Key factors for biodiversity of urban water systems

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Contents

Urban water system Nijmegen. Photo: Kim Vermonden

Chapter 1

Introduction

Kim Vermonden

Urbanization and urban water systems

Our world is rapidly urbanizing (Grimm et al., 2008). In the developed countries urban population already accounts for approximately 73% of the total population (United Nations, 2008). The number of mega-cities with more than 10 million inhabitants increased from two in 1950 to 20 in 2005. In the Netherlands approximately 83% of the population lives in urban areas and this percentage will probably increase to 92% by the year 2050. Cities are microcosms of global change and perfect to study ecosystem dynamics and responses of biodiversity to change (Grimm et al., 2008).

Water systems play an important role in urban areas, providing vital services such as drinking water, irrigation, flood control, transportation, recreation and wildlife habitat (Postel & Carpenter, 1997). According to the European Water Framework Directive, three types of surface waters can be distinguished: natural, (heavily) modified and artificial water bodies (EU, 2000). Natural waters can be rivers, lakes, transitional or coastal waters. The second type refers to natural water systems that are (heavily) modified to fulfil human demands. The third type includes man-made water bodies that are especially constructed to provide services such as drainage of cities and towns.

Urbanization often has negative effects on existing natural water systems, altering the morphology, hydrology, water chemistry, flora and fauna (Paul & Meyer, 2001, Walsh et al., 2005). Water systems in urban areas are often canalized and banks are frequently protected with wooden boards or paved with stones to avoid erosion. The area of impervious or hard surfaces, such as roofs and roads, is large, resulting in higher and more frequent peak discharges. Water quality is usually degraded with high nutrient and contaminant loadings. Native flora and fauna diversity generally declines, while tolerant, often exotic species become more abundant (Paul & Meyer, 2001, Walsh et al., 2005, McKinney, 2006).

Nevertheless urban areas can also be an important habitat for flora and fauna. Urban woodlands in Rennes, France, accommodated more than 50% of the species present in peri-urban woodlands (Croci et al., 2008). Stewart et al. (2009) also found considerable plant diversity in urban woodlands, Christchurch City, New Zealand. Pryke & Samways (2009) showed that urban botanical gardens of indigenous plants had major invertebrate conservation value in South Africa. Langley et al. (1995) found that rotifer species richness was similar in urban ponds and reference sites. Collier et al. (2009) demonstrated that some urban streams can provide an important habitat for a range of native fish and macroinvertebrate species, including sensitive taxa. Le Viol et al. (2009) showed that macroinvertebrate family richness can be just as high in motorway storm water retention ponds, than in surrounding ponds in the wider landscape.

Climate change is associated with increasing amounts of precipitation and more frequent heavy precipitation events in North Western Europe. This requires adaptation of cities and could offer opportunities to rehabilitate surface water systems. Rehabilitation of (heavily) modified water bodies and optimal design of artificial urban waters give opportunities to integrate vital ecosystem services and to create at the same time habitats for biodiversity in urban areas (Savard et al., 2000, Palmer et al., 2004, Kazemi et al., 2009). Recently, many local, regional, national and international initiatives have been taken for the optimization of the design and management of urban water systems (e.g. Arnhemse Waterpartners, 2003, Wang et al., 2006, Arghyam, 2007, Urban Water Project Partnership, 2008). Nowadays, urban water management projects also take into account the potential biodiversity value of urban waters (Bryant, 2006, Kazemi et al., 2009). Knowledge of the structure and functioning of urban surface water systems as habitat for flora and fauna species is needed to determine key factors for aquatic biodiversity in urban areas, necessary to optimize their design and management for biological conservation.

Biological assessment

The ecological quality of water systems, including urban waters, can be assessed by measuring various biodiversity parameters (e.g., richness, Shannon index) at different scales (e.g., alpha, beta or gamma). The diversity of a system depends on many factors. This thesis focuses on the importance of biotic and abiotic conditions for biodiversity of urban surface water systems. Biodiversity in urban water systems is placed within the theoretical frameworks on productivity and disturbance, species pools and invasions as explained below.

Relevance and indicators of biodiversity

Biological diversity includes diversity within species, between species and of ecosystems (UNEP, 1992). In April 2002, the Parties to the Convention on Biological Diversity committed themselves to achieve a significant reduction of the current rate of biodiversity loss by 2010 at the global, regional and national level as a contribution to poverty alleviation and to the benefit of all life on Earth (COP, 2002). Biodiversity is the foundation upon which human civilization has been built. In addition to its intrinsic value, biodiversity provides goods and services that underpin sustainable development in many ways. The Curitiba Declaration on Cities and Biodiversity (2007) affirmed the importance to integrate biodiversity in urban planning and development, with a view on improving the lives of urban residents.

Biodiversity can be measured at three different spatial scales: alpha, beta and gamma (Whittaker, 1972). Taxa richness within a community or area is expressed as alpha diversity. The difference between communities or areas is called beta diversity. Gamma diversity is the overall species diversity for the different communities or areas within for example a geographical region. Biodiversity can be expressed in many different ways. Species richness or the number of species in an area or a sample is the simplest form. Species evenness is usually calculated as the Shannon-index or Simpson-index using species and their abundances. More recent indices also incorporate other values, such as ecosystem values or economic values (Yoccoz et al., 2001). For example rare or endangered species could be weighted more heavily than common species. Single species, such as indicator species, flagships, umbrellas or keystones can also be used as the base of biological conservation, although it is difficult to prove if whole ecosystems profit from this approach (Simberloff, 1998). Biodiversity depends on many different factors, for example the regional species pool, geographical location, nutrient status, frequency and intensity of disturbances or invasion by exotic species.

Intermediate disturbance theory

Urban water systems are regularly disturbed by for example mowing of aquatic weeds, dredging of sediment and storm water peak flows. Theory and empirical evidence suggest that maximum species richness is reached at moderate frequencies or intensities of disturbance, thus at intermediate disturbance (Hobbs & Heunneke, 1992). Disturbance creates possibilities for new species to colonize an area and species richness is hereby increased when various successional stages coexist at intermediate disturbance levels (Connell, 1978). Severe disturbance would extinct much of the later succession species, while absence of disturbance would give no opportunities for early-succession species.

Species pool hypothesis

Large-scale processes determine which and how many species are available for the local community (Zobel, 1997). The local species pool often depends on the size and composition of the regional species pool (Caley & Schluter, 1997, Zobel, 1997, Partel & Zobel, 1999). The terms local and regional refer to the spatial scales at which ecological and biogeographic processes dominate (Cornell & Lawton, 1992). On the regional scale long-distance dispersal, speciation, wide-spread extinction, and fluctuations in species' distributions across broad geographic regions play an important role. Predation, parasitism, competition and abiotic fluctuations or disturbances play an important role on the local scale. Regional processes are likely to determine the species pools from which local communities can be assembled and set the upper limit on local species richness (Caley & Schluter, 1997). The local species pool is therefore determined by the regional species pool, biotic interactions and abiotic limitations.

Regional and local diversity are often related with species-area relationships (Preston, 1960, MacArthur & Wilson, 1967, Rosenzweig, 1995). When sampling a larger area, more species can be found, because larger areas harbour more individuals of different species (higher chance to find rare species), more habitats due to environmental heterogeneity and at much larger scale, more biogeographical provinces.

Invasion biology

City development homogenizes the physical environment on one hand, but creates new habitats on the other hand. Urban-adaptable species become wide-spread across the planet (McKinney, 2006). Many native species are replaced by non-native species. Urbanization is often associated with high exotic species richness, because there is high propagule pressure and there are a lot of open niches due to human activities.

Davis et al. (2000) argued that increased resource availability, often after a disturbance, is a key factor for increased invasibility of ecosystems. In that case, resource availability can increase due to a pulse in resource supply and/ or reduced uptake of resources. Invading species can then capture the increased amounts of resources. Others have also shown that disturbances can enhance invasion of natural communities (Hobbs & Heunneke, 1992, Eschtruth & Battles, 2009). Water quality improvement after periods of water pollution and maintenance measures such as cleaning of the waterways can accelerate invasions by fast-growing exotic species (Van der Velde et al., 2002).

Native biodiversity could also prevent invasion, by lowering the available resources (Elton, 1958, Stachowicz et al., 1999, Van der Velde et al., 2006). Communities composed of many strongly interacting species limit the invasion possibilities of species belonging to taxonomically related groups (Case, 1990).

Aim of the thesis

The main objective of this thesis is to determine the key environmental factors for biodiversity in urban surface water systems. To achieve this objective we investigated how local and regional processes influenced macroinvertebrate and macrophyte species composition and species richness.

Water quality is very important for flora and fauna species composition and diversity (Smith et al., 1999, EU, 2000, Wang et al., 2007). Water quality in urban water systems is usually determined by the total area of impervious surfaces, such as roofs, buildings and roads (Scheuler, 1994, Brabec et al., 2002, Walsh et al., 2005). Nevertheless, in lowland areas along large rivers, there can be a strong influence of river-groundwater-sediment-surface water interactions (Charette & Buesseler, 2004, Krause et al., 2007).

We studied different aspects of biodiversity. Alpha diversity within urban water systems was related to several environmental variables. Beta diversity was related to different disturbance levels, by comparing urban, rural and semi-natural water systems. The diversity in urban water systems was also related to total diversity in the Netherlands. Life-history strategies can be used to understand how species cope with their environment (Verberk et al., 2008). In this study life-history strategies were used to understand differences in chironomid associations and changes over time. Furthermore, invasibility of urban water systems by exotic species was related to resource availability and native biodiversity.

The following specific research questions are addressed in this thesis:

(1) Is water quality in urban water systems determined by local storm water runoff from impervious area or regional upward seepage of groundwater?
(2) Is the local macrophyte species composition and species richness limited by the regional species pool?
(3) What is the biodiversity value (e.g. species richness, number of red list species) of urban water systems in comparison to rural and semi-natural waters?
(4) Which environmental factors determine the occurrence of plant and animal species in urban surface water systems?
(5) Do life-history strategies help to understand the ecological processes and human disturbance in urban surface waters?
(6) Do native biodiversity, resource availability and/ or environmental heterogeneity determine invasiveness of urban water systems by exotic species?
(7) What are the implications of the results for optimization of the design and management of urban water systems with respect to biodiversity?

Outline of the thesis

Urbanization and rehabilitation determine the biological diversity of flora and fauna in urban water systems, either directly by maintenance measures such as mowing and dredging or indirectly, via for example water quality (Figure 1). Chapter 2 analyses local and regional factors determining water quality of urban waters in our study area. We hypothesize that both regional influence of upward seepage and storm water runoff from impervious areas determine the water quality of urban water systems. In the subsequent chapters, the impact of local environmental variables and regional species pools on macrophytes and macroinvertebrates are investigated.

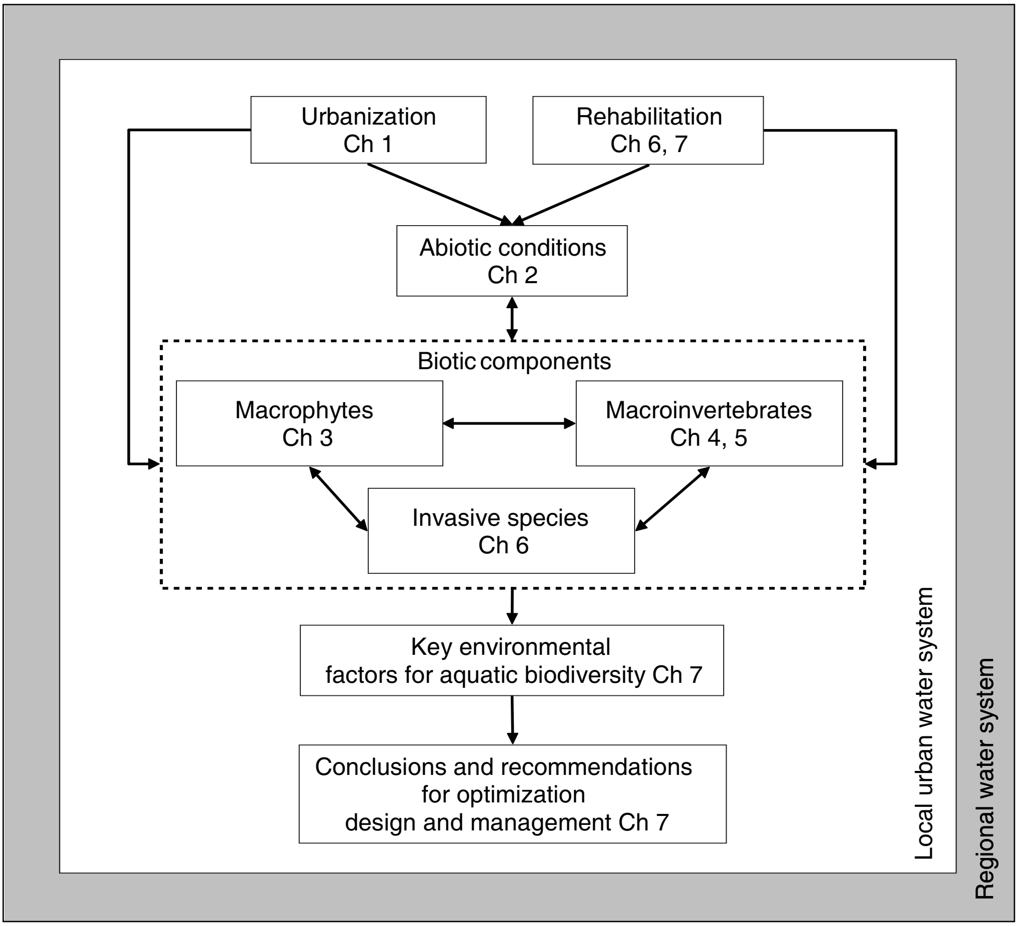

Figure 1 Overview of the issues addressed in this thesis with the corresponding chapters.

Macrophytes are very important for macroinvertebrates as habitat, food source and shelter from predation (Crowder & Cooper, 1982, Dvořak & Best, 1982, Newman, 1991). Macrophytes are directly dependent on soil and water quality and are also important for the production of organic material influencing the bottom sediment processes (Den Hartog & Van der Velde, 1988). In chapter 3, we hypothesize that macrophyte species composition and species richness are determined by local environmental variables and the regional species pool.

Macroinvertebrates also influence their environment and macrophytes, for example by grazing and the decomposition of organic material. Macroinvertebrate assemblages and diversity are related to environmental variables in chapter 4. It is expected that taxa richness, Shannon index, number of red list species, exotic species, and rareness of species are lower in urban water systems than in similar water systems in rural areas.

Special attention is given to chironomids in Chapter 5, because of the high species richness and wide environmental tolerances in this group (Armitage et al., 1995). Life-history strategies are used to explain variability between chironomid assemblages in relation to driving environmental factors and dredging. Dredging removes the polluted sediment and organic debris. Decomposition of organic material is hereby reduced and oxygen conditions are likely to improve. We therefore expect higher chironomid diversity after dredging and colonization by more sensitive species.

It is expected that biodiversity influenced the invasibility of the water systems (Chapter 6, Van der Velde et al., 2006), by reducing the available resources for exotic species (Elton, 1958, Stachowicz et al., 1999). The effects of resource availability and environmental heterogeneity on the invasibility of urban waters is tested.

In chapter 7 we consider the key environmental factors for optimization of biodiversity values in the urban water systems investigated. Some recommendations for urban water management are also given.

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Urban water system Nijmegen. Photo: Kim Vermonden

Chapter 2

Does upward seepage of river water and storm water runoff determine water quality of urban drainage systems in lowland areas? A case study for the Rhine-Meuse delta

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Hydrological Processes 23: 3110-3120

Abstract

The water quality of urban drainage ditches in lowlands in the Rhine-Meuse delta was analysed with Principal Component Analysis (PCA) during a dry period and a rain storm, and related to seepage of polluted river water and effective impervious area (EIA). This was done in order to test the hypothesis that seepage of river water and storm water runoff from impervious areas strongly determine the water quality of urban drainage systems along large lowland rivers.

Our analysis revealed that upward seepage of groundwater originating from rivers Rhine and Meuse was positively correlated with nitrate, potassium, sodium and chloride and negatively correlated with alkalinity, calcium, magnesium and iron. EIA was correlated with very few environmental variables (i.e. phosphate, pH and iron in the dry period and iron during the rain storm). Nickel and zinc concentrations generally exceeded the maximum allowable concentrations (MAC), while lead and phosphorus concentrations were just above the nutrient standards and MAC in a few locations during the rain storm. To optimize water quality in urban water systems, attention should be paid to all sources of pollution and not only to EIA. The impact of local groundwater seepage originating from large rivers in lowlands on the chemistry of urban water systems is often underestimated and should be taken into account when assessing water quality and improving water quality status.

Introduction

In this chapter, a case study is presented on the water quality of urban water systems in the Rhine-Meuse delta. This is done to test the hypothesis that upward seepage of river water and storm water runoff determine water quality of urban drainage systems in lowland areas. This hypothesis underpins a major environmental management issue of national and international significance, because urbanization is a wide-spread phenomenon and the quality of urban water systems becomes increasingly important in towns and cities.

Approximately 3.5 billion residents (50% of the human population) currently live in urban areas (United Nations, 2008). By the year 2030, an increase of approximately 1.5 billion residents is expected in the urbanised areas of the world. Throughout history, cities have developed especially along rivers and in deltas (Grimm et al., 2008). For instance, in the Rhine-Meuse river basins (Europe), urban and industrial areas account for 25.7% of the catchments and population density is on average 319 people per km^2 (World Resources Institute, 2003).

Water systems (e.g. canals, ditches) drain areas for urban development. Moreover, urban water systems are considered attractive for citizens and are therefore given a more prominent place in new suburbs. An optimal design and management of these systems increasingly provides opportunities to conserve biodiversity lost in natural and rural areas (Palmer et al., 2004, Wang et al., 2006, Vermonden et al., 2009). In addition, the European Union Water Framework Directive (WFD) demands an enhanced protection of all water bodies in member states, including artificial and heavily modified ones, with the aim to achieve good ecological potential and good surface water chemical status by 2015 (EU, 2000). The type and magnitude of significant anthropogenic pressures have to be determined and the susceptibility of the surface waters to these pressures has to be assessed.

Existing natural water systems are strongly influenced by urbanization, changing the hydrology, morphology, water chemistry and flora and fauna (Ehrenfeld, 2000, Paul & Meyer, 2001, Walsh et al., 2005, White & Greer, 2006). Other urban waters are constructed and are therefore already different in these aspects from natural waters. Urbanization changes both the type and the magnitude of runoff processes (Booth & Jackson, 1997). Vegetation is cleared, soils are compacted, ditched and drained, and land surface is covered with impervious roofs and roads. An increase in impervious area not only causes a higher runoff volume and velocity, but also higher and more frequent peak discharges (Schueler, 1994). Water levels are kept constant in urban areas by artificial measures (e.g. damming and pumping). Water systems in urban areas are often canalized or are canals. Their morphology is characterized by wider, deeper and less complex systems and usually have higher nutrient and contaminant loadings than natural water systems.

Catchment imperviousness is generally considered one of the primary determinants of the quantity and quality of urban storm water runoff delivered to receiving urban water systems (Schueler, 1994, Booth & Jackson, 1997, Walsh, 2000, Brabec et al., 2002). Water quality variables, algal biomasses, diatom and macroinvertebrate assemblages changed drastically with increasing impervious area (Walsh et al., 2005).

In lowland areas along large rivers, aquatic ecosystems can be strongly influenced by river-groundwater-surface water interactions (Charette & Buesseler, 2004, Krause et al., 2007). While the pollution of groundwater by agricultural, industrial or urban areas and the potential effects of polluted groundwater on (semi) natural water bodies have been assessed in several studies (Van den Brink et al., 1993, Charette & Buesseler, 2004, Lucassen et al., 2004, Rodgers et al., 2004, Kumar et al., 2009), the effects of seepage of polluted river water on aquatic systems in urban areas have not been studied yet. Normally, rivers receive water from the catchment area, either directly from superficial streams or indirectly by seepage of ground water. In lowland areas water levels in rivers and canals can be very high due to dike construction and a decrease of the floodplain area. The hinterland is systematically drained to increase the area for agriculture and urban development (Havinga & Smits, 2000). The drop in groundwater tables causes soil compaction and oxidation of organic soils, which leads to soil subsidence. This subsidence process forces people to deepen drains and ditches and dig canals to lower the water table further, thereby provoking an irreversible subsidence process. The hydrological situation in these lowland areas has changed drastically, and seepage of river water can play an important role in urban and rural areas in the hinterland (Figure 1).

We hypothesized that seepage of river water and storm water runoff from impervious areas strongly determine the water quality of urban drainage systems along large lowland rivers.

Therefore, the following research questions were addressed:

(1) What are the relationships between the storm water runoff, upward seepage of groundwater and water quality of urban drainage systems along large, lowland rivers in a dry period and during a rain storm?

(2) What are the main bottlenecks for improving the water quality in urban drainage systems in lowland areas?

