current measurement scheme of the water board Noorderzijlvest provides hardly any insights in its functioning and in the effects on the water body its applied in.

Another interesting aspect for further research is the degree of interaction of organisms inside the biofilter with the surrounding surface water. Especially more knowledge on horizontal daphnia migration can help to apply biofilters as daphnia incubators and prevent algae blooms in the surrounding surface water by grazing of these daphnia (i.e. direct top-down control).

The composition of the filter bed can easily be adjusted to specific requirements. The conditions of the location of application and explicit objectives should be taken into account and implemented as modifications on the filter bed design. This aspect will especially be important when multiple biofilters are applied in a single water system. Varying filter bed design with serial circuit operation could possibly lead to increased efficiencies.

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Annexes

Annex A Treatment quantity calculation

Phosphorous

1. $TSS = Q * TSSinfluent * \ Efficiency = \left(500\frac{l}{h} * 24\frac{h}{d} * 365\frac{d}{y}\right) * \left(\frac{30}{1000}\frac{g}{l}\right) * \ 0{,}75 = 98550\ g/y$

2. Adsorption = Adsorbent mass x Adsobent capacity = $40\ kg * \ 50\ g/kg = 2000\ g$

3. Reed = P fraction x biomass x filter surface = $3\frac{g}{kg} * 3\frac{kg}{m2\ y} * 4\ m2 = 36\ g/y$

   Periphyton = P fraction x biomass x growth surface = $20\frac{g}{kg} * 0{,}05\frac{kg}{m2\ y} * 10\ m2 = 10\ g/y$

Nitrogen

1. $TSS = Q * TSSinfluent * \ Efficiency = \left(500\frac{l}{h} * 24\frac{h}{d} * 365\frac{d}{y}\right) * \left(\frac{30}{1000}\frac{g}{l}\right) * \ 0{,}75 = 98550\ g/y$

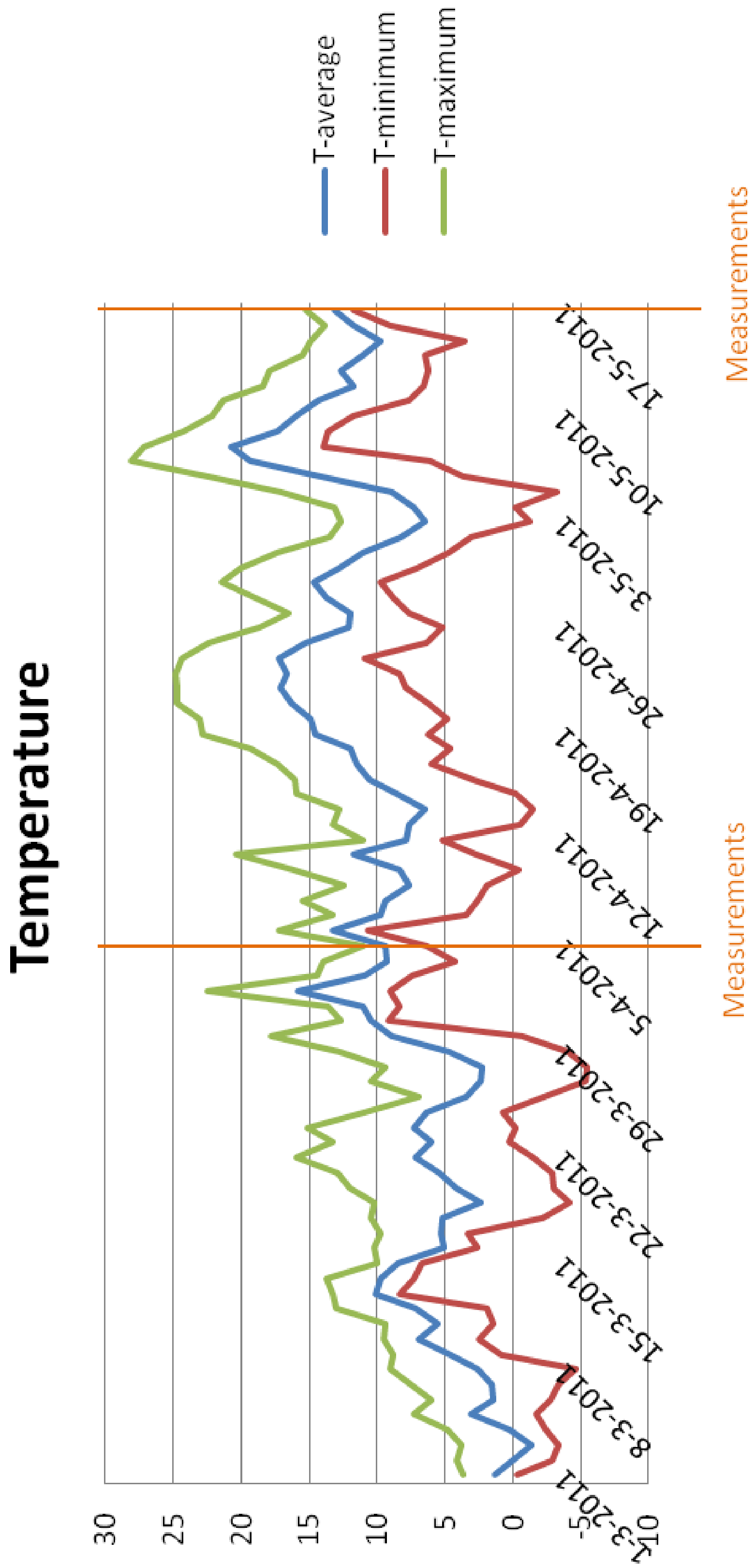
2. Adsorption = Adsorbent mass x Adsobent capacity = $550\ kg \ * \ 24\ g/kg = 13200\ g$