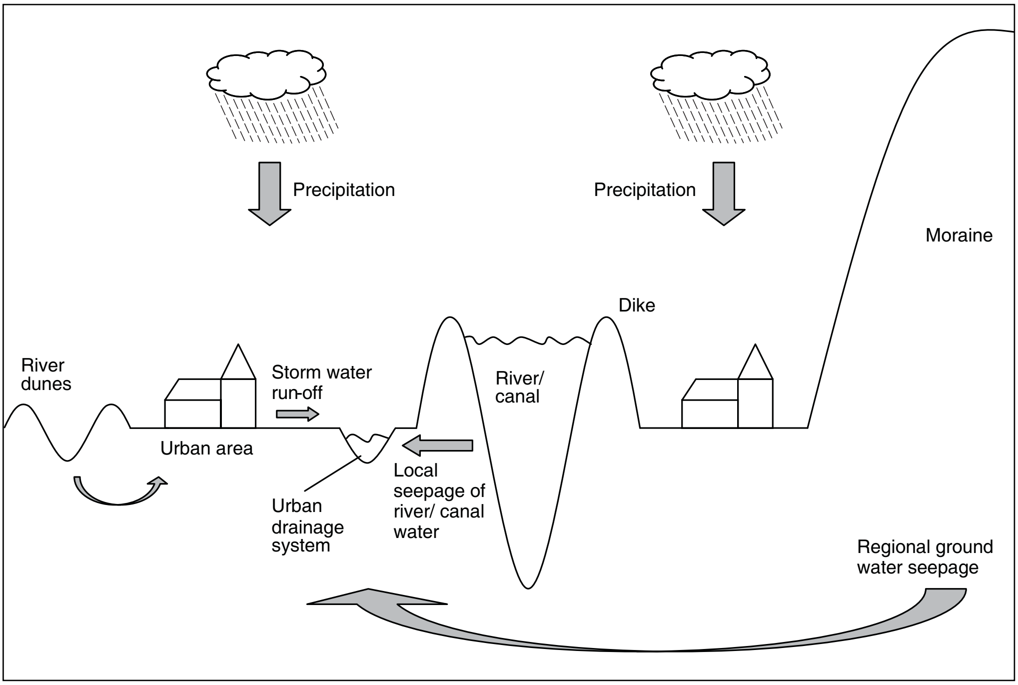

Figure 1 Conceptual overview of the hydrological cycle in the lowland areas of a river basin.

The city of Nijmegen in the Netherlands is selected for a case study to analyse the influence of storm water runoff and seepage of Rhine-Meuse water on the surface water quality (i.e. pH, alkalinity, nutrients and metal content) of urban drainage systems. Principal component analysis (PCA) is used to determine the major hydrological and chemical processes that influence the water quality in urban drainage systems. The implications of the results for cost-effective water pollution control in urban areas are discussed.

Materials and methods

Study area

The study area is situated in the former floodplains of the rivers Meuse and Waal in the eastern part of the Netherlands. The river Waal is the main distributary of the river Rhine in the Netherlands (Figure 2). During the Weichselian-Pleniglacial (~70,000–13,000 BP) the rivers braided through the area, creating a complex geological pattern (Pons, 1957, Theunissen, 1960, Berendsen & Stouthamer, 2000). During the relatively warm Bølling-Allerød interstitial (~13,000–11,000 BP) rivers changed to incised meandering streams. The braided channels were filled with clay or peat. The result of these developments is that layers of clay, sand, gravel and peat are intersecting each other in this area.

From approximately the year 1300 the study area was used for agriculture (pastures and hayfields) and from 1920-1927 the Meuse-Waal canal was constructed (Figure 2). In the 1960s the area was reclaimed for urbanization by digging drainage systems. Currently, approximately 40,000 inhabitants live in the western part of Nijmegen. Approximately 5% of the surface area in the western part of Nijmegen consists of water courses that are

Figure 2 Geographical location of the study area.

connected to each other with culverts (Figure 3). The slow-flowing, permanent water courses can vary from linear ditches to ponds with a width that is generally between 5 and 40 metres and depth up to 3 metres. Land use in the study area is predominantly residential, with an impervious area of approximately 30% (e.g. roads, buildings, houses, parking lots), and 65% is taken up by gardens, parks and other green areas.

A separate sanitary sewer system is constructed in the study area to transport sewage alone. Sewage is pumped directly to the sewage treatment plant and is not discharged into the urban drainage systems. Another pipe system is designed to convey storm water runoff directly to surface waters. The origin of the groundwater is mainly the Meuse-Waal canal, although regional ground water from the moraine on the eastside of Nijmegen and a river dune/ moorland complex on the westside of Nijmegen could infiltrate the urban drainage systems as well (Figure 1). The water level in the urban drainage systems is on average 6 to 6.5 m above Amsterdam Ordnance Datum (average sea level), while the Meuse-Waal canal water level is 7.6 to 7.7 m above the Amsterdam Ordnance Datum (Witteveen+Bos, 2006). Sewage is pumped directly to the sewage treatment plant and does not enter the urban drainage systems. The residence time of the water in the urban drainage systems is approximately 8-15 days.

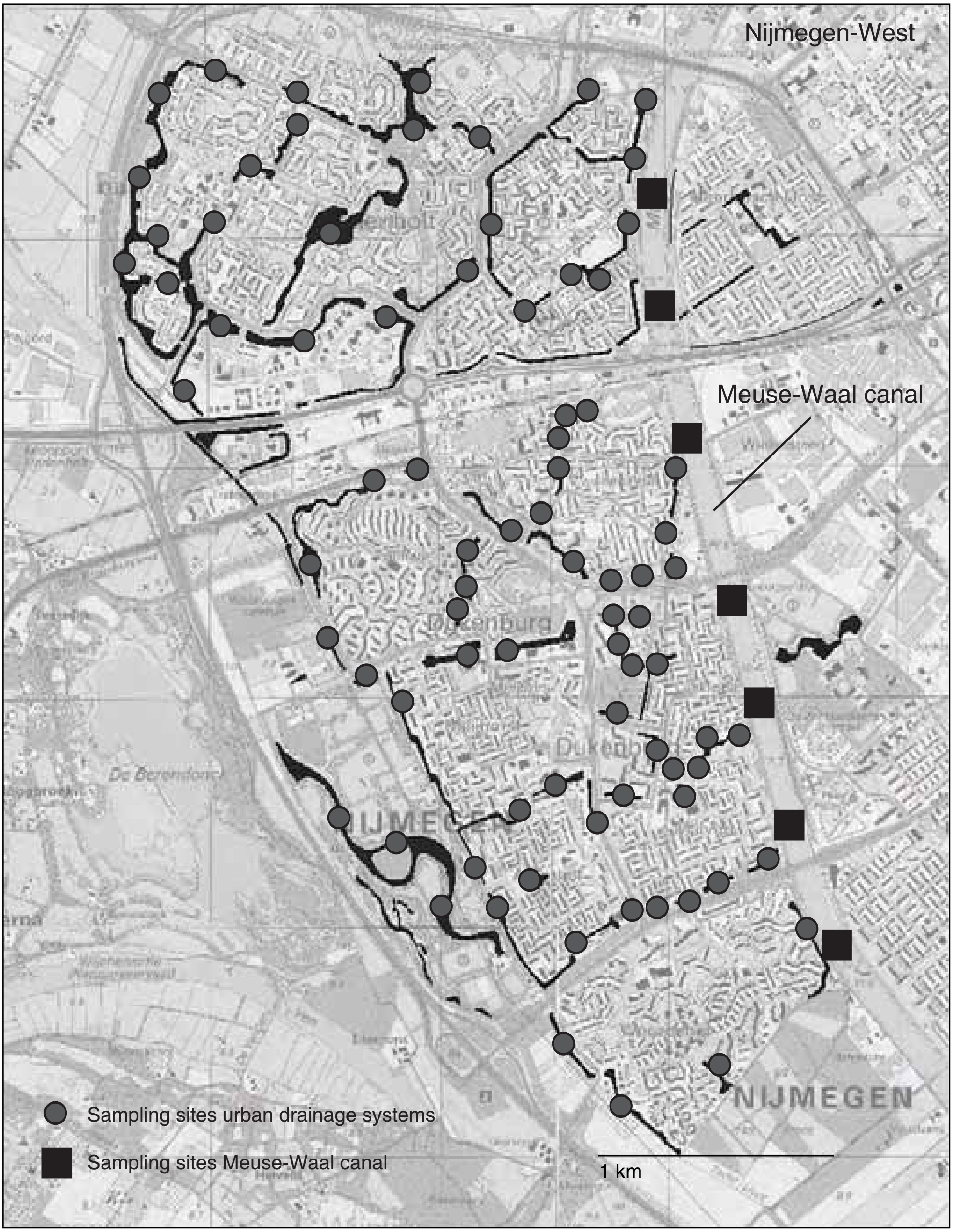

Figure 3 Detailed overview of the study area with the locations of sampling sites.

Sampling and analysis

In the study area, 82 sampling sites in urban drainage systems and 7 in the Meuse-Waal canal were selected and sampled twice (Figure 3). The first sampling took place during a dry period (17 and 18 April 2007 and no rain since 21 March 2007), the second sampling took place during heavy rainfall after the dry period since 21 March 2007 (7 and 8 May 2007, 41 mm and 16 mm rain, respectively) (Royal Netherlands Meteorological Institute, 2008). Surface water was collected in 100-ml polyethylene bottles. The pH and alkalinity were measured the following day, after storage of water samples overnight at 4°C. The pH measured in the laboratory was very consistent with the pH measured in the field (Hanna Combo meter). The following substances were measured colorometrically (Auto Analyzer 3, Digital colorimeter, Bran + Luebbe, Germany): nitrate according to Kamphake et al. (1967), ammonium according to Grasshoff & Johannsen (1972), chloride according to O'Brien (1962) and phosphate according to Henriksen (1965). Sodium and potassium were measured photometrically with a flame photometer (Radiometer, Copenhagen). Metals were measured by inductively coupled plasma mass spectrometry (Thermo Electron Corporation, United Kingdom). Nutrient and metal concentrations were compared with nutrient standards and maximum allowable concentrations (MAC), respectively (Ministry of Transport, Public Works and Water Management, 1998).

Effective impervious area and upward seepage

The effective impervious area (EIA) only includes impervious surfaces that are directly connected to streams (Booth & Jackson, 1997). EIA is the parameter normally used to characterize urban development in hydrological models, and is also used in this study. For each water body, the total acreage of EIA was calculated from a map of the study area visualizing all impervious surfaces connected to the storm water inlets.

Upward seepage was studied in three different ways: ground-water modelling, the determination of permeability of the soil and the Euclidian distance to the Meuse-Waal canal. Model data of upward seepage were acquired from Witteveen+Bos (2006). The upward seepage was modelled with MicroFEM (Harbaugh et al., 2000), after model calibration using field data from 24 piezometers. In this modelling upward seepage was divided in seven classes: 0-2.5, 2.5-5, 5-10, 10-15, 15-20, 20-30, >30 mm day^{-1}. These classes were numbered 1-7 and used to calculate correlations with water quality parameters. A model was made for winter and summer; average classes per sampling site were used for the analyses.

Data on soil permeability were obtained from maps (1:15,000) giving information on soils to a depth of 1 m (Pons, 1957). The soil types were arranged from least to most permeable (Terzaghi et al., 1996) and classified from 1 to 8, corresponding with the following soil types: heavy clay, heavy sand, silt, peat, eolian sand, silty gravel sands, mixed gravel sands and gravel.

The Euclidean distance of the sampling sites to the Meuse-Waal canal was determined with ArcGIS 9. Other factors such as (micro)biological activity, sorption, light regime and temperature influence the water quality of surface waters as well, but were not measured.

Data analysis

PCA was used to elucidate correlations between environmental variables and to identify the main patterns in the data set. Data on total nitrogen, total phosphorus, sodium and calcium concentrations were deleted before the PCA, because of strong correlations with nitrate, phosphate, chloride and alkalinity, respectively. Correlations were determined with Spearman's Rho correlation coefficient, because data was not parametric. A distinction was made between significance levels $p<0.05$, $p<0.01$ and $p<0.001$. Differences between water quality parameters in the urban drainage system, Meuse-Waal canal, the dry period and the rain storm were tested with a Mann Whitney test ($p<0.05$).

Results

The first five principal components account for 70% of the total variation in the environmental variables (Table 1). The dry period is clearly different from the wet period on the first principal component axis (PC1, Figure 4a). Ammonium, zinc and lead concentrations were most closely and positively associated with PC 1 (Table 1). Sulphur, magnesium and pH were negatively correlated with PC1. Upward seepage, soil permeability and the distance to the Meuse-Waal canal were associated with the second and third principal component (Table 1, Figure 4b and c). The following parameters were most strongly and positively correlated with these axes: nitrate, total nitrogen, phosphate, potassium, sodium and chloride. Alkalinity, calcium, magnesium and iron concentrations were negatively correlated with PC 2.

Table 1 Spearman's Rho correlations (R) of environmental variables, upward seepage, soil permeability, distance to the Meuse-Waal canal and EIA with principal components, eigenvalues of principal components and percentage of variance explained by the eigenvalues.

	PC 1	PC 2	PC 3	PC 4	PC 5
NO_3^-	–0.21**	0.68***	0.64***	–0.01	0.29***
NH_4^+	0.69***	–0.18*	–0.13	–0.02	0.20*
Total N	–0.01	0.71***	0.55***	0.05	0.34***
PO_4^{3-}	0.34***	0.38***	0.61***	–0.36***	0.16*
Total P	0.29***	0.27***	0.45***	–0.08	0.15
Total S	–0.66***	–0.03	0.23**	0.60***	0.20**
K^+	–0.16*	0.64***	0.79***	0.21**	0.24**
Na^+	–0.32***	0.48***	0.73***	0.08	0.04
Cl^-	–0.43***	0.38***	0.68***	0.09	–0.03
pH	–0.57***	–0.17*	–0.46***	0.34***	–0.35***
Alkalinity	–0.44***	–0.73***	–0.39***	–0.03	0.06
Ca^{2+}	–0.38***	–0.72***	–0.31***	–0.04	0.13
Mg^{2+}	–0.55***	–0.56***	–0.14	0.18*	–0.03
Li	–0.03	0.22**	0.32***	–0.55***	0.20**
Total Fe	0.17*	–0.59***	–0.36***	–0.14	0.30***
Al	0.21**	0.09	–0.14	0.57***	0.17*
Cd	0.16*	0.31***	0.23**	0.09	0.53***
Cu	–0.21**	0.41***	–0.09	0.68***	0.12
Ni	–0.17*	–0.08	0.06	0.23**	0.02
Pb	0.65***	–0.04	–0.06	0.13	0.14
Zn	0.77***	–0.03	0.11	–0.19*	0.22**
Upward seepage	0.03	0.58***	0.47***	–0.03	0.23**
Soil permeability	0.05	0.58***	0.47***	0.18*	0.18*
Distance to Meuse-Waal canal	–0.04	–0.74***	–0.60***	–0.06	–0.18*
EIA	0.02	–0.09	0.12	–0.13	0.02
Eigenvalues	3.88	3.33	2.36	1.66	1.28
% Variance	21.54	18.52	13.12	9.21	7.09

* $p<0.05$, ** $p<0.01$, *** $p<0.001$

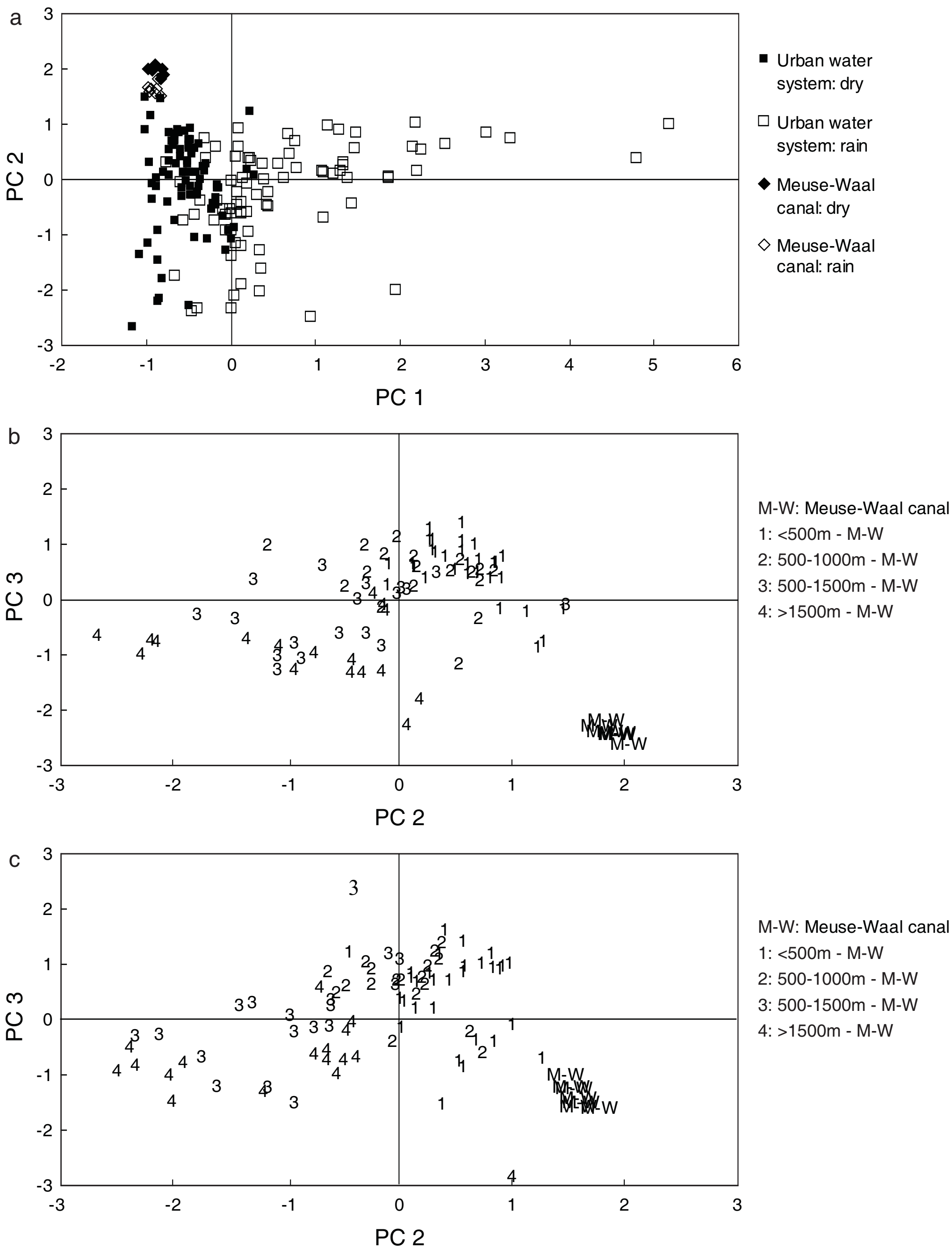

Figure 4 Principal component analysis (PCA) on the environmental variables of the urban drainage systems and the Meuse-Waal canal, with principal components (PC) 1 and 2 (a), and PC 2 and 3 in the dry period (b) and during the rain storm (c). The Euclidian distance to the Meuse-Waal canal is indicated with numbers 1-4 corresponding with <500m, 500-1000m, 1000-1500m and >1500m from the Meuse-Waal canal (M-W), respectively.

Most environmental variables were correlated with upward seepage, soil permeability and distance to the Meuse-Waal canal during the dry period as well as during the rain storm (Table 2). Figure 5 and 6 exemplify these relationships for nitrate, iron and phosphate. Nitrate, total nitrogen, phosphate, potassium, sodium and chloride concentrations correlated positively with upward seepage and soil permeability and negatively with distance to the Meuse-Waal canal. pH, alkalinity, calcium, magnesium and iron decreased with upward seepage and permeability and increased with distance to the Meuse-Waal canal.

Very few significant correlations were found with EIA; phosphate was positively correlated with EIA in the dry period and pH was negatively correlated with EIA in the dry period, while iron was positively correlated to EIA both in the dry period and during the rain storm. The maximum concentration of nitrate seemed to decrease with increasing EIA (Figure 5).

Ammonium, phosphorus, lead and zinc concentrations were higher in the urban drainage systems during the rain storm than in the dry period (Table 3). Nitrate, total-nitrogen, lithium, cadmium, copper concentrations, pH and alkalinity were higher in the Meuse-Waal canal than in the urban drainage systems.

Table 2 Spearman's Rho correlations (R) between environmental variables and upward seepage, soil permeability, distance to the Meuse-Waal canal and EIA in a dry period and during a rain storm.

	Dry				Rain storm (57mm)			
	Upward seepage	Soil permeability	Distance to Meuse-Waal canal	EIA	Upward seepage	Soil permeability	Distance to Meuse-Waal canal	EIA
NO_3^-	0.65***	0.50***	–0.70***	0.09	0.61***	0.52***	–0.68***	0.13
NH_4^+	–0.41***	–0.42***	0.60***	–0.10	0.09	0.26*	–0.19	0.10
Total N	0.61***	0.48***	–0.65***	0.11	0.58***	0.57***	–0.72***	0.04
PO_4^{3-}	0.19	0.17	–0.34**	0.27*	0.40***	0.41***	–0.60***	0.06
Total P	0.03	0.11	–0.03	0.02	0.43***	0.43***	–0.69***	–0.03
Total S	–0.06	0.06	0.07	0.09	–0.03	0.00	–0.10	0.10
K^+	0.57***	0.73***	–0.78***	–0.04	0.50***	0.54***	–0.66***	0.09
Na^+	0.50***	0.66***	–0.70***	0.00	0.31**	0.21	–0.37**	0.20
Cl^-	0.48***	0.66***	–0.68***	–0.08	0.19	0.13	–0.24*	0.15
pH	–0.22*	–0.06	0.16	–0.23*	–0.38***	–0.31**	0.41***	–0.17
Alkalinity	–0.29**	–0.21*	0.41***	–0.05	–0.43***	–0.57***	0.63***	0.03
Ca^{2+}	–0.36***	–0.24*	0.51***	–0.02	–0.42***	–0.61***	0.63***	0.06
Mg^{2+}	–0.35**	–0.27*	0.48***	0.19	–0.39***	–0.56***	0.51***	0.14
Li	0.32**	0.10	–0.27*	0.08	0.31**	0.05	–0.24*	0.06
Total Fe	–0.50***	–0.40***	0.59***	0.28**	–0.29**	–0.35**	0.43***	0.22*
Al	–0.27**	–0.04	0.23*	0.02	0.18	0.37**	–0.32**	–0.15
Cd	0.16	0.27*	–0.14	0.14	0.49***	0.45***	–0.50***	0.03
Cu	–0.09	–0.03	0.13	–0.10	0.48***	0.68***	–0.66***	–0.14
Ni	0.18	–0.04	–0.11	–0.11	–0.18	–0.17	0.26*	0.14
Pb	–0.25*	–0.04	0.24*	0.04	0.20	0.40***	–0.36**	–0.07
Zn	–0.14	–0.06	0.22*	0.18	0.34**	0.42***	–0.41***	–0.01

* p<0.05, ** p<0.01, *** p<0.001

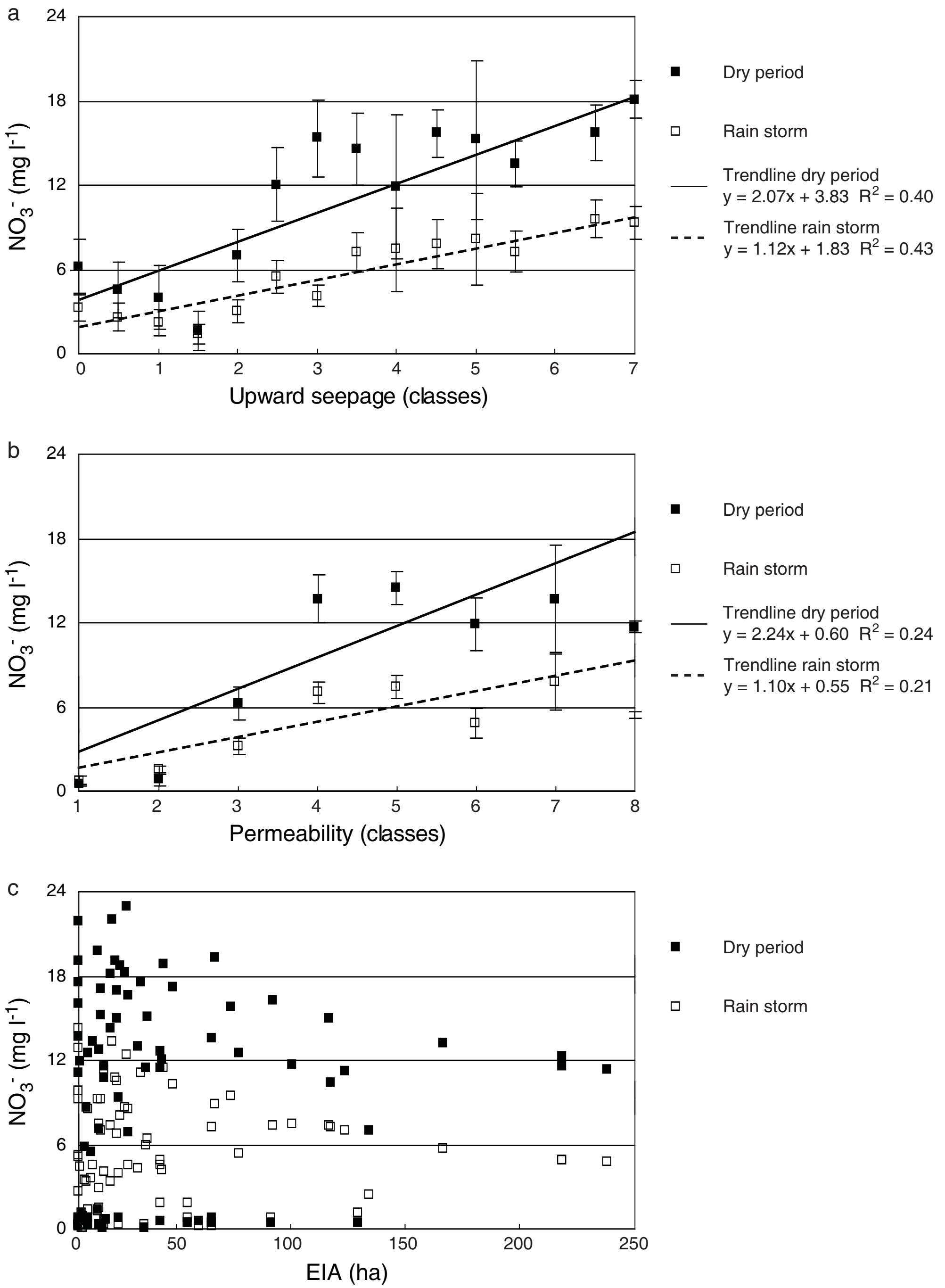

Figure 5 Relationship between nitrate content of surface water in urban drainage systems, (a) upward seepage, (b) soil permeability, and (c) effective impervious area (EIA) during the dry period and rain storm (average values for upward seepage and permeability classes are displayed with standard errors; the regression line is based on all measurements in the urban drainage systems (n=82), instead of averages).

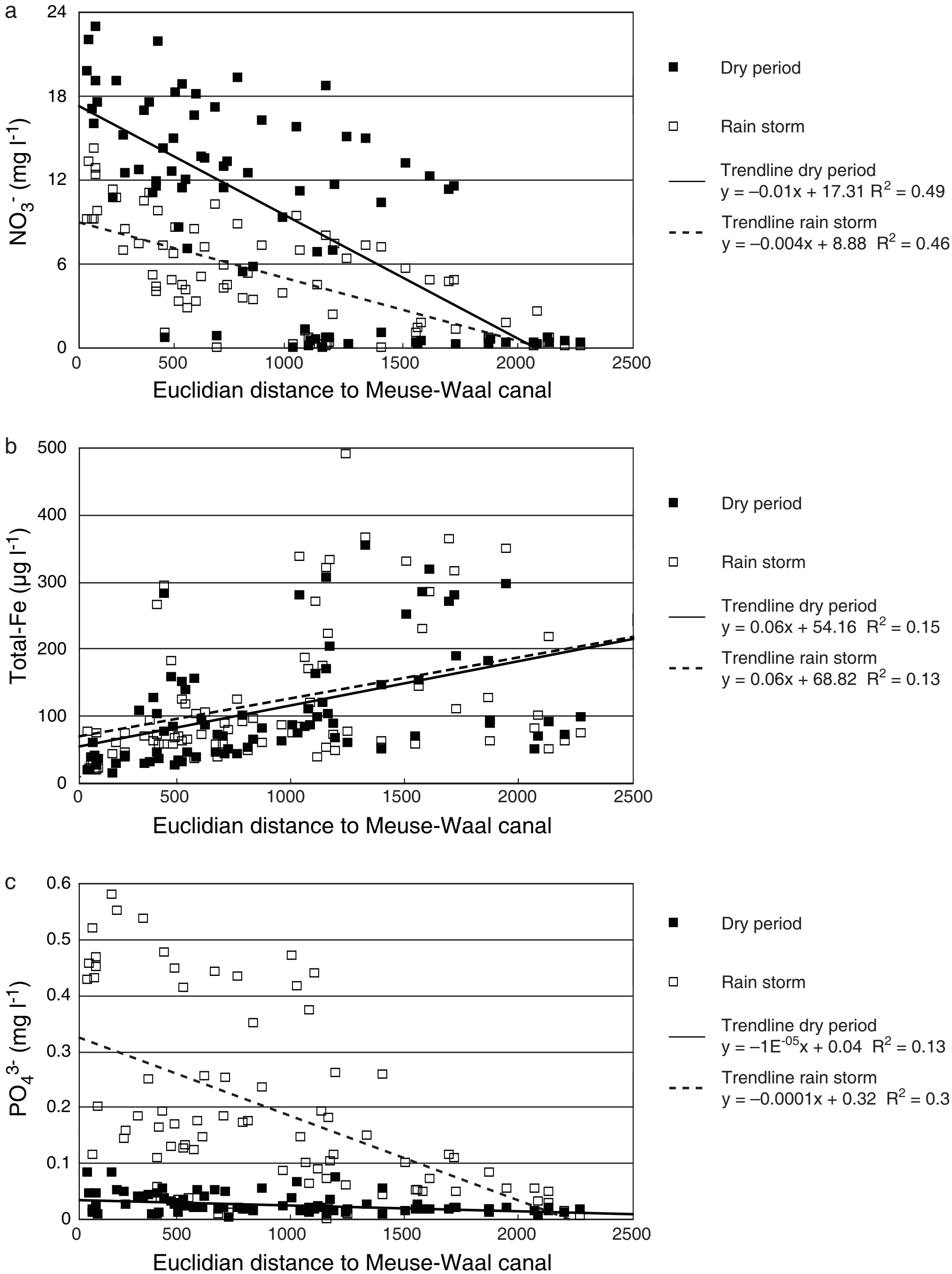

Figure 6 Nitrate (a), iron (b) and phosphate (c) concentrations of surface water in urban drainage systems in relation to Euclidean distance to the Meuse-Waal canal during the dry period and rain storm (n=82).

Nickel and zinc were generally above the maximum allowable concentration (MAC: Ni 5.1 μg l^{-1}, Zn 9.4 μg l^{-1}, Table 3). The maximum concentration of phosphorus and lead exceeded the nutrient standards and MAC in the urban drainage system during the rain storm (nutrient standard: Total-P 0.15 mg l^{-1}, MAC: Pb 11 mg l^{-1}).

Table 3 Average (minimum-maximum) values for water quality parameters in urban drainage systems and in the Meuse-Waal canal in the dry period and during the rain storm.

	Dry period		Rain storm (57 mm)		Nutr. std/ MAC
	Urban drainage system (82)	Meuse-Waal canal (7)	Urban drainage system (82)	Meuse-Waal canal (7)	
NO_3^- (mg l^{-1})	10 (0-26)^a	32(31-33)^b	5 (0-14)^c	15 (13-16)^d	
NH_4^+ (mg l^{-1})	0.1 (0.0-0.2)^{ac}	0.0 (0.0-0.0)^a	0.3 (0.0-1.1)^b	0.1 (0.1-0.1)^c	
Total N (mg l^{-1})	0.5 (0.0-1.3)^a	1.6 (1.6-1.7)^b	0.4 (0.0-0.9)^c	0.8 (0.7-0.9)^d	2.2
PO_4^{3-} (mg l^{-1})	0.1 (0.0-0.3)^a	0.1 (0.1-0.2)^b	0.2 (0.0-0.6)^{b}c	0.2 (0.1-0.2)^c	
Total P (mg l^{-1})	0.01 (0.00-0.05)^{ac}	0.01 (0.00-0.02)^{ab}	0.04 (0.00-0.16)^{bd}	0.02 (0.01-0.02)^{cd}	0.15
Total S (mg l^{-1})	18 (8-22)^a	16 (15-17)^b	16 (8-21)^{b}c	15 (14-15)^c	
K^+ (mg l^{-1})	4.0 (2.0-6.2)^a	3.4 (3.2-3.6)^{a}b	3.7 (2.0-4.8)^b	4.6 (4.6-4.7)^c	
Na^+ (mg l^{-1})	31 (17-41)^a	21 (21-22)^b	28 (13-38)^c	31 (30-32)^{ac}	
Cl^- (mg l^{-1})	49 (30-61)^a	31 (30-34)^b	45 (21-95)^c	41 (40-41)^c	200
pH	7.6 (6.8-8.5)^a	8.4 (8.3-8.5)^b	7.4 (6.9-8.0)^c	8.1 (7.8-8.2)^d	
Alkalinity (meq l^{-1})	2.6 (2.1-3.8)^a	2.6 (2.6-2.7)^b	2.4 (1.4-3.7)^c	2.8 (2.8-2.8)^d	
Ca^{2+} (mg l^{-1})	66 (19-103)^a	67 (63-70)^a	63 (31-95)^a	76 (73-78)^b	
Mg^{2+} (mg l^{-1})	9.1 (2.8-12.2)^a	7.3 (7.0-7.5)^b	8.5 (4.6-11.3)^c	8.5 (8.3-8.7)^{ac}	
Li (μg l^{-1})	2.3 (0.2-5.4)^a	5.2 (5.0-5.5)^b	2.5 (0.4-5.1)^a	8.5 (8.1-8.8)^c	
Total Fe (μg l^{-1})	112 (14-641)^a	36 (23-52)^b	123 (18-489)^a	48 (28-79)^b	
Al (μg l^{-1})	9 (2-79)^a	11 (9-13)^b	13 (2-166)^{ab}	5 (3-9)^a	
Cd (μg l^{-1})	0.0 (0.0-0.4)^a	0.2 (0.1-0.2)^b	0.0 (0.0-0.1)^a	0.1 (0.1-0.2)^b	0.4
Cu (μg l^{-1})	3.7 (1.4-6.9)^a	5.3 (4.2-7.2)^b	2.2 (0.5-7.0)^c	3.4 (3.1-3.7)^a	
Ni (μg l^{-1})	6.2 (2.1-25.5)^a	4.9 (3.8-5.8)^a	5.6 (2.1-36.1)^a	5.8 (4.8-9.3)^a	5.1
Pb (μg l^{-1})	0.1 (0.0-1.0)^a	0.2 (0.2-0.3)^b	1.1 (0.0-14.2)^b	0.1 (0.1-0.2)^a	11
Zn (μg l^{-1})	3 (0-23)^a	4 (3-5)^b	54 (18-208)^c	20 (16-23)^d	9.4

Superscript letters ^{a}, ^{b}, ^{c} and ^{d} indicate significant differences (Mann Whitney test, $p<0.05$) using the same letters for systems that are not significantly different and different letters for systems that are significantly different. Nutrient standards and MAC according to Ministry of Transport, Public Works and Water Management (1998).

Discussion

Upward seepage was modelled by Witteveen+Bos (2006). Although the model was calibrated with 24 piezometers, there could be differences between the modelled upward seepage and the actual upward seepage due to interpolation of model results. In fact, the study area shows large soil heterogeneity due to palaeogeographical and biogeomorphological processes (Berendsen & Stouthamer, 2000). These factors might have decreased the accuracy of correlations between water quality and upward seepage.

Groundwater seepage from the moraine east of Nijmegen and the river dunes west of the study area were not taken into account in this study, because it is expected that the upward seepage from the Meuse-Waal canal overrules the influence of regional groundwater fluxes (Witteveen+Bos, 2006). Soil permeability was used as a proxy for upward seepage and estimated on the basis of geological maps (Pons, 1957). Unfortunately, these maps gave only one soil type for the top-soil (1m depth) on each location. The actual vertical profiles of the soil may consist of various soil types.

Therefore permeability could only be used to confirm relationships with upward seepage and not as an independent parameter.

The ordination of the environmental variables in the urban drainage systems of Nijmegen revealed that the water quality was influenced both by a rain storm and upward seepage from the Meuse-Waal canal (Table 1, Figure 4). The rain storm mainly determined the concentration of ammonium, lead, zinc and phosphorus, while the upward seepage of the Meuse-Waal canal formed the main source for nitrate, potassium, sodium and chloride. Due to the high input from the Meuse-Waal canal, nitrate acts as a redox buffer. This means that at short distance to the canal iron remains oxidized. Under these conditions iron binds phosphate, precipitates and is immobilized in the sediment. With increasing distance from the canal, mineralization of organic material is stimulated by denitrification of nitrate. Denitrification can only take place in anaerobic soil layers that contain reactive organic matter or soils rich in reduced iron and/ or sulphur. If in the subsoil reactive organic matter or reduced iron and/or sulphur is absent, upward welling of nitrate only becomes denitrified once it reaches the organic matter rich upper layer of the sediment. Under these conditions, iron reduces and hence it is mobilized and dissolves in the surface water (Figure 6, Smolders et al., 2006). Furthermore, farther from the canal, by the decomposition of organic material, internal alkalinisation takes place by the production of bicarbonate. This process goes hand in hand with iron reduction and sulphate reduction. During this process alkalinity as well as calcium concentrations increase in the surface water (Table 2, Smolders et al., 2006).

During the rain storm, nitrate levels in the urban drainage systems were lower and ammonium levels higher (Table 3). Apparently nitrate concentrations in the Meuse-Waal canal and thus in the urban drainage systems are diluted during rain storms. On the other hand, ammonium, phosphorus, lead and zinc are delivered to the urban drainage systems during rain storm events. A rain storm introduces high levels of ammonium (i.e. 1.7 mg l^{-1} on average in storm water during the summer of 1999; Stolk, 2001) in the urban drainage systems. The increased ammonium, phosphorus, lead and zinc concentrations after the rain storm could not have been derived from the Meuse-Waal canal as the concentrations in the canal were (much) lower compared to the concentrations in the urban drainage system. Nevertheless, the concentrations of these compounds after the rainstorm were significantly correlated with soil permeability and upward seepage. This suggests that delivery of substances flushed from the impervious areas merely took place via subsurface runoff resulting in higher delivery rates towards surface waters with sandy soils compared to surface waters with clayey soils.

In contrast to other studies (Schueler, 1994, Booth & Jackson, 1997, Walsh et al., 2005) EIA was significantly correlated with only very few environmental variables (i.e. phosphate, pH and iron in the dry period and iron during the rain storm, Table 2). In previous studies approximately 10% EIA caused significant physical and biological effects (Schueler, 1994, Booth & Jackson, 1997). In our study area EIA was approximately 30%, but the water quality was not directly correlated to EIA. The water quality was influenced by the rain storm, but not via the amount of impervious area. Furthermore local upward seepage from the Meuse-Waal canal had a large effect on the water quality of the urban drainage systems. EIA did not correlate with zinc, nickel or lead concentrations. This suggests that the type of impervious area (e.g. roads/ buildings/ material used for gutters) might be much more important for the runoff of metals than the amount of impervious area.

Potassium, sodium and chloride were generally positively correlated to upward seepage, soil permeability and negatively related to distance to the Meuse-Waal canal (Table 2). The potassium, sodium and chloride concentrations were similar in the Meuse-Waal canal and the urban drainage systems as well as during the dry period and the rain storm.

Total nitrogen (0.0-1.7 mg l^{-1}) and phosphorus (0.0-0.6 mg l^{-1}) concentrations were similar to other studies. Walsh et al. (2001) found total nitrogen levels between 0.7-3.7 mg l^{-1} and total phosphorus between 0.03-0.5 mg l^{-1} in urban streams in the Melbourne region, Australia. In an urban stream in North Carolina, United States, total nitrogen concentration was 1.4 mg l^{-1} on average and the total phosphorus concentration 0.1 mg l^{-1} on average (Lenat and Crawford, 1994). Cadmium (0.0-0.4 μg l^{-1}), copper (0.5-7.1 μg l^{-1}) and nickel (2-36 μg l^{-1}) concentrations were lower or similar to average concentrations in the urban streams of North Carolina (Cd: 1.0 μg l^{-1}, Cu: 12.5 μg l^{-1}, Ni: 3.5 μg l^{-1}, Lenat & Crawford, 1994) and lower or similar to concentrations in surface water in the city of Birmingham, United Kingdom (Cd: 0.1-0.6 μg l^{-1}, Cu: 13-25 μg l^{-1}, Ni: 61-224 μg l^{-1}, Shepherd et al., 2006). During the dry period lead (0.0-0.9 μg l^{-1}) and zinc (0.0-23 μg l^{-1}) concentrations were lower than in the studies of Lenat & Crawford (1994): lead 14 μg l^{-1}, zinc 39 μg l^{-1} and Shepherd et al. (2006): lead 1.8-43 μg l^{-1}, zinc 25-101 μg l^{-1}. During the rain storm lead and zinc concentrations were higher or similar to the other studies: 0.0-15 μg l^{-1}, and 18-207 μg l^{-1}, respectively.

Nutrients generally complied with water quality standards (Table 3). Total nitrogen concentrations might be underestimated, because they were calculated from nitrate and ammonium, excluding other nitrogen sources (i.e. organic nitrogen). Phosphorus concentrations were just above the nutrient standards in a few locations during the rain storm. Although nutrients are generally below or close to the water quality standard, they could still have an impact on the flora, for example resulting in shifts from submerged to nymphaeid vegetation or algal blooms (Wetzel, 2001), and should therefore be taken into account when assessing the water quality.

Zinc and nickel concentrations exceeded the MAC, while lead concentrations exceeded the MAC at a few locations during the rain storm. Probably storm water runoff increased the nickel and zinc concentrations in the surface water. Zinc concentrations in storm water also exceeded MAC (i.e. 11.1 μg l^{-1} on average in storm water during the summer of 1999; Stolk, 2001).

In the Netherlands, landscapes have changed drastically after land reclamation (Wolff, 1993). Most of the land is below sea and river water level (Van Stokkom et al., 2005). While rivers are normally fed by the catchment, in this case, river water feeds the lowland areas via the groundwater (Figure 1) and subsequently drainage water is pumped into rivers. The model of Witteveen+Bos (2006), for instance, showed that upward seepage from the Meuse-Waal canal was the main source of water for the urban water systems; 20-41% in the winter period and 62-85% in the summer period. Owing to transnational and regional water pollution, the quality of the rivers Rhine, Meuse and the Meuse-Waal canal do not yet comply with environmental quality standards for nutrients and several toxic substances (Ministry of Transport, Public Works and Water Management, 2005). The river water pollutes the groundwater, and via upward seepage the river water is indirectly introduced in the urban water systems. With increasing sea level rise and precipitation, more and more low lying areas will be influenced by seepage of groundwater. The impact of groundwater seepage on chemistry of urban waters is often underestimated or not taken into account when considering management measures to improve the ecological status of these water bodies. This study shows that upward seepage from polluted and salinated rivers has a negative impact on the water quality of urban drainage systems.