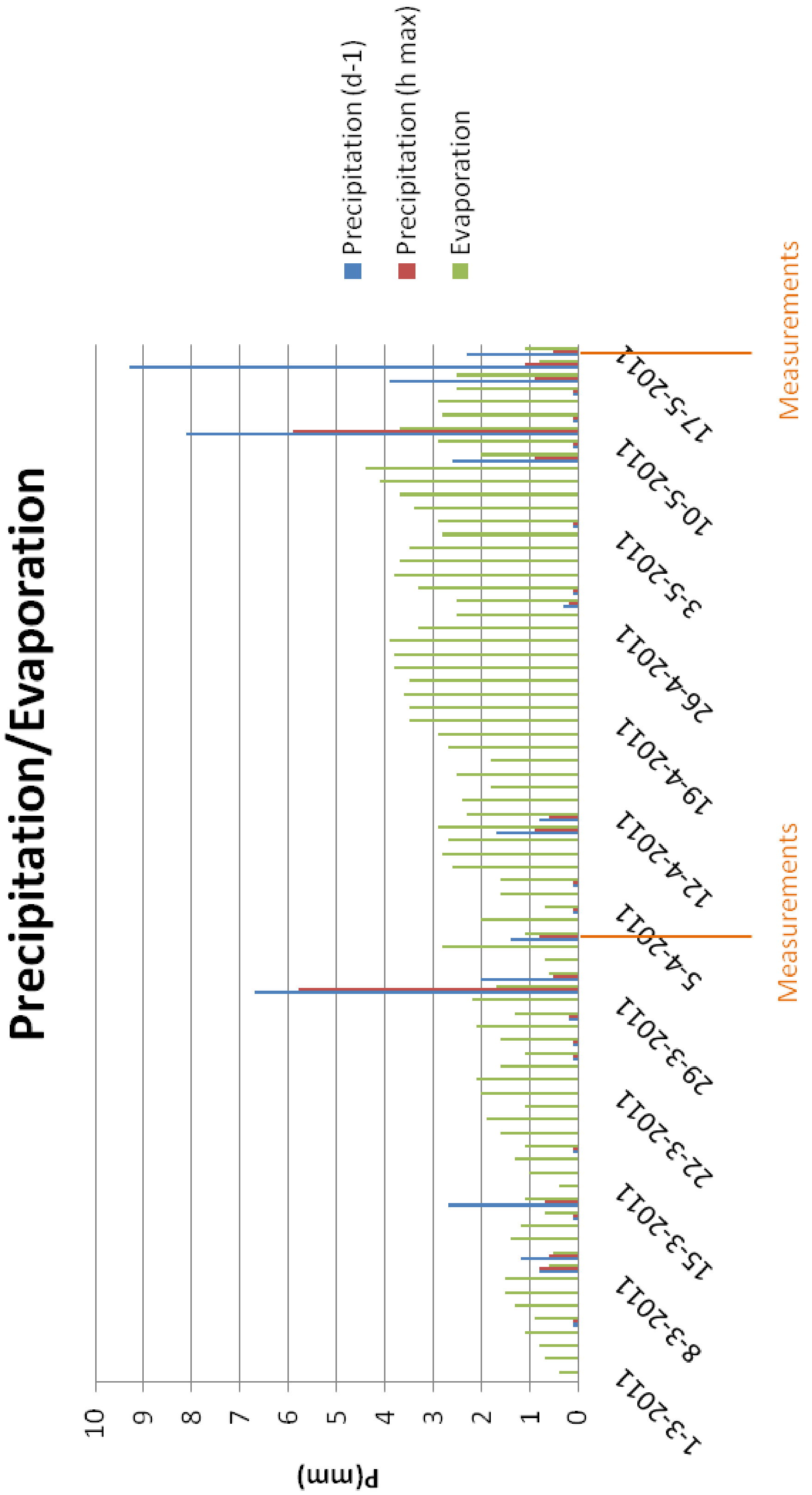
3. Reed = N fraction x biomass x filter surface = $20\frac{g}{kg} * 3\frac{kg}{m2\ y} * 4\ m2 = 240\ g/y$