To optimize water quality in urban drainage systems, the first step is to investigate the most important sources of pollution. Investing in expensive measures to clean storm water runoff (e.g. sand filter and soil bank passage) might not be the most cost-effective solution, when other sources play a much more important role in the water quality. In urban drainage systems of lowland areas along large rivers, such as the urbanised districts of Nijmegen, the influence of upward seepage from river water cannot be completely reduced. Therefore the water quality of large rivers and canals should be further improved with respect to nutrients and salinity to diminish pollution of ground water and surface water systems.

Conclusions

Water quality in the urban drainage systems of Nijmegen was both influenced by a rain storm and upward seepage from the Meuse-Waal canal (Figure 4, Table 1). Ammonium, lead, zinc and phosphorus concentrations in the urban drainage systems of Nijmegen were much higher during the rain storm than in the dry period (Table 3). Upward seepage was positively correlated with nitrate, potassium, sodium and chloride and negatively correlated with alkalinity, calcium, magnesium and iron (Table 2). EIA was correlated with very few environmental variables.

Nickel and zinc exceeded the MAC, while lead and phosphorus concentrations exceeded the nutrient standards and MAC only in a few urban drainage systems during the rain storm (Table 3). This case study showed the impact of rivers on local water systems, but further studies are needed to examine the long-term dynamics in these hydrological processes.

To optimize water quality in urban water systems, attention should be paid to all sources of pollution and not only to impervious areas. The impact of local river seepage in lowland areas on the hydrology and chemistry of urban areas is often underestimated and should be taken into account when assessing water quality and improving water quality status. Improvement of regional water quality in lowland areas will not only require local measures to reduce emissions from impervious areas, but also ask for further reduction of transnational river pollution.

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Urban water system Arnhem. Photo: Kim Vermonden

Chapter 3

Species pool versus site limitations of macrophytes in urban waters

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Abstract

Biodiversity in urban areas is affected by a multitude of stressors. In addition to physico-chemical stress factors, the native regional species pool can be greatly reduced in highly urbanized landscapes due to area loss and fragmentation. In this study, we investigated how macrophyte composition and diversity in urban water systems are limited by the regional species pool and local environmental conditions. Canonical correspondence analysis of the macrophyte species composition revealed that urban and semi-natural water systems differed and differences could be related to local abiotic variables such as pH and iron concentrations. Macrophytes in the semi-natural area were typical for slightly acid and oligotrophic conditions. In urban water systems, exotic species characteristic of eutrophic conditions were present. In the semi-natural areas, the number of macrophyte species exceeded the number of species expected from species-area relationships of artificial water bodies in rural areas. In urban areas the number of macrophyte species was similar to artificial water systems in rural areas. Macrophyte species present in the study areas also were generally found within 20-30 km distance to the study area. Macrophyte species composition in urban water systems and semi-natural water systems appeared to be influenced by the regional species pool within approximately 30 km of the locations. Nevertheless, site limitation ultimately determined the local macrophyte species composition and diversity in urban water systems and in semi-natural water systems.

Introduction

Traditionally, patterns of species composition and diversity have been linked to small-scale processes such as geomorphology, hydrochemistry, competition and disturbance (Cornell & Lawton, 1992). In the last two decades, processes at regional and historical scales, including long-distance dispersal, speciation and extinction, have been taken into account as well (Cornell & Lawton, 1992, Zobel, 1997, Ricklefs, 2004). The local species pool often depends on the size and composition of the regional species pool (Caley & Schluter, 1997; Zobel, 1997, Partel & Zobel, 1999).

Biodiversity in urban areas is affected by a multitude of stressors (Paul & Meyer, 2001). In addition to physico-chemical stress factors, the native regional species pool can be greatly reduced in highly urbanized landscapes due to area loss and fragmentation (Collinge, 1996, Drayton & Primack, 1996, McKinney, 2005). Restoration of ecosystems within these landscapes often obtains poor results (Larson et al., 2001, Booth, 2005, Suren & McMurthrie, 2005).

The effects of urbanisation on macrophytes in water systems have not been studied extensively (Paul & Meyer, 2001). Ranta & Toivonen (2008) found that aquatic macrophyte species composition had changed considerably in an increasingly urbanised Finnish lake area during the 20th century, but the number of species remained constant. The number of macrophyte species in Sydney streams was higher in urban systems than in non-urban systems, but the number of native species did not differ (King & Buckney, 2000). Cheruvelil & Soranno (2008) found that the abundance of emergent and nymphaeid vegetation in Michigan lakes was negatively related to road density and urban land use. There is very little information on the limiting factors for aquatic macrophytes in urban areas.

In this study, we investigate how macrophyte composition and diversity in urban waters is related to local environmental conditions and if local macrophyte assemblages are also limited by the regional species pool. The towns of Arnhem and Nijmegen in the Netherlands were chosen as a case study on urban water systems. Multivariate analysis was used to relate macrophyte assemblages in urban and semi-natural water systems to environmental variables. We used a power function to relate local species diversity in urban and semi-natural water systems to the regional species diversity in surrounding rural areas. The species area relationship (SAR) can be used to predict the number of species (species richness) in a certain area (Evans et al., 1955, He & Legendre, 1996), and test if specific areas are low in the number of species (Hamilton et al., 2009).

This study focussed on three research questions:

(1) Does macrophyte species composition and richness in urban drainage systems differ from semi-natural and artificial water systems in surrounding rural areas and can these differences be related to environmental variables?

(2) Does macrophyte species diversity in urban water systems and semi-natural water systems correspond to predicted species diversity using species-area relationships of artificial water systems in surrounding rural areas?

(3) Is species composition in urban water systems and semi-natural water systems limited by the regional species pool and/or local environmental conditions?

Methods

Study area

The municipalities of Nijmegen and Arnhem are situated in the eastern part of the Netherlands along distributaries of the River Rhine (rivers Waal and Nederrijn, respectively) (Figure 1). The municipalities of Nijmegen and Arnhem have approximately 2,803 and 1,414 inhabitants per km^2, respectively (Statistics Netherlands, 2009). In the 1970s, urban water systems were designed in the polder areas of both cities to manage groundwater levels and to drain seepage and storm water into rivers (Vermonden et al., 2009a). Separate sanitary sewer systems were constructed in the study area to transport sewage (Vermonden et al., 2009b). Sewage is pumped directly to the sewage treatment plant and is not discharged into the urban drainage systems. Another pipe system was designed to convey storm water runoff directly to surface waters. The lotic water systems in the study are therefore only fed by stormwater runoff and upward seepage of groundwater. Approximately 4% of the surface area in these cities consists of watercourses that are connected via culverts. The water level is regulated via weirs in the main watercourses. Slow-flowing (current velocity on average 3-14 cm s^{-1}), permanent watercourses can vary from linear ditches to ponds, generally 5-40 m wide and up to 3 m deep. Water systems are mowed once or twice a year depending on vegetation development. Four sites in Nijmegen and 11 sites in Arnhem were dredged 1-3 years before monitoring took place. Land use is predominantly residential, with an impervious area of approximately 30% (roads, buildings, parking lots), and 66% is taken up by gardens, parks and other green areas (Vermonden et al., 2009a).

For this study, 30 water bodies in Nijmegen and 15 in Arnhem were selected and monitored in August-September 2007. Locations varied in morphology, water quality, and vegetation composition. Urban water systems were compared to similar water bodies in a semi-natural area in the immediate vicinity. Since natural water systems do not exist now in the Netherlands, semi-natural water systems in rural areas were used, and were located

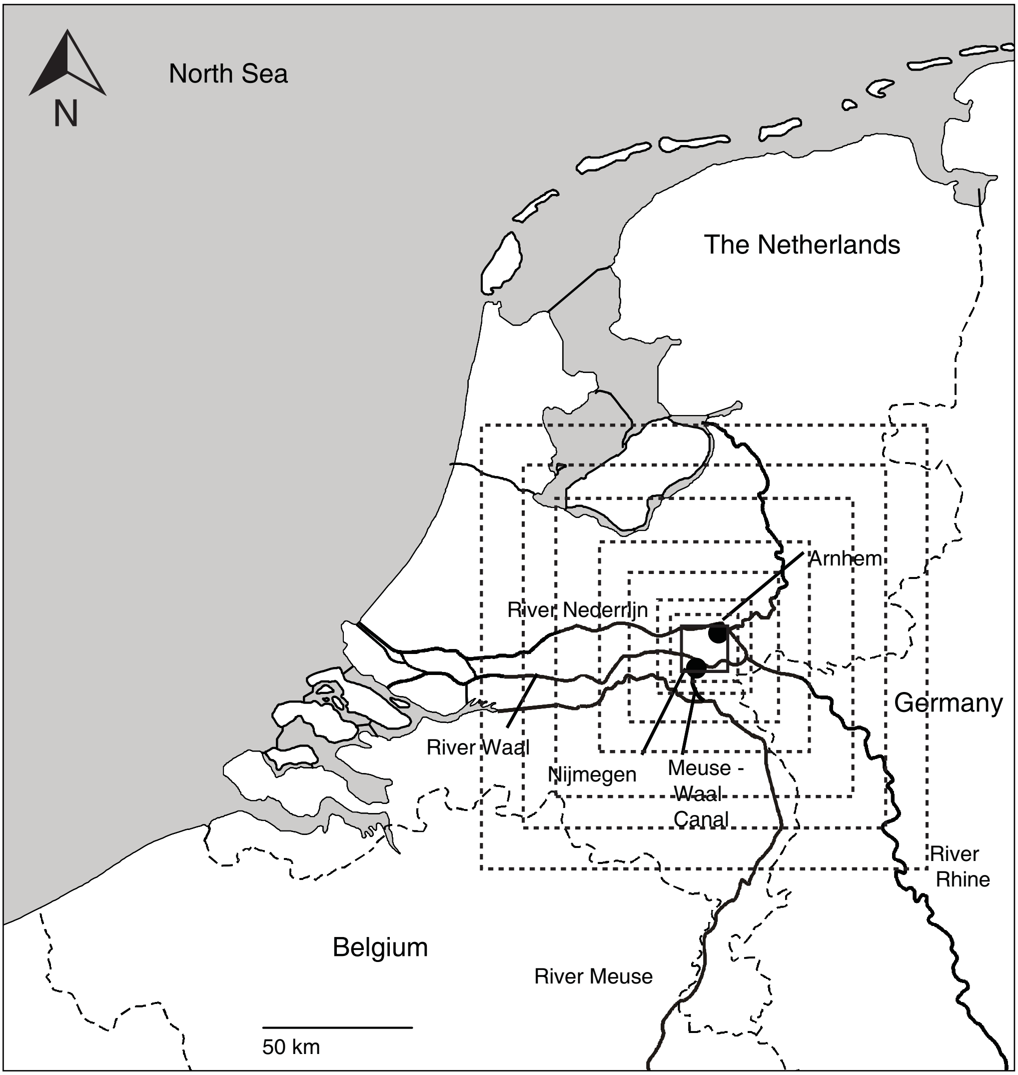

Figure 1 Schematic overview of the study area, with bold square indicating the surface area around Arnhem and Nijmegen and dashed squares indicating the surface areas used to calculate species-area relationships in rural areas.

on the eolian sand area (Figure 2). Monitoring of semi-natural water systems was done in eight locations, 1-3 km SW of Nijmegen in April 2009. Four locations were oligotrophic, slightly acidic ditches or ponds. The other four locations were ditches under the influence of upward seepage of groundwater and therefore less acidic and somewhat enriched with nutrients. Sites were mowed once a year.

To determine the influence of the regional species pool on the local species pool (urban and semi-natural) in the study area, we used the national database of the Dutch waterboards (STOWA, 2008). From this database we developed the regional database by selecting data on artificial water bodies at varying distances from the study area (Figure 1). The national database mostly included ditches (34%), (small) lakes (26%) and lotic water bodies such as small streams and rivulets (19%), and the remaining 21% were rivers,

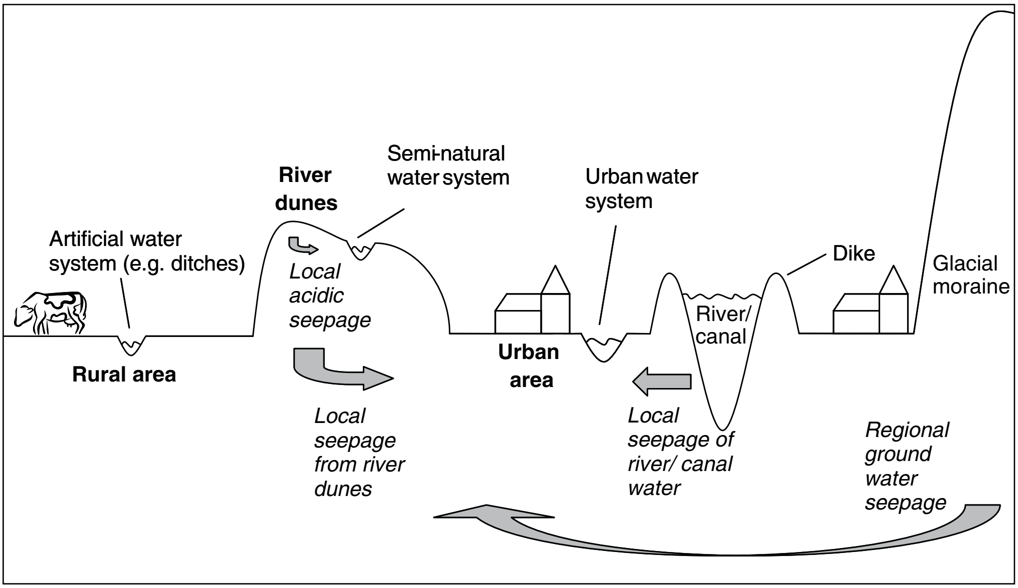

Figure 2 Main groundwater flows of the urban water systems, semi-natural water systems and artificial water systems in rural areas.

softwater lakes, canals and springs (STOWA, 2008). The sampling sites included in the national database were mostly located in rural areas. Data in the national database were collected in the period 2000-2006.

Macrophytes

Macrophyte species composition was recorded on an abundance scale from 1 to 9, corresponding to classes of the Tansley scale (Tansley, 1946): sporadic, rare, occasional, locally frequent, frequent, locally abundant, abundant, co-dominant, dominant, respectively. Hydrophytes and helophytes were monitored for a stretch of approximately 20 metres along each sampling location. Hydrophytes were defined as aquatic plants living in or on the surface water (Antoine et al., 2004). Hydrophytes were observed in the surface water within two to three metres of the bank. Helophytes were rooted in the wet soil of the banks or in the sediment just below the surface water with shoots in the air (Antoine et al., 2004). Marsh and other plants that were associated with the aquatic environment were also included as helophytes, but plants associated with the terrestrial environment were excluded from monitoring.

Environmental variables

Two surface water samples were taken at each location in the urban and semi-natural water systems. The pH and alkalinity were measured on the following day, after samples had been stored overnight at 4°C. Total Inorganic Carbon (TIC) was measured with an ABB Advance Optima Infrared Gasanalyser, and CO_2, HCO_3^- and CO_3^{2-} were calculated from TIC, pH, and temperature. Water samples were stored at -20°C until further analysis, after adding citric acid (125 mg L^{-1}). The following ions were measured colorimetrically (Auto Analyzer 3, Digital colorimeter, Bran + Luebbe, Germany): NO_3^- according to Kamphake et al. (1967), NH_4^+ according to Grasshoff & Johannsen (1972), Cl^- according to O'Brien (1962) and PO_4^{3-} according to Henriksen (1965). Na^+ and K^+ were measured photometrically with a flame photometer (Radiometer, Copenhagen). Metals, Total-S, Total-P, Ca^{2+} and Mg^{2+} were measured by inductively coupled plasma mass spectrometry (Thermo Electron Corporation, United Kingdom). All physico-chemical factors were measured at least in duplicate and average values were used for data analysis. Altitude was determined using an interactive altitude map (www.ahn.nl).

Data analysis

Canoco for Windows Version 4.0 (Ter Braak & Šmilauer, 1998) was used to perform Canonical correspondence analysis (CCA) in order to relate macrophyte assemblages in urban and semi-natural water systems to environmental variables. Rural waters were not included in CCA, because data on environmental variables of these waters was not available. CCA is a direct ordination method incorporating linear correlations and regressions between species data and environmental variables. The ordination axes are a result from the joint variation in species and environmental data (Jongman et al., 1995). A unimodal response model was selected, because there was a wide range in environmental variables. Environmental variables with log-normal distributions (NH_4^+, total Fe and Al^{3+}) were log-transformed. Significance of environmental variables was tested with CCA, using 500 Monte Carlo permutations under full model conditions.

Differences of environmental variables between urban and semi-natural water systems were tested with a Mann-Whitney test. Species composition of urban and semi-natural water systems was also compared with artificial water bodies in rural areas. In the national database of the water boards (STOWA, 2008) 36 locations were selected within a 500 km² area around Arnhem and Nijmegen and macrophyte species composition and abundance were determined.

Total macrophyte diversity in urban and semi-natural water systems was compared with macrophyte diversity in artificial water bodies in rural areas (STOWA, 2008). A power function was used to relate number of species (S) with rural area (A).

$$S = cA^z$$

where c is a constant and z the scaling exponent (Preston, 1960, MacArthur & Wilson, 1967, Rosenzweig, 1995).

The number of macrophyte species in the rural area was calculated for eight concentric squares with Arnhem and Nijmegen as the centre, with an increasing area of 250, 500, 975, 2,285, 4,416, 8,680, 12,698 and 17,841 km² (Figure 1). These areas included both terrestrial and wet areas, but Belgium and Germany were excluded from these surface areas. A linear regression was fitted to log-transformed species number and area data. From the linear regression, the expected number of species was calculated for the urban and semi-natural area, with 95% confidence limits. The surface area of the urban and semi-natural area was calculated by drawing a square around all urban water systems and semi-natural water systems (total urban area: 273 km², total semi-natural area 7.5 km², Figure 1).

Additionally, average macrophyte species richness was compared between the 45 urban water systems, eight semi-natural water systems and 36 artificial water bodies in rural areas within 500 km² around Arnhem and Nijmegen with a Mann-Whitney test ($p<0.05$).

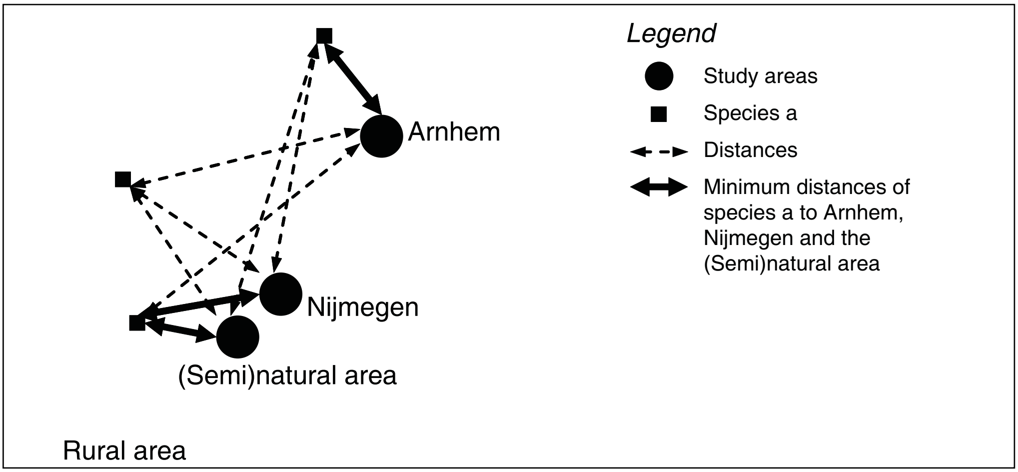

Figure 3 Schematic overview of the calculation of minimum distances.

For each species, the minimum distance was calculated to the location closest to the study areas where the species was found in artificial water bodies in rural areas (STOWA, 2008, Figure 3). The minimum distances of species present and absent in the study area were compared with a Mann-Whitney test ($p<0.05$).

Results

Species composition and local environmental factors

Canonical correspondence analysis (CCA) clearly separated helophyte assemblages of semi-natural water systems from urban water systems (Figure 4a). The pH, alkalinity, CO_2, altitude, NO_3^-, total-Fe and Zn^{2+} concentrations were significantly correlated to variation in helophyte assemblages. The distinction of urban and semi-natural water

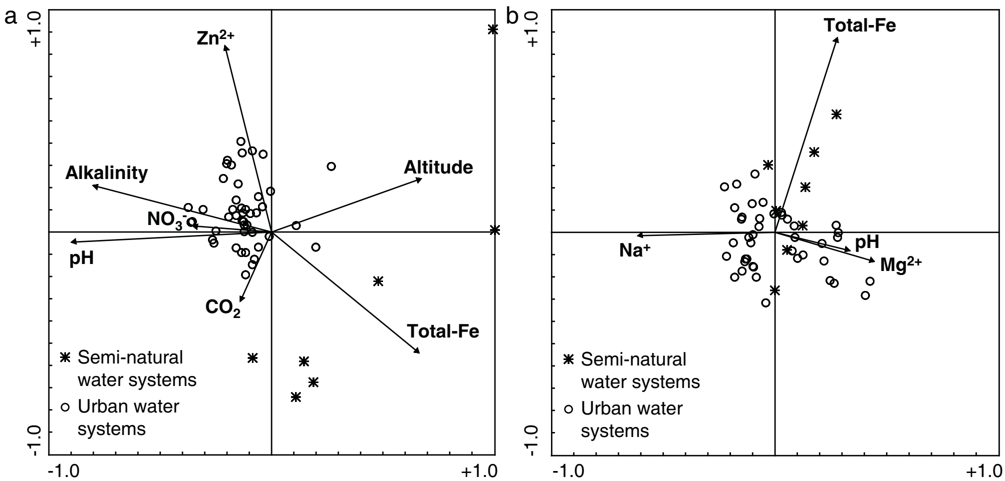

Figure 4 Canonical correspondence analysis of helophyte (a) and hydrophyte (b) species in semi-natural water systems and urban water systems. Circles and stars represent locations, while arrows represent environmental variables explaining a significant proportion of variation in helophyte and hydrophyte assemblages between locations.

systems was less clear for hydrophyte assemblages according to CCA (Figure 4b). The pH, Mg^{2+}, Na^+ and total-Fe were significantly related to variation in hydrophyte assemblages.

Nutrient concentrations (NO_3^-, PO_4^{3-}, total-S) were significantly higher in urban water systems than in semi-natural water systems (Table 1). The pH, alkalinity, HCO_3^- and Ca^{2+} were also significantly higher in urban water systems, while total-Fe and Al^{3+} concentrations were significantly higher in semi-natural water systems. Zn^{2+} concentrations were highest in urban water systems. On average semi-natural water systems were located at significantly higher altitude than urban water systems.

Table 1 Average values of environmental variables (minimum and maximum values in brackets). Significant differences between urban water systems and semi-natural water systems in bold ($p<0.05$, Mann-Whitney test).

	Urban (n=45)	Semi-natural (n=8)
NO_3^- (µmol l^{-1})	**64 (1-446)**	**16 (0-76)**
NH_4^+ (µmol l^{-1})	85 (1-4544)	9 (0-851)
PO_4^{3-} (µmol l^{-1})	**1.5 (0.1-5.9)**	**0.4 (0.1-0.9)**
Total-P (µmol l^{-1})	1.5 (0.3-5.9)	1.2 (0.3-2.8)
Total-S (µmol l^{-1})	**441 (17-626)**	**283 (12-474)**
K^+ (µmol l^{-1})	**102 (15-236)**	**64 (17-150)**
Na^+ (µmol l^{-1})	**1157 (63-1811)**	**737 (210-1638)**
Cl^- (µmol l^{-1})	1125 (109-1674)	848 (111-1938)
pH	**7.3 (6.6-8.0)**	**6.5 (5.1-7.8)**
Alkalinity (meq l^{-1})	**2.91 (1.40-4.55)**	**1.19 (0.07-2.07)**
CO_2 (µmol l^{-1})	314 (75-697)	297 (39-1034)
HCO_3^- (µmol l^{-1})	**2474 (190-4548)**	**1030 (4-2073)**
Ca^{2+} (µmol l^{-1})	**1438 (97-2443)**	**726 (18-1271)**
Mg^{2+} (µmol l^{-1})	331 (11-511)	225 (17-406)
Total-Fe (µmol $^{-1}$)	**1 (0-60)**	**156 (19-1432)**
Al^{3+} (µmol l^{-1})	**0.1 (0.0-0.9)**	**2.3 (0.1-7.5)**
Zn^{2+} (µmol l^{-1})	**0.6 (0.3-1.2)**	**0.2 (0.0-1.3)**
Altitude (m above NAP)	**6.9 (5.9-9.8)**	**8.0 (7.1-10.7)**

Epilobium hirsutum, *Persicaria amphibia*, *Lythrum salicaria*, *Ceratophyllum demersum* and *Hydrocharis morsus-ranae* were abundant in urban artificial water systems and rural areas, but absent in semi-natural water systems (Table 2). *Lotus penduncculatus*, *Carex acuta*, *Galium palustre* and *Lysimachia vulgaris* were most abundant in urban and semi-natural water systems. The following species were found exclusively in semi-natural water systems: *Pilularia globulifera*, *Potentilla palustris*, *Ranunculus repens*, *Sphagnum fallax*, *Carex rostrata*, *Ranunculus flammula*, *Cardamine pratensis*, *Agrostis stolonifera*, *Potamogeton natans* and *Potamogeton polygonifolius*. However, *Ranunculus repens*, *Cardamine pratensis* and *Agrostis stolonifera* can also occur in grasslands around urban waters.

The exotic species *Hydrocotyle ranunculoides*, *Ludwigia grandiflora*, *Pontederia cordata* and *Lemna minuta* appeared exclusively in urban water systems (Table 2), while *Azolla filiculoides* was only present in artificial water bodies in rural areas. Each exotic species was found in < 7% of the sampling locations, with the exception of *Lemna minuta*, which was present in 47% of the sampling locations in the urban area.

Table 2 Presence (%) and average abundance of characteristic helophytes and hydrophytes in urban, rural and semi-natural areas. Presence is expressed as percentage of locations where a species occurred and abundance on Tansley scale (0-9) (**Exotic species** in bold, defined according to Van der Velde et al., 2002; native area between brackets).

	Urban (n=45)		Artificial water bodies in rural area (n=36)		Semi-natural water systems in rural area (n=8)	
	Presence	Abundance	Presence	Abundance	Presence	Abundance
Helophytes						
Hydrocotyle ranunculoides (NSA)	7	6	0	0	0	0
Ludwigia grandiflora (tropics, SA)	2	3	0	0	0	0
Pontederia cordata (NSA)	2	2	0	0	0	0
Epilobium hirsutum	51	4	34	2	0	0
Persicaria amphibia	47	4	37	2	0	0
Lythrum salicaria	31	4	11	2	0	0
Mentha aquatica	24	5	9	4	0	0
Symphytum officinale	13	3	23	2	0	0
Filipendula ulmaria	9	3	29	2	50	4
Alisma plantago-aquatica	2	3	26	2	50	3
Equisetum fluviatile	0	0	14	4	38	4
Lotus pedunculatus	44	4	3	3	38	3
Carex acuta	27	7	9	3	50	5
Galium palustre	22	4	0	0	63	3
Lysimachia vulgaris	18	4	0	0	88	3
Eleocharis palustris	2	5	6	6	50	5
Juncus articulatus	4	4	0	0	38	3
Pilularia globulifera	0	0	0	0	25	6
Potentilla palustris	0	0	0	0	25	3
Ranunculus repens	0	0	0	0	25	2
Sphagnum fallax	0	0	0	0	25	4
Carex rostrata	0	0	0	0	38	6
Ranunculus flammula	0	0	0	0	50	3
Cardamine pratensis	0	0	0	0	63	2
Agrostis stolonifera	0	0	0	0	75	6
Hydrophytes						
Lemna minuta (NA)	47	6	0	0	0	0
Lemna gibba	7	3	0	0	0	0
Ceratophyllum demersum	27	5	16	2	0	0
Hydrocharis morsus-ranae	11	5	19	4	0	0
Spirodela polyrhiza	11	2	47	3	0	0
Enteromorpha sp.	9	3	13	3	0	0
Nuphar lutea	4	8	16	4	0	0
Azolla filiculoides (NA)	0	0	3	2	0	0
Fontinalis antipyretica	0	0	9	2	0	0
Sparganium emersum	4	4	6	5	25	2
Lemna trisulca	9	4	19	2	38	4
Potamogeton natans	0	0	0	0	38	4
Potamogeton polygonifolius	0	0	0	0	13	7

NSA = North/ South America, NA = North America, SA = South America

Macrophyte species diversity and species pool limitations

Log-transformed numbers of helophyte and hydrophyte species were linearly correlated to the log-transformed area of the rural environment ($p<0.05$, $R^2 = 0.97-0.99$, Figure 5). In the semi-natural area, the number of helophyte species exceeded the number of species expected from linear species-area regressions of rural areas (Figure 5). The number of hydrophyte species was within the expected 95% confidence interval. The number of helophyte and hydrophyte species in urban areas exceeded the expected number of species, but the value for hydrophyte species was within the expected 95% confidence interval. Average helophyte species richness was significantly higher in semi-natural water systems, than in urban and artificial water systems (Figure 6). Average hydrophyte species richness was not significantly different between groups.

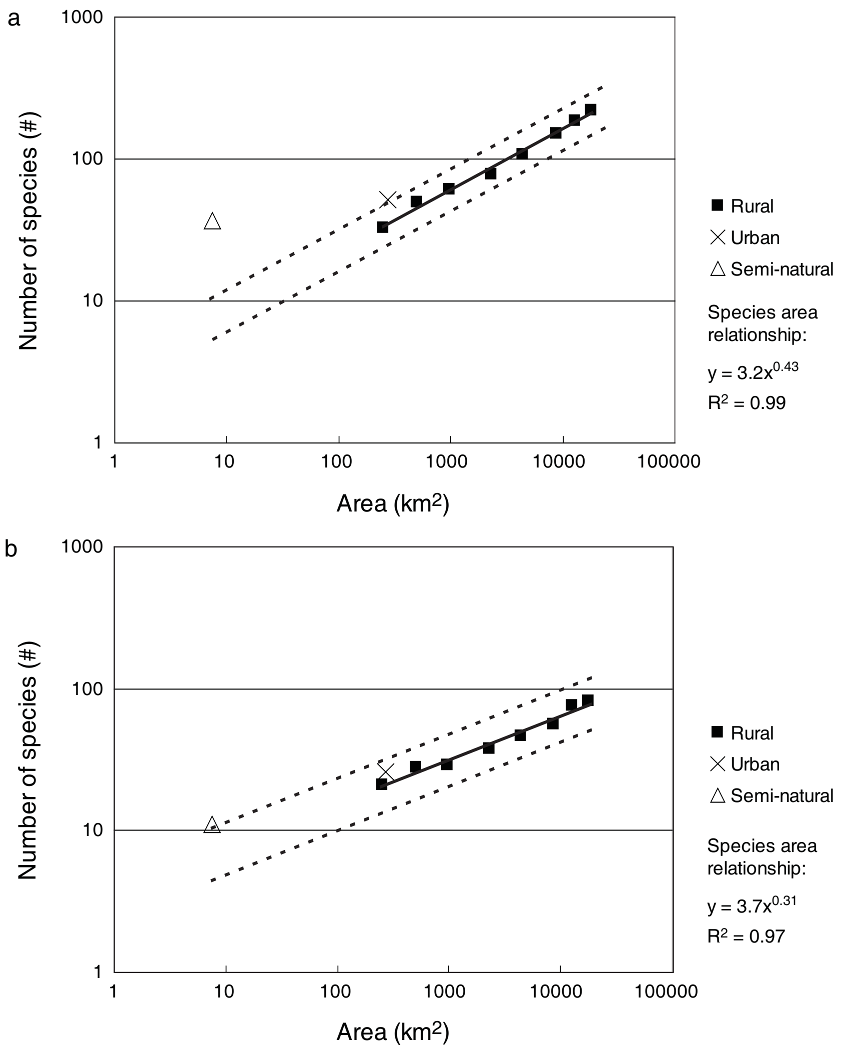

Figure 5 Relationship between number helophyte (a) and hydrophyte (b) species and surface areas. Dotted lines indicate confidence intervals of species-area relationship in rural areas (STOWA, 2008).

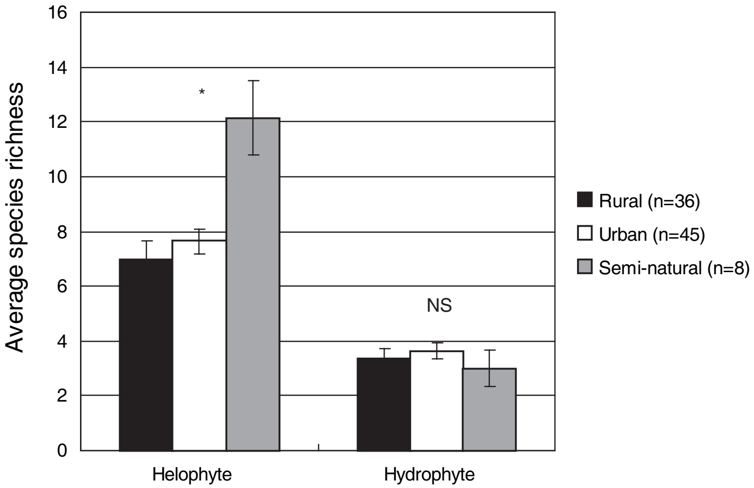

Figure 6 Average species richness in urban, rural and semi-natural water systems. Significant difference is indicated with asterisk (Mann-Whitney test, $p<0.05$).).

The minimum distance of macrophyte species present in the study areas was significantly lower than the minimum distance of macrophyte species absent in the study areas (Figure 7). Helophyte species present in the study areas were generally also found within 30 km distance to the study area, whereas hydrophyte species that occurred in the study area were found within approximately 20 km distance of the study areas.

Discussion

Species composition and site limitation

The CCA of the helophytes distinguished the semi-natural water systems more clearly than that of the hydrophytes (Figure 4). The pH and iron levels significantly accounted for the variation in both helophyte and hydrophyte composition. Upward seepage of iron-rich groundwater reduced nutrient availability in the semi-natural area and thereby changed species composition (Lucassen et al., 2006). Urban water systems were enriched in nutrients compared to the semi-natural area, which is consistent with other studies (e.g. Paul & Meyer, 2001, Walsh et al., 2005). The pH, alkalinity, HCO_3^- and Ca^{2+} were significantly lower in the semi-natural water systems than in urban water systems. This could be related to the influence of local groundwater, which introduced acidic water in the semi-natural area with a pH between 3.9 and 5.2 (Lucassen & Smolders, 2008). Iron and aluminium diffuse from the sediment at low pH (Wetzel, 2001) in the semi-natural water systems. Higher zinc concentrations in the urban water systems were related to storm-water run-off from impervious areas (Paul & Meyer, 2001, Brabec et al., 2002, Vermonden et al., 2009b).

Most species occurred in both urban water systems and artifical water bodies in rural areas (Table 2). Nine species were exclusively present in semi-natural water systems, among which *Pilularia globulifera*, *Potentilla palustris*, *Sphagnum fallax*, *Carex rostrata*, *Ranunculus flammula*, and the hydrophytes *Potamogeton natans* and *Potamogeton polygonifolius* were typical for slightly acid, oligotrophic waters (De Lyon & Roelofs, 1986).

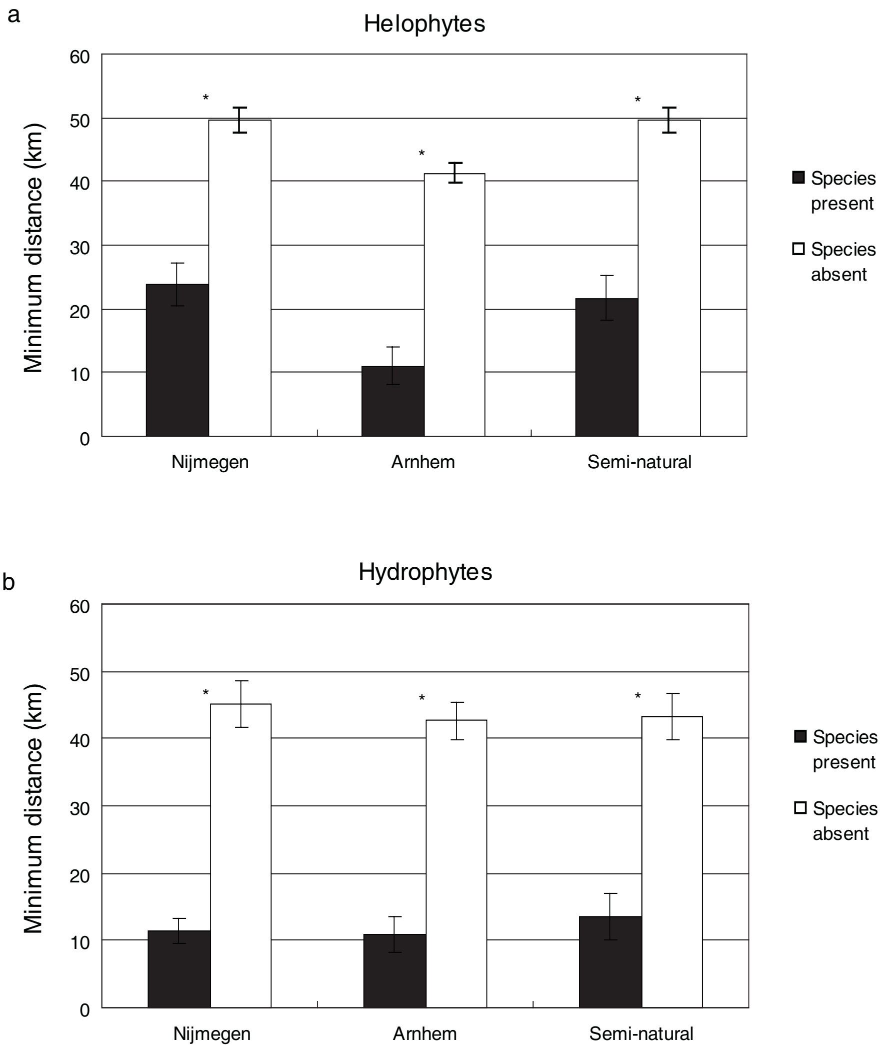

Figure 7 Average distances to nearest location where helophyte (a) and hydrophyte (b) species were found in the national database (STOWA, 2008). Averages calculated for species present and absent in the urban water systems of Nijmegen, Arnhem and semi-natural water systems. Significant differences are indicated with asterisk (Mann-Whitney test, p<0.05).

Four out of five exotic species in this study appeared exclusively in urban water systems, the other one was present exclusively in artificial water bodies in rural areas (Table 2). No exotic species were present in the semi-natural water systems. Exotic species also appeared exclusively in urban streams in the northern Sydney region, which could be related to higher nutrient levels in urban sediments (King & Buckney, 2000). Kercher & Zedler (2004) also suggested that invasive macrophyte species could easily dominate native species where nutrient availability is high. Ehrenfeld (2008) related invasive species specifically to residential areas.

Propagule pressure of exotic species is high in urban areas, because of dispersal from ponds in gardens and parks and human introductions (e.g. by dumping of plant material from aquaria and ponds). *Ludwigia grandiflora* and *Pontederia cordata* are sold as garden plants in the Netherlands and *Hydrocotyle ranunculoides* was sold until 2001. Anthropogenic disturbance, such as vegetation removal, also contributes to plant invasions (Detenbeck et al., 1999). The combined influence of eutrophication, propagule pressure and disturbance make the urban waters vulnerable for species invasions. Besides the exotic species, only *Lemna gibba* occurred exclusively in the urban water systems. *Lemna gibba* is associated with eutrophic water (Papastergiadou & Babalonas, 1993). The presence of lemnids in the urban water systems can also be associated with stagnant water systems and muddy sediment (Boedeltje et al., 2005). Water systems in the rural and semi-natural area could have higher flow velocities and could therefore be less suitable for the growth of lemnids.

Macrophyte species diversity

Although dredging and mowing can reduce macrophyte species richness temporarily, the local species pool was probably unaffected because all water systems were interconnected and species could disperse very quickly through the systems. Moreover, mowing and dredging activities were mainly performed in the middle of the ditches to guarantee discharge capacity, and vegetation near the banks was partly left intact according to nature-oriented mowing and dredging schemes.

Log-transformed numbers of helophyte and hydrophyte species in artificial water bodies in rural areas were strongly correlated to log-transformed area (Figure 5, $R^2 = 0.99$ and 0.97, respectively). The slope of species-area relationships (z) depends on the scale, habitat diversity and taxonomic groups included in the analysis (Wright, 1981, Rosenzweig, 1995, Ricklefs & Lovette, 1999, Crawley & Harral, 2001, Koh et al., 2002, Turner & Tjørve, 2005, Drakare et al., 2006, Kallimanis et al., 2008). Crawley & Harral (2001) showed that z values for plants were in the range of 0.1-0.2 at small scales (<100 m^2), 0.4-0.6 at intermediate scales (10,000 m^2 - 10 km^2) and 0.1-0.4 at the largest scale (10-1,000 km^2). Our species-area relationships had z-values of 0.43 (helophytes) and 0.31 (hydrophytes) at scales of 250-17,841 km^2. Slopes are at the upper end of the range noted by Crawley & Harral (2001).

Species-area relationships of different taxonomic groups also result in different slopes, as was shown for very different groups such as plants, springtails, butterflies, reptiles, amphibians, mammals and birds (Wright, 1981, Ricklefs & Lovette, 1999, Koh et al., 2002). Koh et al., (2002) found that species with higher dispersal ability were related to lower z-values because they were less sensitive to the area effect than species with low dispersal ability. Our study showed that even within highly related groups such as helophytes and hydrophytes, there can be differences in species-area relationships. Ricklefs & Lovette (1999) suggested that habitat specialization would make species richness more sensitive to habitat diversity and less sensitive to area.

The number of helophyte species is somewhat (urban) and substantially larger (semi-natural) in comparison to the richness-area regression for rural regions. By contrast, hydrophyte richness is within (urban) and just above (semi-natural) the 95%-confidence interval noted for rural areas. Comparisons of average species richness agreed with these results. Hydrophytes show broad distribution ranges and limited taxonomic differentiation because they have to be adapted to a stressful environment, characterised by low carbon availability, shaded conditions, sediment anoxia and wave exposure

(Santamaría, 2002). This explains why we found fewer hydrophyte species than helophyte species and less differentiation between habitat types. This could be an additional explanation why *z* values in species-area relationships are lower for hydrophytes than for helophytes. Next to that, hydrophytes might be influenced more by eutrophication of water bodies than helophytes.

Average helophyte species richness per site was significantly higher in semi-natural water systems than in urban and rural waters (Figure 6). Artificial water systems in urban and rural areas sustain fewer species because these water systems are eutrophic and often disturbed. Other studies also document a decrease in macrophyte species richness with increased disturbance and associated eutrophication (Riis & Sand-Jensen, 2001, Egertson et al., 2004, Lougheed et al., 2008).