   Periphyton = N fraction x biomass x growth surface = $80\ \frac{g}{kg} * 0{,}05\frac{kg}{m2\ y} * 10\ m2 = 40\ g/y$

   Denitrification = rate x days x surface = $1\frac{g}{m2} * 365\frac{d}{y} * 4\ m2 = 1460\ g/y$

Annex B Weather data (KNMI station 280; Eelde)




Precipitation/Evaporation
P(mm)
0
1
2
3
4
5
6
7
8
9
10
1-3-2011
8-3-2011
15-3-2011
22-3-2011
29-3-2011
5-4-2011
12-4-2011
19-4-2011
26-4-2011
3-5-2011
10-5-2011
17-5-2011
Measurements
Measurements
Precipitation (d-1)
Precipitation (h max)
Evaporation

Annex C Aquatic organisms found in the filter reservoir

Figure C1 Filtration of 200 ml water from the inner reservoir showed over a hundred daphnia and various other zooplankton.

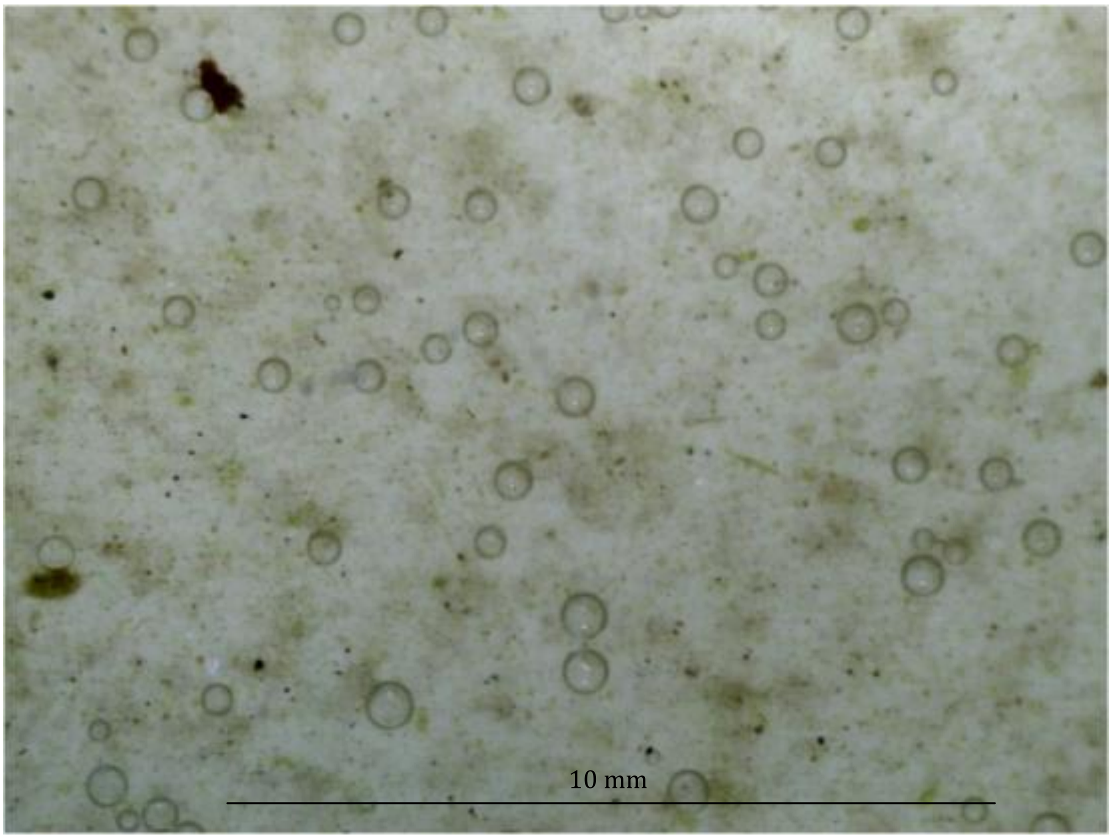


Figure C2 Examination of the filter residue with a microscope.

Figure C3 Protozoa (Paramecium)

Figure C4 Unknown.

Figure C5 Rotifer (Euchlanis triquetra)

Figure C6 Copepod (Eudiaptomus gracilis)

Figure C7 Water flea (Daphnia)

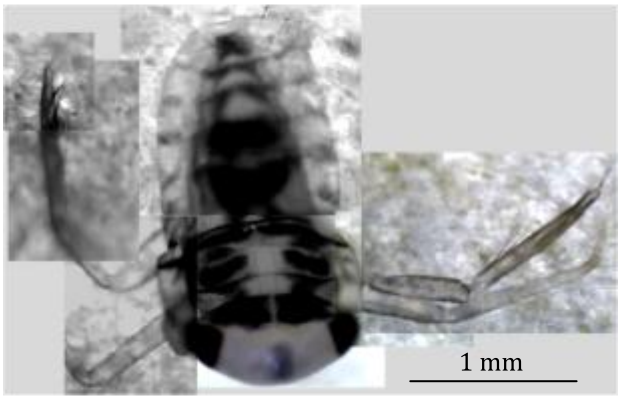


Figure C8 Larva of lesser water boatman (Corixa punctata)

Figure C9 Lesser water boatman

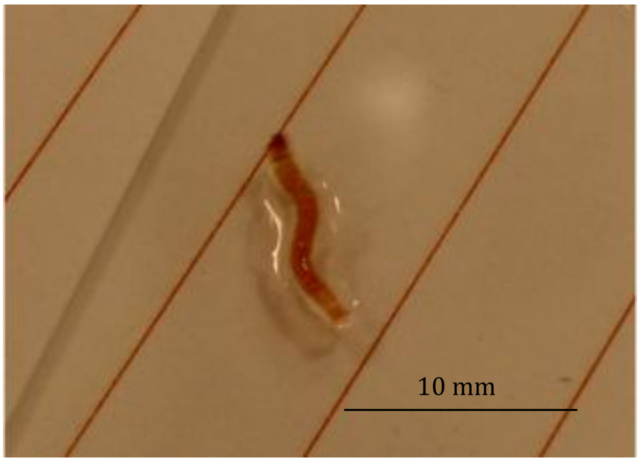