Some studies showed that the size of individual water systems is positively related to species richness (e.g. Barbour & Brown, 1974, Dodson, 1992). Other studies did not show a significant relationship between aquatic plant diversity and lake size, especially in temperate, small to medium-sized lakes (e.g. Declerck et al., 2005, Kruk et al., 2009). In our study, the species richness of macrophytes were not significantly related to the size of the individual ditches and ponds. On average, the sizes of water systems in urban, rural and semi-natural water systems were very similar and individual water systems were also interconnected. The size of individual water systems was therefore considered to be a minor factor affecting macrophyte species richness in our study area.

Species pool versus site limitation

Species present in the urban and semi-natural area were also present within relatively short distances in the regional species pool of artificial water bodies in rural areas (Figure 7). Helophyte species were, on average, found between 5 and 30 kilometres from the monitored sites and hydrophyte species were found between 5 and 20 kilometres from the monitored sites. In contrast, the nearest growing site of species that were not recorded in our surveys was, on average, more than 40 kilometres away. This could imply that the local species pool may be limited by the regional species pool, corresponding with the species pool hypothesis (Caley & Schluter, 1997, Zobel, 1997, Partel & Zobel, 1999). Abiotic conditions might also be more similar within a short distance and therefore result in a more similar species composition.

Macrophyte species composition and diversity were also strongly dependent on local abiotic processes (Figures 4, 6). The available data did not allow quantification of variation in species composition attributable to site characteristics versus the regional species pool. Quantification of this variation may be possible with additional multivariate analysis, but will require data on species pools of individual sites. Large-scale processes such as dispersal determine how many and which species are available for the local community (Zobel, 1997). Our study shows that the actual local species composition in urban water systems and semi-natural water systems was determined by the ability of species to cope with local biotic and abiotic circumstances.

Conclusions

Macrophyte species composition in urban and semi-natural water systems differed and could be related to local abiotic variables, such as pH and iron concentrations. In urban water systems, exotic species typical of eutrophic conditions were present. In semi-natural water systems, exotic species were absent and indicators for slightly acidic and oligotrophic conditions were present. Macrophyte species composition in the urban water systems and semi-natural water systems also appeared to be influenced by the regional species pool located within approximately 30 km of the sites. Nevertheless, site limitation eventually determined the local macrophyte composition and diversity in urban water systems and semi-natural water systems. This is illustrated by the higher species richness in the semi-natural water systems in the immediate vicinity of the urban areas.

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Urban water system Arnhem. Photo: Kim Vermonden

Chapter 4

Urban drainage systems: An undervalued habitat for aquatic macroinvertebrates

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Abstract

Knowledge about the ecology of urban water systems is very scarce. We assessed the conservation value of urban drainage systems in lowland areas and compared these with similar watercourses in rural areas. A total of 36 water bodies in urban areas were selected to investigate the macroinvertebrate biodiversity in relation to environmental variables. Multivariate analysis of aquatic macroinvertebrate assemblages was used to distinguish urban water types and to link these types to key environmental variables. Several biodiversity indices for urban water systems were compared with those for other drainage systems in The Netherlands. Four types of macroinvertebrate assemblages were distinguished in the urban water systems, differing in environmental conditions and values of ecological indicators. The variation in macroinvertebrate assemblages was significantly explained by nitrate, pH, grain size (sediment composition), transparency, nymphaeid and submerged vegetation. Urban drainage systems can sustain a macroinvertebrate biodiversity comparable to that of drainage systems in rural areas (ditches and canals) and (semi)natural watercourses (lotic waters such as small streams and rivulets) and can even be a habitat for red list species. To optimize biodiversity values, urban water management should aim at lowering nutrient levels, stimulating vegetation (diversity of habitat structure) and increasing transparency, which are key factors for macroinvertebrate diversity. We show the potential conservation benefits of water systems in urban areas, but further studies are needed to investigate the optimal design of cities to include biodiversity as an integrated part of the urban environment, thereby sustaining a higher biodiversity in an increasingly urbanizing world.

Introduction

Although urban areas cover less than 3% of the earth's land surface, towns and cities play an important role in biodiversity changes (Grimm et al., 2008). Urbanization leads to ecosystem destruction, habitat fragmentation and species extinction (Malmqvist & Rundle, 2002, McKinney, 2006, Grimm et al., 2008). Hence, restoration, preservation and enhancement of biodiversity in urban areas are becoming more and more important (Savard et al., 2000). Urban waters are regarded as attractive for citizens and are therefore given a more prominent place in newly designed suburbs. A novel opportunity could be to design them so as to provide vital ecological services such as biodiversity (Palmer et al., 2004, Wang et al., 2006). In increasingly fragmented landscapes, urban areas can play an important role as greenways, providing habitat for flora and fauna, and connecting natural areas (Bryant, 2006). High quality habitats for species dispersing from one natural area to another should become increasingly available in urban areas. With the prospect of climatic changes to come, the potential availability of quality stepping stones in towns and cities seems in this light an opportunity we cannot afford to miss. At present, there is a lack of data to assess the conservation value of urban areas.

Aquatic ecosystems in urban areas differ in many ways from natural ones, for example in hydrology, morphology, water chemistry and the composition of flora and fauna (Ehrenfeld, 2000, Paul & Meyer, 2001, Walsh et al., 2005). Hydrology of urban water often shows higher and more frequent peak discharges due to fast run-off from impervious areas, while water levels are kept constant by artificial means (e.g. damming up and pumping). Natural water systems in urban areas are often canalized, which changes their morphology to wider, deeper and less complex systems. Nutrient and contaminant loadings are usually enhanced. Flora and fauna diversity generally declines, as tolerant species increase while sensitive species decrease or disappear (Paul & Meyer, 2001, Walsh et al., 2005).

Although many studies have investigated the influence of urbanization on water systems (Lenat & Crawford, 1994, Wear et al., 1998, Paul & Meyer, 2001, Roy et al., 2003, Booth et al., 2004, Miller & Boulton, 2005), very few have focused specifically on urban water systems as a habitat for flora and fauna (Heckman, 1982, Girgin et al., 2003). Paul & Meyer (2001) stressed the importance of examining the ecology of urban streams and the challenge to integrate physical, chemical and biological processes in impaired systems.

Throughout history, cities have preferentially developed along rivers and deltas (Grimm et al., 2008). In the Rhine-Meuse river basins, for instance, urban and industrial areas account for 25.7% of the catchments, and average population density is 319 people per km^2 (World Resource Institute, 2003). Urbanization is especially high in the lowland areas of the river catchments, viz. The Netherlands, Belgium and Germany. Lowland area is defined as low-lying land reclaimed in river floodplains (alluvial plains) and deltas. In The Netherlands, drainage is essential, because half of the land is below sea and river level (Waterman et al., 1998). Intensive drainage networks (ditches, canals, ponds) collect storm water and groundwater (Hiscock et al., 2001, Nguyen & Sukias, 2002, Krause et al., 2007), and an important part of the drainage systems in The Netherlands consists of urban water systems.

This study focuses on the biodiversity value of urban water systems in lowland areas in comparison with man-made drainage systems in rural areas (ditches and canals) and natural or seminatural watercourses (lotic waters such as small streams and rivulets) in forests or other natural or seminatural areas).

The following hypotheses were tested:

(1) The conservation value of optimally managed urban water systems for macroinvertebrates is comparable to that of similar man-made drainage systems in rural areas and natural and seminatural watercourses.

(2) Stimulating habitat structure and implementing pollution control measures are key factors for the conservation of macroinvertebrates in urban water systems.

These hypotheses resulted in the following research questions:

(1) Which urban water types can be distinguished based on aquatic macroinvertebrate assemblages?

(2) Are there characteristic environmental conditions related to these types of urban water bodies?

(3) Can the urban water body types be characterized by ecological indicators such as characteristic species, taxa richness, Shannon index, number of red list species, exotic species and rareness of species, and can these indicators be related to key environmental factors?

(4) What is the contribution of urban water systems to the macroinvertebrate species richness in The Netherlands?

(5) Are the values of the ecological indicators in urban water bodies different from those in similar man-made drainage systems in rural areas and natural or seminatural watercourses?

The towns of Nijmegen and Arnhem (The Netherlands) were selected to investigate urban water systems in lowland areas. Macroinvertebrates were used to determine the importance of urban waters for biodiversity, because of their high abundance, high species diversity and varying sensitivity of species to changing environmental conditions, as well as the fact that they are relatively easy to catch (Metcalfe, 1989). Moreover, invertebrates are a key element of ecosystem processes in the water system, as they process organic material from both autochthonous and terrestrial sources (Vannote et al., 1980) and they form the basic food chain for many species of fish, birds and other wildlife present in towns and cities. Multivariate analysis was used to identify various types of macroinvertebrate assemblages associated with water types and the key environmental variables for biological conservation. Ecological indicators were used to characterize the assemblages in the various water bodies and estimate the conservation value of urban water systems.

Material and methods

Study area

The municipalities of Nijmegen and Arnhem are situated in the eastern part of The Netherlands along two distributaries of the River Rhine (Rivers Waal and Nederrijn, respectively) (Figure 1). Both towns have approximately 150,000 inhabitants and the surface areas of the municipalities of Nijmegen and Arnhem are 72 km^2 and 102 km^2, respectively (Statistics Netherlands, 2006). In the 1970s, urban water systems were designed to regulate groundwater levels and to manage the discharge of water into the rivers. Approximately 4% of the surface area of these towns consists of watercourses that are connected via culverts. The slow-flowing, permanent watercourses range from small linear ditches to large ponds, generally with a width of between 5 and 40 m and a depth of up to 3 m. Land use in the study area is predominantly residential, with an impervious area of approximately 30% (roads, buildings, parking lots), while 66% is taken up by gardens, parks and other green areas.

For this study, we selected 25 water bodies in the western part of Nijmegen and 11 water bodies in the southern part of Arnhem. The urban water bodies in Nijmegen were monitored in the April-May 2005 period and a second time in the August-September 2005 period. The Arnhem water bodies were monitored in September 2005 and May 2006. Locations were chosen so as to include variety in morphology, water quality and vegetation.

Macroinvertebrates

Aquatic macroinvertebrates were sampled using a 20 by 30 cm pond net with 0.5 mm mesh size. A sample consisted of two sweeps over a length of approximately 2 m in open water just above the sediment, one sweep starting from the open water towards the bank, and one sweep parallel to the bank. Benthic macroinvertebrates in the top layer of the sediment were sampled three times, using a core sampler (diameter 7 cm; height 9 cm) pushed fully into the sediment at distances of approximately 75 cm, 150 cm and 225 cm from the bank at each sampling station. All samples were washed over three sieves with 2, 1 and 0.5 mm mesh size, and sorted in the laboratory. When possible, macroinvertebrates were identified to species level, although some taxa, such as Oligochaeta, Lepidoptera and Diptera, were only identified to a higher taxonomic level.

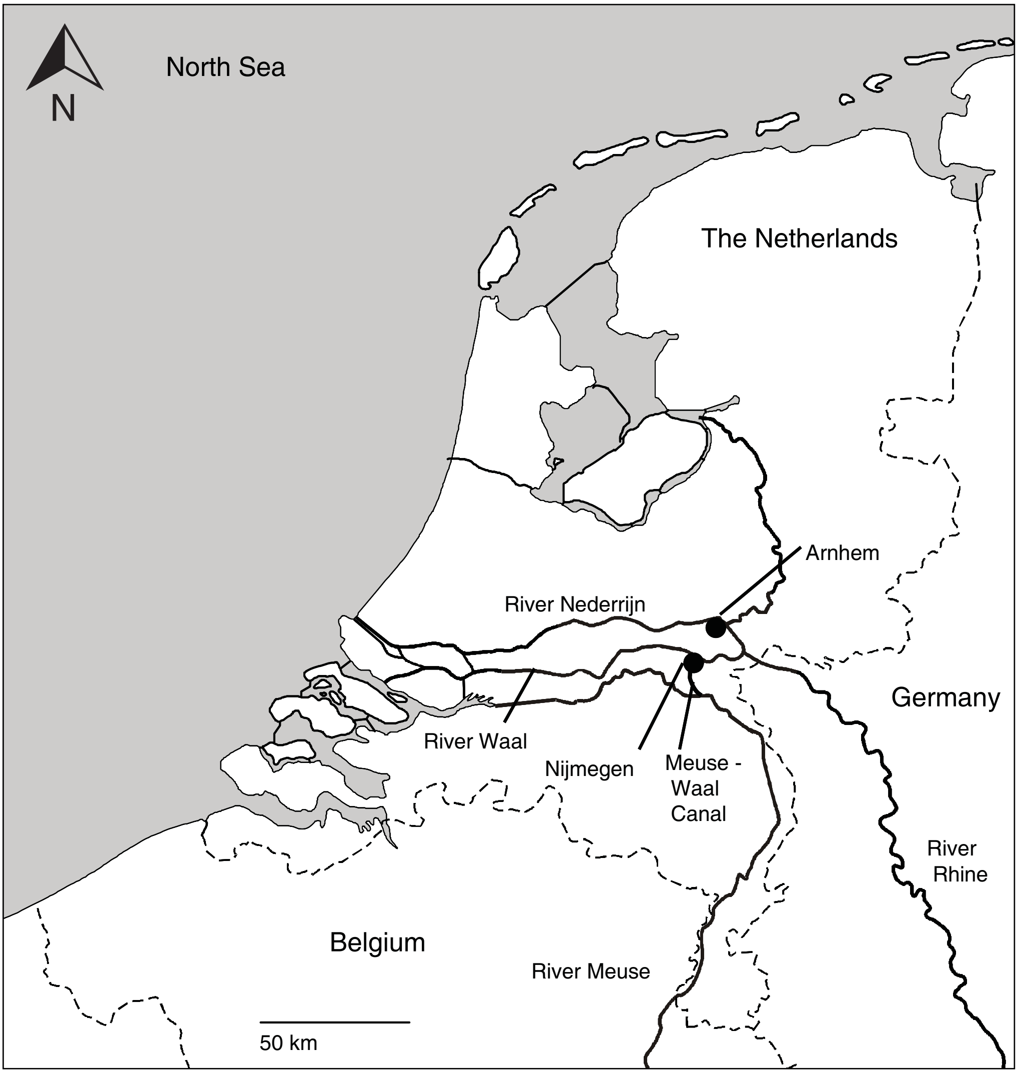

Figure 1 Geographical locations of the areas studied

Macroinvertebrates were identified to different taxonomic levels, because some groups and several young larvae are difficult to identify to species level. Although species-level data might reveal more differences between locations, a higher taxa resolution should be sufficient to distinguish the larger between-site differences (Lenat & Resh, 2001). Hewlett (2000) found similar patterns when using species-, family- or genus-level data in classifying stream sites in Australia.

Multivariate analyses of parts of the data set (for example only including Gastropoda, Crustacea, Ephemeroptera and Trichoptera, which were all identified to species level) did not yield different results than analyses including all taxa, identified to different taxonomic levels. Macroinvertebrates from sediment and water samples, as well as from the two seasons, were pooled in the analysis.

Environmental variables

The following parameters were measured in the field: electrical conductivity (Hanna Combo meter), stream velocity (SENSA-RC2 water velocity meter), dimensions of the water body, percentage of shade, slope of the bank, depth near the bank, transparency (Secchi depth) and percentage cover by submerged vegetation (e.g. *Elodea nuttallii*, *Ceratophyllum demersum*), nymphaeid vegetation (e.g. *Nuphar lutea*, *Nymphaea alba*) and lemnid vegetation (*Lemna* sp.). Abundance of water birds was recorded for 30 min. Dominant species were always sedentary (e.g., *Anas platyrhynchos* and *Fulica atra*). Four abundance classes were distinguished: 0 = absent, 1= 1-20 individuals ha^{-1}, 2 = 20-50 individuals ha^{-1}, 3 = > 50 individuals ha^{-1}. Water bodies that were transparent down to the bottom were assigned a Secchi depth of 1 m, because deeper clear waters would otherwise have a larger relative influence on transparency than shallow clear waters. Two water samples, four pore water samples and three sediment samples were taken for further analysis. The pH and alkalinity were measured the following day, after water and pore water samples had been stored overnight at 4°C. The pH measured in the laboratory was very consistent with that measured in the field (Hanna Combo meter). CO_2 was measured with an ABB Advance Optima Infrared Gas Analyzer (ABB Automation Products, Germany); CO_3^{2-} and HCO_3^- were calculated from CO_2 and pH. Water samples were stored at -20°C until further analysis after adding citric acid (125 mg l^{-1}). The following ions were measured colorimetrically (Auto Analyzer 3, Digital colorimeter, Bran+Luebbe, Germany): NO_3^- according to Kamphake et al. (1967), NH_4^+ according to Grasshoff & Johannsen (1972), Cl^- according to O'Brien (1962) and PO_4^{3-} according to Henriksen (1965). Na^+ and K^+ were measured photometrically with a flame photometer (Radiometer, Copenhagen). Metals were measured by inductively coupled plasma mass spectrometry (Thermo Electron Corporation, United Kingdom). Sediment samples were dried for 24 h at 100°C and grain size was determined with a Coulter LS 230 laser diffraction device (Beckman Coulter, Inc, Fullerton CA, USA). Carbon and nitrogen content of the sediment were measured with a Carbo Erba NA 1500 Nitrogen Carbon Sulphur Analyzer. All physico-chemical factors were measured at least in duplicate and average values were used for the data analysis.

Data analysis

Macroinvertebrate assemblages were distinguished by Two Way Indicator Species Analysis (TWINSPAN; Hill, 1979). If the dissimilarity between macroinvertebrate assemblages was larger than the dissimilarity within the macroinvertebrate assemblages, they were regarded as different assemblages and hence as different water types (Verberk et al., 2006).

Canoco for Windows Version 4.0 (Ter Braak & Šmilauer, 1998) was used to perform Canonical Correspondence Analysis (CCA) in order to relate urban water types to environmental variables. A direct method was used to identify the environmental variables significantly contributing to the variation in macroinvertebrate assemblages (Jongman et al., 1995). A unimodal response model was selected, because there was a wide range in environmental variables. Before analysis, macroinvertebrate abundances were transformed according to Preston (1962): Preston class = $^2\log$ (abundance + 1).

Environmental variables with wide ranges (NH_4^+, PO_4^{3-}, total P, Li^+, total Fe, Cu^{2+}, Pb^{2+}) were log-transformed; Al^{3+} and stream velocity were transformed to an ordinal scale. An ordinal scale was also used for the abundance of water birds and the cover by submerged, nymphaeid and lemnid vegetation. The percentage of clayey and silty fraction

(<64 μm) was used as a measure of sediment composition. Since nutrients and metals in sediment and pore water were highly correlated with nutrients and metals in surface water, only nutrients and metals in surface water were used for data analysis. Nutrients and metals in surface water were also more relevant, since most macroinvertebrates occurred in the water layer. Significance of environmental variables was tested with CCA, using 500 Monte Carlo permutations under full model conditions. Differences between urban water types were tested with ANOVA (post-hoc Gabriel, $p<0.05$).

The following ecological indicators were used with respect to biodiversity: taxa richness, Shannon index, number of red list species, number of exotic species and rareness.

Taxa richness was expressed as the number of taxa in each location. The Shannon index (H') was calculated according to Shannon (1948) using natural logarithms:

$$H' = -\sum_{i=1}^{S} p_i \ln p_i$$

where S is the number of taxa and p_i the relative abundance of each taxon. Numbers of red list species were determined according to the national list (Veerman, 2004), which describes species that are extinct, threatened or vulnerable in The Netherlands.

Exotic species were determined based on the list by Van der Velde et al. (2002). Species are considered exotic species when they have established themselves after intentional or unintentional introduction by human activities far from their original biogeographic area.

Based on Foster et al. (1990), a score for rareness was calculated for each location, including only taxa identified to species level. Each species was assigned a score according to its rareness in The Netherlands over the 2000-2005 period (STOWA, 2006). The species were given a score between 1 and 6, corresponding to > 2,000, 1,000-2,000, 500-1,000, 200-500, 100-200 and <100 occurrences in the national database, respectively. The rareness score of a location was calculated as the sum of all species rareness scores of the location divided by the number of species present.

The key environmental variables determined by the multivariate analysis were correlated to the ecological indicators for biodiversity (taxa richness, Shannon index, number of red list species, number of exotic species and rareness) with Spearman's Rho correlation on all urban water locations.

Two methods were used to compare biodiversity in urban water systems with that in other aquatic systems in The Netherlands. First of all, the urban water systems were compared with total aquatic macroinvertebrate species richness in The Netherlands, within the taxa that were generally identified to species level. According to the species pool hypothesis the size of the species pool strongly influences the species richness of the local community (Eriksson, 1993). The total aquatic macroinvertebrate species richness in The Netherlands represents the national species pool, which partly determines the regional species richness in the urban water systems. Secondly, we compared conservation values of urban waters with those of other drainage systems by selecting 30 locations from the national database held by the water boards (STOWA, 2006), including 10 canals, 10 ditches and 10 lotic watercourses. Canals and ditches were chosen to represent man-made drainage systems in rural areas, while lotic waters (i.e. small streams and rivulets) were selected to include natural and seminatural watercourses. The locations were randomly chosen and were situated across The Netherlands, because no data were available on macroinvertebrates in rural water systems close to Arnhem and Nijmegen. A random selection of sampling sites was made to minimize the influence of systematic differences in monitoring methods. To allow comparisons, the data was transformed to the same taxa resolution as our data on urban waters, and the data of the two seasons were pooled. Macroinvertebrates were sampled over a length of 5 m with a pond net (STOWA, 2006), instead of 4 m and 3 benthic samples (our surveys in urban waters). Urban water systems were compared pairwise with other drainage systems using a Student's t-test ($p<0.05$).

Results

Four different macroinvertebrate assemblages were distinguished, based on taxa and their abundances (Figure 2). Multivariate analysis and TWINSPAN showed that nitrate, pH, grain size (sediment composition), transparency and nymphaeid and submerged vegetation explained a significant proportion of the variation in macroinvertebrate assemblages in urban water systems (approximately 27%). The macroinvertebrate assemblages were found to be associated with the urban water types created by different environmental conditions within the urban area (Figure 3).

Water bodies of type 1 were characterized by low nutrient levels, sandy soils, turbid water and poorly developed vegetation (Table 1). Type 2 also had low nutrient levels, but with a clayey soil and vegetation always present. Nymphaeids (e.g. *Nuphar lutea*, *Nymphaea alba*) were present in almost half of the locations of this type. Type 3 was characterized by high nutrient levels, high transparency and a high cover of submerged vegetation. Type 4 had the highest nutrient levels and poorly developed submerged vegetation. Lemnids (*Lemna* sp.) were dominant in four locations.

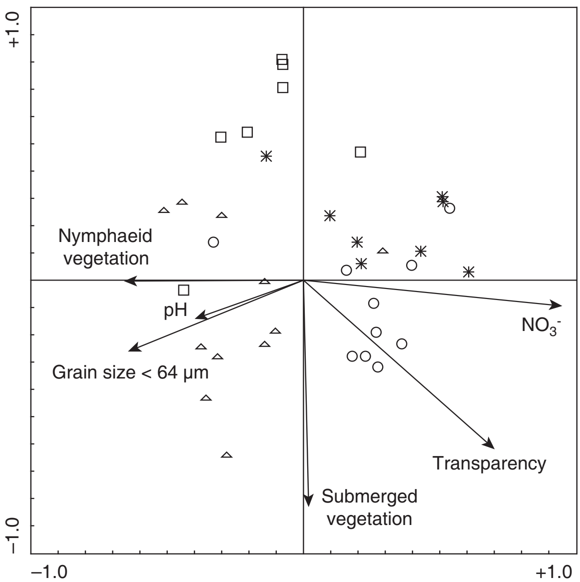

Figure 2a Canonical correspondence analysis of invertebrate taxa. Symbols represent different water types, distinguished with TWINSPAN (squares = type 1, triangles = type 2, circles = type 3, stars = type 4). Arrows represent environmental variables explaining a significant proportion of variation in macroinvertebrate assemblages between locations.

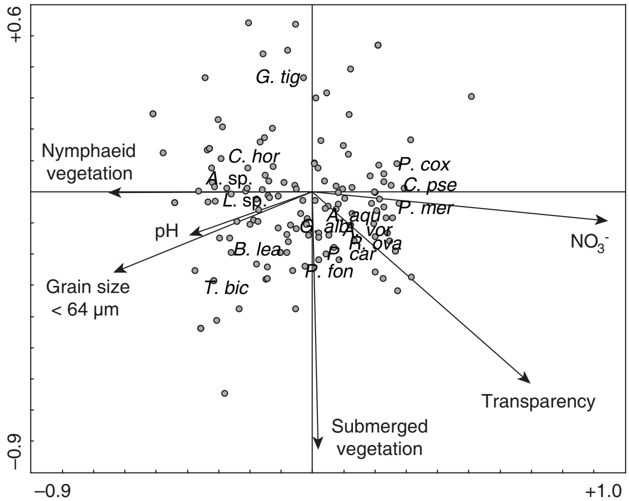

Figure 2b Canonical correspondence biplot of invertebrate taxa. Circles represent taxa and arrows represent environmental variables that significantly explain the variation in macroinvertebrate assemblages (*A. vor*: *Anisus vortex*, *B. lea*: *Bithynia leachii*, *G. alb*: *Gyraulus albus*, *P. fon*: *Physa fontinalis*, *P. car*: *Planorbis carinatus*, *R. ova*: *Radix ovata*, *A. aqu*: *Asellus aquaticus*, *C. hor*: *Caenis horaria*, *T. bic*: *Triaenodes bicolor*, *A.* sp.: *Arrenurus* sp., *L.* sp.: *Limnesia* sp., *C. pse*: *Crangonyx pseudogracilis*, *G. tig*: *Gammarus tigrinus*, *P. cox*: *Proasellus coxalis*, *P. mer*: *Proasellus meridianus*).

Figure 3 Four types of urban water bodies distinguished in this study.

Table 1 Average values of environmental variables (minimum and maximum values in brackets). [a] and [b] indicate significant differences between types (in bold, ANOVA, post-hoc Gabriel: $p<0.05$).

	Type 1 (8)	Type 2 (11)	Type 3 (8)	Type 4 (9)
NO_3^- (mg l^{-1})	**2.0[a] (0.5-6.8)**	**1.3[a] (0.0-4.7)**	**5.1[b] (0.4-7.8)**	**5.7[b] (0.7-10.3)**
NH_4^+ (mg $^{-1}$)	0.18 (0.05-0.81)	0.15 (0.02-0.30)	0.11 (0.03-0.33)	0.12 (0.04-0.16)
PO_4^{3-} (mg l^{-1})	0.17 (0.03-0.93)	0.06 (0.02-0.12)	0.10 (0.02-0.37)	0.12 (0.03-0.19)
Total-P (mg l^{-1})	0.06 (0.00-0.26)	0.03 (0.01-0.08)	0.04 (0.01-0.13)	0.05 (0.01-0.08)
Total-S (mg l^{-1})	11.4 (5.3-18.8)	10.0 (3.4-18.4)	12.6 (3.7-18.2)	13.6 (7.6-19.7)
K^+ (mg l^{-1})	4.5 (2.5-5.5)	4.7 (0.8-9.7)	4.0 (2.6-4.8)	5.5 (3.9-10.6)
Na^+ (mg l^{-1})	26 (18-36)	26 (19-36)	30 (27-34)	32 (27-36)
Cl^- (mg l^{-1})	**42[ab] (33-57)**	**40[a] (29-55)**	**48[ab] (42-55)**	**50[b] (46-58)**
pH	7.8 (7.5-8.1)	7.7 (7.4-9.2)	7.6 (7.1-8.6)	7.4 (7.1-7.9)
Alkalinity (meq l^{-1})	3.0 (2.1-4.8)	3.2 (0.9-5.1)	2.5 (1.9-4.2)	2.5 (2.0-4.3)
CO_2 (mg l^{-1})	5.9 (2.2-8.9)	9.0 (0.1-15.4)	10.6 (3.9-16.3)	10.8 (5.5-16.6)
HCO_3^- (mg l^{-1})	187 (125-313)	207 (56-344)	148 (108-256)	151 (116-266)
EC (µS m^{-1})	520 (393-647)	533 (288-667)	520 (477-632)	531 (482-656)
Ca^{2+} (mg l^{-1})	67 (49-87)	63 (14-88)	57 (47-73)	58 (49-79)
Mg^{2+} (mg l^{-1})	9.3 (7.9-11.2)	9.2 (3.0-13.3)	8.1 (7.2-9.6)	8.7 (7.4-13.0)
Li^+ (µg l^{-1})	2.0 (1.0-4.1)	3.5 (1.4-6.7)	2.5 (1.1-3.6)	2.2 (0.3-6.4)
Total-Fe (µg l^{-1})	67 (35-119)	168 (50-653)	146 (55-354)	68 (25-188)
Al^{3+} (µg l^{-1})	8.4 (4.6-15.5)	7.6 (4.2-13.3)	4.7 (2.3-7.1)	15.0 (2.3-90.2)
Cu^{2+} (µg l^{-1})	1.5 (0.7-3.3)	1.9 (0.8-6.3)	1.9 (0.7-3.9)	2.3 (0.7-5.1)
Zn^{2+} (µg l^{-1})	26 (19-40)	29 (10-55)	31 (23-48)	38 (26-68)
Pb^{2+} (µg l^{-1})	0.2 (0.1-0.3)	0.4 (0.1-1.1)	0.2 (0.1-0.5)	1.0 (0.1-3.1)
C/N content soil	11 (7-20)	14 (9-20)	13 (10-17)	11 (6-15)
% Grain size < 64 µm	**12[a] (2-63)**	**54[b] (1-97)**	**21[ab] (3-69)**	**11[a] (1-60)**
Transparency (Secchi (m))	**0.6[a] (0.4-1)**	**0.8[ab] (0.4-1)**	**0.9[b] (0.5-1)**	**0.9[b] (0.4-1)**
Profile (depth (m) × slope)	14 (6-26)	11 (3-27)	16 (3-27)	11 (4-23)
Stream velocity (cm s^{-1})	4.1 (2.8-10.5)	4.8 (2.8-9.7)	13.8 (3.6-60.4)	3.1 (2.6-4.0)
Width (m)	**22[a] (12-30)**	**14[ab] (5-30)**	**11[b] (6-25)**	**12[b] (8-20)**
Shade (%)	58 (30-80)	55 (25-80)	44 (0-80)	58 (5-95)
Nymphaeid vegetation[c]	**0.2[ab] (0-1)**	**0.5[a] (0-1)**	**0[b] (0-0)**	**0[b] (0-0)**
Submerged vegetation[d]	**0.7[a] (0-3)**	**1.8[ab] (0.5-3)**	**2.4[b] (0.5-3)**	**0.8[a] (0-2)**
Floating vegetation (*Lemna* sp.)[d]	0 (0-0)	0.2 (0-1)	0.6 (0-1.5)	1.1 (0-3)
Water birds[e]	1.9 (1-3)	1.5 (0-3)	2.2 (1-3)	2.4 (1-3)

[c] 0=absent, 1=present, [d] 0=absent, 1=<10%, 2=10-50%, 3=>50% cover, [e] 0=0 ha^{-1}, 1=1-20 ha^{-1}, 2=20-50 ha^{-1}, 3=>50 ha^{-1}

Characteristic macroinvertebrate species in the various urban water types are shown in table 2 and figure 2b. Type 1 was mainly characterized by the lack of many common species, while *Gammarus tigrinus* was a characteristic species in this type. The snails *Gyraulus albus*, *Physa fontinalis*, the caddis fly *Triaenodes bicolor* and the mites *Arrenurus* sp. and *Limnesia* sp. were most characteristic of type 2 with its low nutrient levels. All locations of type 3, with its high abundance of submerged vegetation, harboured the gastropods *Anisus vortex*, *Gyraulus albus*, *Physa fontinalis* and *Radix ovata*. Type 4, with its nutrient-rich conditions, was characterized by many species being less abundant than in types 2 and 3. Most of the type 4 locations harboured the exotic crustaceans *Crangonyx pseudogracilis*, *Proasellus coxalis* and *Proasellus meridianus*.

Our analysis of the correlations between the environmental variables and ecological indicators in urban water systems demonstrated that nitrate was negatively correlated with taxa richness, number of red list species and rareness, and positively correlated with the number of exotic species (Table 3). Rareness was positively correlated with pH. Going

Table 2 Percentage of locations where characteristic (in bold), red list and exotic macroinvertebrate species were present within the four different urban water types: type 1 – turbid, taxa-poor; type 2 – nutrient- poor, taxa-rich; type 3 – richly vegetated, taxa-rich and type 4 – nutrient-rich, taxa-poor.

Characteristic species	Type 1	Type 2	Type 3	Type 4
Gastropoda				
Anisus vortex	0	45.5	100.0	44.4
Bithynia leachii	12.5	72.7	12.5	0
Gyraulus albus	25.0	100.0	100.0	33.3
Physa fontinalis	12.5	81.8	100.0	22.2
Planorbis carinatus	0	45.5	87.5	11.1
Radix ovata	0	27.3	100.0	22.2
Crustacea				
Asellus aquaticus	50.0	100.0	100.0	100.0
Ephemeroptera				
Caenis horaria	87.5	100.0	75.0	44.4
Trichoptera				
Triaenodes bicolor	37.5	72.7	25.0	0
Acari				
***Arrenurus* sp.**	37.5	90.9	25.0	11.1
***Limnesia* sp.**	50.0	90.9	25.0	22.2
Red list species	Type 1	Type 2	Type 3	Type 4
Tricladida				
Planaria torva	0	9.1	0	0
Trichoptera				
Leptocerus tineiformis	12.5	45.5	25.0	0
Exotic species	Type 1	Type 2	Type 3	Type 4
Bivalvia				
Dreissena polymorpha (PC)	12.5	63.6	87.5	55.6
Gastropoda				
*Ferrissia wautieri** (NA)	12.5	27.3	0	0
Physella acuta (NA)	0	9.1	0	11.1
Potamopyrgus antipodarum (NZ)	12.5	27.3	25.0	44.4
Tricladida				
Dugesia tigrina (NA)	25.0	45.5	25.0	11.1
Crustacea				
Crangonyx pseudogracilis (NA)	0	9.1	62.5	77.8
Gammarus tigrinus (NA)	75.0	27.3	12.5	11.1
Limnomysis benedeni (PC)	62.5	27.3	0	33.3
Proasellus coxalis (EE)	0	36.4	25.0	77.8
Proasellus meridianus (SE)	25.0	27.3	75.0	77.8

* possibly *F. gracilis*
Origin: PC = Ponto-Caspium, NA = North America, NZ = New Zealand, EE = Eastern Europe, and SE = Southern Europe

Table 3 Spearman's Rho correlations between environmental variables explaining a significant proportion of variation in macroinvertebrate assemblages and ecological indicators in urban water systems.

	Taxa richness	Shannon Index	Number of red list species	Number of exotic species	Rareness
NO_3^- (mg l^{-1})	–0.29*	–0.00	–0.50***	0.30*	–0.51***
pH	0.08	0.10	0.16	–0.11	0.52***
% Grain size < 64 μm	0.43**	0.20	0.39*	–0.28*	–0.12
Transparency (Secchi (m))	0.06	0.27	–0.27	0.23	–0.48**
Nymphaeid vegetation	0.29*	–0.02	0.22	–0.15	0.40**
Submerged vegetation	0.54***	0.43**	0.14	–0.02	–0.11

* p<0.05, ** p<0.01, *** p<0.001

from sandy to clayey sediment, taxa richness and the number of red list species increased, while the number of exotic species decreased. Transparency was negatively correlated with rareness. Nymphaeid vegetation was positively correlated with taxa richness and rareness, while submerged vegetation was positively correlated with taxa richness and Shannon index values.

Approximately 13% of the aquatic macroinvertebrate species occurring in The Netherlands were found in the urban water systems of Arnhem and Nijmegen (Table 4), with various groups found there contributing considerably to the biodiversity in The Netherlands. For instance, approximately 50% of the triclad and gastropod species were present in the urban water systems, while Plecoptera were absent.

Taxa richness under nutrient-poor conditions (type 2) and richly vegetated conditions (type 3) did not differ significantly from that of similar drainage systems in rural areas (Figure 4). Taxa richness was lowest in turbid (type 1) and nutrient-rich urban waters (type 4), but not significantly different from taxa richness in canals. The Shannon index was highest in ditches, while urban water systems had an intermediate Shannon index. No red list species were found in the nutrient-rich urban water bodies (type 4) and in natural or seminatural lotic waters in rural areas. The number of exotic species was highest in the nutrient-rich urban waters (type 4) and lowest in turbid urban waters (type 1) and natural and seminatural lotic waters. Rareness was highest in the turbid (type 1) and nutrient-poor (type 2) urban waters and lowest in the richly vegetated (type 3) and nutrient-rich (type 4) urban waters, but not significantly different from canals, ditches and lotic waters.

Table 4 Number of macroinvertebrate species found in the urban water systems in Arnhem and Nijmegen, as a percentage of the total number of species occurring in fresh water in The Netherlands.

	Urban water (%)	References
Tricladida	50.0	Mol, 1984
Gastropoda	50.0	Gittenberger et al., 1998
Bivalvia[a]	25.0	Gittenberger et al., 1998
Hirudinea[b]	41.2	Nederlands Soortenregister, 2008
Crustacea	14.0	Mol, 1984, Van der Velde et al., 2000
Odonata[c,d,e]	24.4	Bos & Wassscher, 1997, Dijkstra et al., 2002
Ephemeroptera	9.8	Mol, 1984
Plecoptera	0	Koese, 2008
Heteroptera[f,g]	33.3	Aukema et al., 2002
Coleoptera[h]	2.8	Drost et al., 1992
Trichoptera	12.2	Higler, 2005, Higler, 2008
Total	13.4	

The following species were merged to form one group:
[a]*Pisidium* sp., [b]*Erpobdella* sp., [c]*Aeshna* sp., [d]*Libellulidae*, [e]*Coenagrion puella/ pulchellum* and *Ischnura elegans*, [f]*Gerris* sp., [g]*Notonecta* sp., [h]Hydrophilidae.

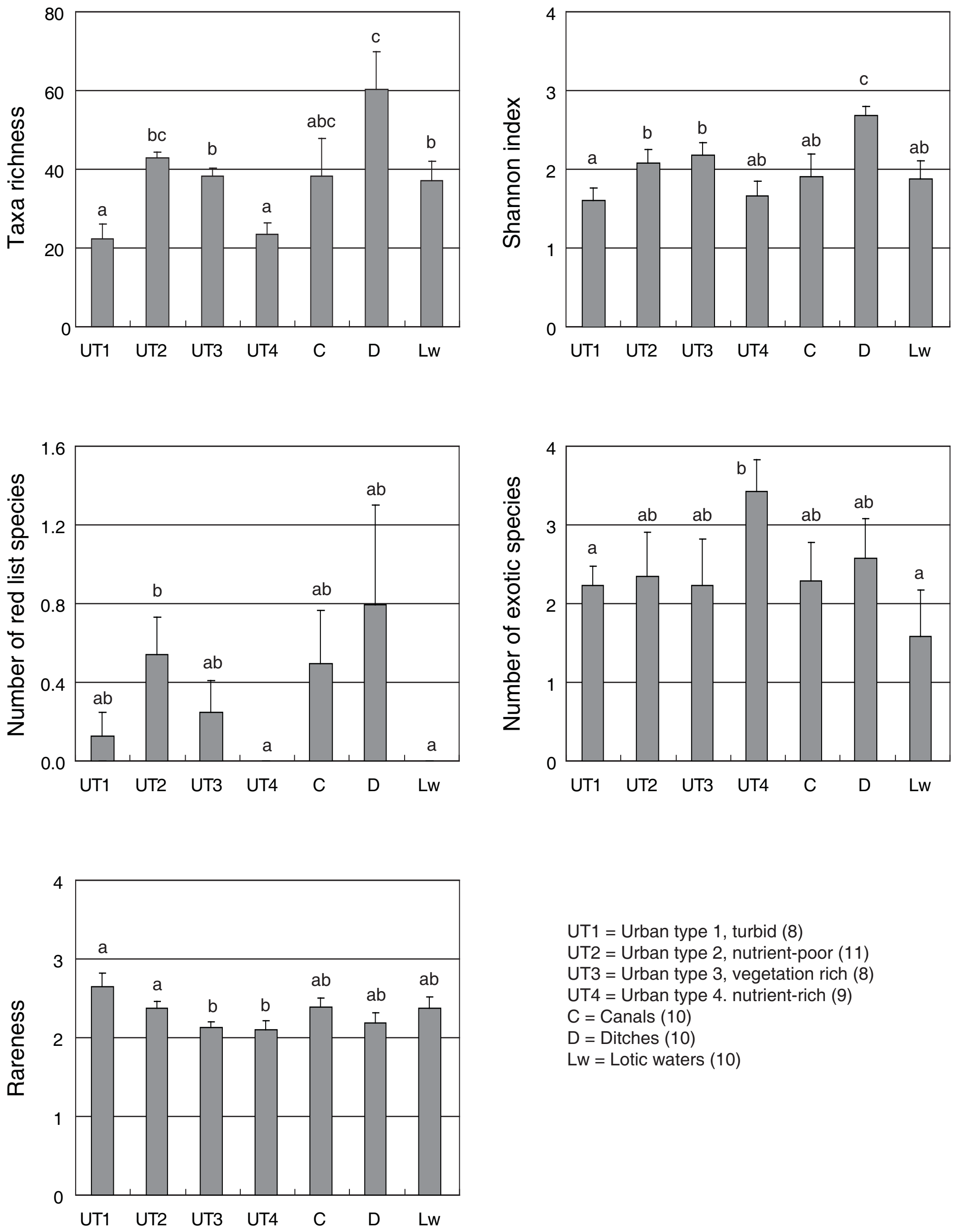

Figure 4 Average values of ecological indicators in the four types of urban water systems and non-urban drainage systems. Error bars represent the standard error. Significant differences between water body types are indicated by a, b and c (Student's t-test, p<0.05).

Discussion

Urban water conditions

The nutrient-poor urban water bodies harboured the highest numbers of macroinvertebrate taxa among the urban water systems, as well as the highest number of red list species. In addition to submerged vegetation, some locations also had nymphaeid vegetation present, creating a more diverse habitat structure for macroinvertebrate species (Den Hartog & Van der Velde, 1988).

The richly vegetated water bodies were also rich in macroinvertebrate taxa. Although this water body type had high nutrient levels, submerged vegetation may have kept the water bodies in a 'clear' state (Scheffer et al., 1993). The abundant vegetation created ideal circumstances for the herbivorous gastropods that characterize the macroinvertebrate assemblage of this water type.

The turbid urban water bodies were characterized by the absence or very low density of submerged vegetation, and harboured the lowest number of macroinvertebrate taxa. If vegetation is absent, waves and fish foraging activities can cause high turbidity due to resuspension of suspended matter (Madsen et al., 2001, Gulati & Van Donk, 2002). This high turbidity prevents the vegetation from developing fully. The lack of submerged vegetation probably explains the low number of macroinvertebrate taxa in the turbid water bodies.