Figure C10 Midge larva (Chironomus larva)

Annex D Results of the laboratory analyses

Total suspended solids (mg/l), April 5th							
	Influent			Effluent			Surface water
Time	Duplomin	Duplomax	Average	Duplomin	Duplomax	Average	
9:30	37	47	42	5	16	11	53
10:00	41	41	41				
10:30							
11:00	39	40	39				
11:30							
12:00	46	48	47				
12:30				7	11	8	
13:00							
13:30							
14:00				14	17	15	
14:30				15	17	16	
15:00				9	12	10	43
15:30	41	41	41	5	8	6	
16:00				6	18	12	
16:30							
17:00							
17:30				1	3	2	

Total Suspended Solids (mg/l), May 17th					
	Influent			Effluent	Surface water
Time	Duplomin	Duplomax	Average		
9:30	149	153	151		
10:00				9	
10:30	136	146	141		63
11:00	108	115	111		
12:00	95	104	100	16	
13:00				9	
14:00				11	
14:30					
15:00	87	91	89	8	
15:30					
16:30	52	53	53	6	
16:30				7	

Volatile fraction of Total Suspended Solids (-)					
Time	Influent	Effluent1	Effluent2	Surface water	Sediment
9:30	0,37				
10:00		0,46			
10:30	0,41			0,37	0,05
11:00	0,40				
12:00	0,40	0,34			
13:00		0,35	0,67		
14:00			0,76		
14:30					
15:00	0,38		0,56		
15:30					
16:30	0,45	0,61			
16:30			0,60		