Macroinvertebrate taxa richness was also low in the nutrient-rich water bodies, but the number of exotic species was highest there. The increased nutrient concentrations probably stimulated the growth of algae and floating plants (*Lemna* sp.), inhibiting the growth of submerged vegetation (Hough et al., 1989). The exotic species may be able to cope better with harsh environmental circumstances (e.g. anoxia), and hence fill vacant niches left by the more sensitive native species. Brauns et al. (2007) also found more invasive species in hypertrophic lakes. Grabowski et al. (2007) argued that invasive gammarids are more tolerant to environmental stressors (salinity, pollution and habitat degradation) than native species, facilitating invasion. Another explanation for the presence of these exotics in the highly eutrophied water bodies is the much higher resource availability in such waters, increasing the chances of invasion (Van der Velde et al., 2006).

Correlations between environmental variables and ecological indicators

Our analysis of the correlations between the environmental variables and the ecological indicators indicated that increasing nitrate levels were negatively related to taxa richness, number of red list species and rareness (Table 3). Exotic species tended to increase with nitrate levels. Nitrate and other nutrients probably did not influence the macroinvertebrate assemblages directly, but did influence the vegetation composition and structure (Hough et al., 1989) and hence the macroinvertebrates (Van den Brink & Van der Velde, 1991). Nutrient enrichment of urban waters in Arnhem and Nijmegen was probably caused by upward seepage from nutrient-rich river water (Vermonden, unpublished data), as the water quality of the rivers Nederrijn, Waal and Meuse and the Meuse-Waal canal does not yet comply with environmental quality standards for nutrients (Ministry of Transport, Public Works and Water Management, 2005). Other factors contributing to the eutrophication of urban waters include storm-water run-off, sewage outlets, faeces of water birds and fishes, leaf fall and subsequent delivery of nutrients from sediments and lack of flow. Nutrient levels in urban water systems will thus have to be limited to optimize indigenous biodiversity and prevent invasions by exotic species. This can be done by regular dredging, by avoiding the inlet of nutrient-rich (sewage) water and by preventing the (excessive) feeding of water birds and benthivorous fish.

Rareness correlated positively with pH, but overall pH did not differ much between urban water types. Effects of pH differences were not expected between the relatively neutral values of 7.1 and 9.2.

Taxa richness and numbers of red list species increased going from sandy to clayey sediment, while the number of exotic species decreased. In these urban water systems, the water bodies with sandy sediment were more strongly influenced by the nutrient-rich upward seepage from rivers and canals. It is therefore likely that sediment composition only indirectly influenced the ecological indicators by affecting the water quality and hence the vegetation. The negative impact of nutrient-rich upward seepage from rivers and canals could be decreased by further reducing the nutrient loading in the rivers.

Rareness decreased with transparency. Although the turbid urban water bodies sustained fewer species, these species might be a little rarer, because the pioneer species found in this type of water body were not present in many other water bodies (e.g. *Caenis luctuosa* and *Micronecta minutissima*). Maintaining some urban water bodies in a turbid, pioneer state could thus increase overall biodiversity.

Nymphaeid vegetation was positively correlated with taxa richness and rareness, while submerged vegetation was positively correlated with taxa richness and Shannon index. Vegetation is very important for macroinvertebrates, as a habitat, food source and shelter from predation (Crowder & Cooper, 1982, Dvořak & Best, 1982, Newman, 1991), and vegetation is known to have a favourable influence on macroinvertebrates, depending on the growth form (Den Hartog & Van der Velde, 1988). An optimal mowing regime and the development of natural banks can stimulate the development of submerged and nymphaeid vegetation.

Comparison of urban water systems with other water systems

A significant proportion of the aquatic macroinvertebrate species pool in The Netherlands was also present in the urban water systems of Arnhem and Nijmegen, including two red list species (Tables 2 and 4). Taxa richness and conservation values (e.g. rareness and numbers of red-listed and exotic species) in the nutrient-poor and richly vegetated urban water systems were more or less comparable to those in the other drainage systems: canals, ditches and natural or seminatural lotic waters (Figure 4). These figures may be biased by slight differences in sampling methods (e.g. effort and period), species pool and environmental conditions (e.g. temperature). Temperature can influence both macroinvertebrate species composition and abundance (Haidekker & Hering, 2008). However, it is not yet clear whether differences in temperature regime between urban and rural areas are large enough to induce changes in the macroinvertebrate species. We expect that differences in water quality, morphology and habitat structure override possible effects of differences in water temperature between urban and rural waters. High numbers of individuals per sampling site and low standard errors indicate low within-group variation. Moreover, the Shannon index, which is relatively independent of sampling effort and allows direct comparisons, yielded a similar conclusion.

Our results in terms of conservation values disagree with those of other studies, which always found lower macroinvertebrate diversity in urban areas (Paul & Meyer 2001, Lenat & Crawford, 1994, Roy et al., 2003, Walsh et al., 2001). The urban areas investigated in these studies may have been more degraded than the urban systems we studied. In addition, urban areas in previous studies were usually situated in downstream parts of the catchment (Walsh et al., 2001), which means that differences might be a result of the upstream to downstream gradient, rather than of differences in land use only. Another reason for the discrepancy could be that the urban water systems were investigated only as part of a natural-rural-urban gradient (Paul & Meyer, 2001, Lenat & Crawford, 1994, Roy et al., 2003, Walsh et al., 2001), disregarding any possible variation within the urban areas, while our study included an in-depth analysis of various urban water types.

Conclusions

We studied urban drainage systems in lowland areas along large rivers and distinguished four types of water bodies based on various macroinvertebrate assemblages, differing in environmental conditions and values of ecological indicators. Two types had low macroinvertebrate taxa richness, viz. turbid water bodies and nutrient-rich water bodies with very poorly developed vegetation. The nutrient-rich water bodies were characterized by the highest numbers of exotic species. The two other types showed high macroinvertebrate taxa richness, viz. nutrient-poor water bodies and water bodies with a high cover of submerged vegetation. The highest number of red list species was found in the nutrient-poor water bodies. The water bodies with a high cover of submerged vegetation were characterized by herbivorous gastropods.

Key factors for the conservation of macroinvertebrates in urban water systems are nitrate, sediment composition, transparency and nymphaeid and submerged vegetation. To optimize biodiversity in urban water systems, management should aim at lowering nutrient levels (e.g. by regular dredging, avoiding the inlet of nutrient-rich water and decreasing (excessive) feeding of water birds and fish), stimulating aquatic as well as helophyte vegetation and increasing transparency (e.g. by optimizing mowing regimes and developing natural banks). Although increasing transparency generally results in a higher biodiversity, maintaining a few urban water bodies in a turbid, pioneer state with mineral sediment could increase overall biodiversity, yielding more rare species.

Although urban water systems were previously considered to have a low biodiversity value, a significant proportion of the aquatic macroinvertebrate species in The Netherlands was actually present in the urban water systems of Arnhem and Nijmegen, including two red list species. Ecological indicators of biodiversity did not differ significantly between urban water systems and rural drainage systems. This study shows that urban water systems can sustain a biodiversity comparable to man-made drainage systems in rural areas and natural and seminatural watercourses, and can even offer habitats for several red list species. Our study showed the potential conservation benefits for biodiversity of urban water systems. Further studies are needed to investigate the optimal design of towns and cities to improve biodiversity values as an integrated part of the urban environment, and to quantify the functioning of urban water systems as greenways for the dispersal of flora and fauna species in increasingly fragmented landscapes.

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Chironomus anthracinus (pupa), Esrum Sø, Denmark. Photo: Klaus P. Brodersen

Chapter 5

Key factors for chironomid diversity in urban waters

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Submitted

Abstract

There is a growing interest in the biodiversity value of urban waters. Understanding key ecological processes is essential for effective management of these aquatic ecosystems. This paper focuses on identifying the key factors structuring chironomid assemblages in urban waters. Chironomid assemblages were studied in urban surface water systems in lowland areas of the Rhine-Meuse river district in the Netherlands. Multivariate analysis was used to identify the key environmental factors. Taxon richness, Shannon-Index, rareness of species and life-history strategies were compared with available data on similar water bodies in rural areas. We also assessed the effectiveness of dredging on restoring chironomid diversity in urban waters.

Three different chironomid associations were distinguished with TWINSPAN. Variation within and between chironomid associations were significantly related to substrate (sludge layer and substrate type: sand vs. clay), the presence of lemnids, submerged vegetation, filamentous algae, and transparency. Chironomid taxon richness and Shannon-index were similar in urban and rural waters, while values for rareness were higher in urban waters.

Nutrients and sludge layer played a key role for chironomids in urban water systems. Chironomid taxon richness was negatively related to sludge layer and abundance of lemnids. Dredging resulted in different chironomid species composition, higher taxon richness and more species with life-history strategies indicative for good oxygen conditions. Therefore, dredging can be regarded as an effective measure to restore diversity of chironomid communities in urban waters.

Introduction

In our rapidly urbanizing world, freshwater ecosystems are under pressure (Grimm et al., 2008). Water systems are modified or constructed to meet a wide range of services, such as drinking water, irrigation, flood control, transportation, recreation, and wildlife habitat (Postel & Carpenter, 1997). Urbanization has profound effects on existing natural water systems, changing the hydrology, morphology, water chemistry and biota (Ehrenfeld, 2000, Paul & Meyer, 2001, Walsh et al., 2005a). Urban water systems are prone to higher and more frequent peak discharges due to an increased impervious area in cities (Schueler, 1994). Often they also have higher nutrient and toxicant loadings (Schueler, 1994, Paul & Meyer 2001, Walsh et al., 2005a). Under these conditions flora and fauna diversity subsequently decreases (Lenat & Crawford, 1994, Paul & Meyer, 2001).

Benthic macroinvertebrates are often used in environmental quality assessments, because of their high abundance, high species diversity, and varying sensitivity of species to changing environmental conditions. Furthermore, they are relatively easy to sample in a standardized way (Metcalfe, 1989). Benthic macroinvertebrate communities are often degraded by urbanization. Sensitive taxa, such as Ephemeroptera, Plecoptera, and Trichoptera tend to decrease in diversity and abundance, while tolerant taxa, such as Chironomidae and Oligochaeta tend to increase (Lenat & Crawford, 1994, Paul & Meyer, 2001, Walsh et al., 2001, Roy et al., 2003, Kratzer et al., 2006, Moreno & Callisto, 2006).

Species traits can help to unravel the causal mechanisms between species patterns and ecological processes (e.g., Townsend & Hildrew, 1994, Dolédec et al., 2006, Van Kleef et al., 2006, Díaz et al., 2008). Species traits are often linked together by evolution and they can not be treated as independent entities (Statzner et al., 1997, Usseglio-Polatera et al., 2000, Poff et al., 2006, Verberk et al., 2008). Life-history strategies combine species traits, based on relationships among species traits (Verberk et al., 2008). Analysis of life-history strategies can give direct information on how a particular environment is experienced by the species present and it is expected to be a useful method to elucidate causality between species assemblages and environmental factors.

Understanding the relationship between biological communities and the local environmental factors is essential for effective rehabilitation of urban water systems. There is a growing interest in the ecosystem-based management of urban water systems (Walsh, 2000, Bernhardt & Palmer, 2007). Unfortunately, in several cases the effects of rehabilitation measures in urban streams on biological communities and ecosystem functioning were very limited (Larson et al., 2001, Booth, 2005, Suren & McMurtrie, 2005). Most projects focused on restoration of the riparian zone, in-stream habitat augmentation, bank stabilization, and fish passage development. Walsh et al. (2005b) suggested that these local-scale measures are probably ineffective, because of the catchment-scale effects of impervious area. Others suggested that dispersal and recruitment might limit rehabilitation success of streams (Bond & Lake, 2003, Blakely et al., 2006). Vermonden et al. (2009a) showed that the nutrient status is one of the key factors driving the macroinvertebrate diversity of urban water systems in our study area. Regular dredging could reduce internal loading of nutrients (Jeppesen et al., 1999, Ruley & Rusch, 2002).

Chironomids are very suitable for analyzing relations between biodiversity and environmental factors and for testing the effect of dredging on macroinvertebrates. This taxon comprises many benthic species that depend on sediments (Armitage et al., 1995) as source of food (De Haas et al., 2002) or shelter from predators (Macchiusi & Baker, 1992). Furthermore, chironomids show a relatively high species richness and exhibit a wide range of environmental tolerances, with different adaptations to specific types of impairment (Pinder, 1983, Armitage et al., 1995, King & Richardson, 2002, Carew et al., 2007). This study aims at identifying the key factors structuring chironomid assemblages in urban waters and analyzing the effectiveness of dredging in restoring diversity in chironomid communities. The following research questions are addressed:

(1) What is the variation in chironomid assemblages and can different chironomid associations be distinguished based on chironomid assemblages?
(2) What is the relationship between chironomid assemblages, diversity indices and environmental factors such as nutrients, vegetation, and dredging?
(3) Can life-history strategies explain variability between chironomid assemblages in relation to these environmental factors?
(4) How do chironomid diversity indices and life-history strategies in urban waters compare with available data on similar water bodies in rural areas?

Methods

Study area

The sampling sites were located in polder areas of the municipalities of Nijmegen and Arnhem in the eastern part of the Netherlands along 2 distributaries of the river Rhine (rivers Waal and Nederrijn) and the Meuse-Waal canal. Vermonden et al. (2009a) described the geographical location of sampling sites. The population density of the municipalities of Nijmegen and Arnhem approximated 2,800 and 1,400 inhabitants per km^2, respectively (Statistics Netherlands, 2009). In the 1970s, urban water systems were designed to regulate groundwater levels, to manage rain water run-off, and to discharge drainage water into the rivers. Land use in the study area is predominantly residential, with an impervious area of approximately 30% (roads, buildings, parking lots), while 66% is taken up by gardens, parks, and other green areas and approximately 4% by surface waters (Vermonden et al., 2009a, b). The latter are slow-flowing, permanent watercourses ranging from linear ditches to small drainage ponds, generally with a width of between 5 and 40 metres and a depth of up to 3 metres (Vermonden et al., 2009a). The slow-flowing, permanent watercourses are connected via culverts (current velocity on average 3-14 cm s^{-1}). Macrophyte cover and species composition varies between water bodies (Vermonden et al. 2010). Examples of macrophyte species present in urban waters are: *Ceratophyllum demersum*, *Elodea nuttallii*, *Callitriche* sp., *Lemna* sp., and *Nymphaea alba*.

In total 29 locations were studied. In the western part of Nijmegen 13 water bodies were monitored in 2005, 2006, and 2007. Four of these locations were dredged in the winter of 2005-2006. Additionally, 3 locations in the western part of Nijmegen and 13 locations in the southern part of Arnhem were monitored only in 2006 and 2007. All locations in Arnhem were dredged before monitoring started (2003-2005). Sampling of each site took place twice a year; once in April-May and once in August-September. Locations were chosen to cover a range in morphology, nutrient status, sediment quality, and vegetation (Vermonden et al. 2009a, b). Our sampling sites were regarded to be representative for urban drainage systems in polder areas along large lowland rivers in North-western Europe. The rivers influence the urban water systems by introducing nutrient-rich river water via upward seepage (Vermonden et al., 2009b).

Sampling chironomids

Benthic chironomids in the top layer of the sediment were sampled, using a core sampler (diameter 7 cm; height 9 cm) pushed fully into the sediment once at distances of approximately 75 cm, 150 cm, and 225 cm from the bank at each sampling station. Chironomids in the surface water were sampled using a pond net with an opening of 20 by 30 cm and 0.5 mm mesh size. A sample consisted of 2 sweeps over a length of approximately 2 m in open water just above the sediment, one sweep starting from the open water towards the bank, and one sweep parallel to the bank. In the laboratory all samples were washed over 3 sieves with 2, 1, and 0.5 mm mesh size to facilitate sorting. Identification was done to the lowest taxonomic level possible (usually species level).

Environmental factors

The following parameters were measured in the field: electrical conductivity (Hanna Combo meter), thickness of the sludge layer (measured by pushing a plastic rod into the organic matter of the sediment), transparency (Secchi depth), and an estimation of the percentage cover by submerged vegetation (e.g. *Elodea nuttallii*, *Ceratophyllum demersum*), nymphaeid vegetation (e.g. *Nuphar lutea*, *Nymphaea alba*), filamentous algae, floating algae beds (FLAB), and lemnid vegetation (*Lemna* sp.). Water bodies that were transparent down to the bottom were assigned a Secchi depth of 1 m, because a lot of water bodies were not deeper than 1 m and Secchi depth could not be measured beyond the bottom. Two surface water samples and 3 sediment samples were taken for further analysis. The following ions were measured colorimetrically on surface water samples (Auto Analyzer 3, Digital colorimeter, Bran + Luebbe, Germany): NO_3^- according to Kamphake et al. (1967), NH_4^+ according to Grasshoff and Johannsen (1972), PO_4^{3-} according to Henriksen (1965), and Cl^- according to O'Brien (1962). Total-P, total-S, and total-Fe were measured by inductively coupled plasma mass spectrometry (Thermo Electron Corporation, United Kingdom). Sediment samples were dried for 24 hours at 100°C and grain size was determined with a Coulter LS 230 laser diffraction device (Beckman Coulter, Inc, Fullerton CA, USA). Organic content was determined by ashing the dried samples for 4 hours at 550°C. Carbon and nitrogen content of the sediment were measured with a Carbo Erba NA 1500 Nitrogen Carbon Sulfur Analyzer. All physico-chemical factors were measured at least in duplicate and average values of duplicates and the 2 seasons were used for the data analysis.

Multivariate analysis

For data analysis chironomids from the net and core sampling and the 2 seasons were pooled by adding all data, to obtain a chironomid assemblage per location. Chironomid associations were distinguished by Two Way Indicator Species Analysis (TWINSPAN; Hill, 1979) on abundance data. If the dissimilarity between macroinvertebrate associations was larger than the dissimilarity within the macroinvertebrate associations, they were regarded as different associations (Verberk et al., 2006). Two multivariate techniques (Detrended Correspondence Analysis and BIOENV) were used to relate chironomid associations to environmental factors.

Canoco for Windows Version 4.0 (Ter Braak & Šmilauer, 1998) was used to perform Detrended Correspondence Analysis (DCA). An indirect method was chosen, because the prime interest is the variation between chironomid species associations (Jongman et al., 1995). A unimodal response model was chosen, because there was a broad gradient in environmental factors. Furthermore detrended analysis was used to remove the arch effect. Before analysis chironomid abundances were transformed according to Preston (1962): Preston class = $\log_2$ (abundance+1). Bray-Curtis similarity (Bray & Curtis, 1957) of individual locations with those of all locations within a chironomid association was calculated to distinguish if the chironomid assemblage of a location was more similar to one chironomid association than another. In this way we could evaluate if the chironomid assemblage of a location shifted from one chironomid association to another in the following years.

Some environmental factors were deleted prior to analysis, because their values varied too little in the study area to be relevant for chironomids and including these factors did not assist interpretation of the results. Effects of pH differences were not expected between the relatively neutral and slightly alkaline values (pH 7.1 - 9.2) and were therefore not included in the analysis. Aluminum, copper, and lead concentrations complied with ecological quality standards in normal weather conditions (Vermonden et al., 2009b) and were therefore excluded from the analysis. Environmental factors with wide ranges (NO_3^-, NH_4^+, PO_4^{3-}, Cl^-, total P, C/N content sediment) were log-transformed. An ordinal scale was used for the cover by submerged, nymphaeid, lemnid vegetation, and filamentous algae. Within filamentous algae a distinction was made between filamentous algae in the water column and filamentous algae beds floating on the surface water (FLAB). The percentage of clay and silt fraction (0-64 µm) was used as a measure of sediment composition. Significance of environmental factors was tested with Canonical Correspondence Analysis (CCA), using 500 Monte Carlo permutations under full model conditions. Differences of environmental factors between chironomid assemblages were tested with Kruskal-Wallis followed by a Mann-Whitney test ($p<0.05$). Spearman's Rho correlations between environmental factors and chironomid assemblages were examined with the BIOENV procedure (Clarke & Ainsworth, 1993) in the software package PRIMER 5 (Clarke & Warwick, 1994).

Biodiversity indices

We compared biodiversity indices of urban waters in this study with those of other drainage systems by selecting 20 canals and ditches from the national database in the Netherlands held by the water boards (STOWA, 2006). Canals and ditches were chosen to represent man-made drainage systems in rural areas. Locations were only chosen if chironomids were identified to the same taxonomic level as in the urban water systems. The locations were situated across The Netherlands within a radius of 120 km of the urban water systems. Furthermore, locations were selected that were monitored once in August or September in the period 2000-2005. To compare the rural areas with the urban water systems, data of the urban water systems was used of the period August-September 2007. Comparisons were based on the net samples only, because STOWA (2006) did not include benthic samples. Differences of average biodiversity indices between urban and rural drainage systems were determined with a Kruskal-Wallis test ($p<0.05$).

Taxon richness was expressed as the number of taxa in each location. The Shannon index (H') was calculated according to Shannon (1948) using natural logarithms. Based on Foster et al. (1990), a score for rareness was calculated for each location. Each species or genus was assigned a score according to its rareness in the Netherlands over the period 2000-2005 (STOWA, 2006). The species were given a score between 1 and 6, corresponding to > 2,000, 1,000-2,000, 500-1,000, 200-500, 50-200, and <50 occurrences in the national database, respectively. The rareness score of a location was calculated as the sum of all species rareness scores of the location divided by the number of species present (Vermonden et al. 2009a). In this way rareness of chironomid species is expressed as the rareness of all species in a location according to their occurrence in the Netherlands.

The change of biodiversity indices in the urban chironomid assemblages in Nijmegen in the period 2005-2007 was tested with a Kruskal-Wallis test ($p<0.05$). Changes in Arnhem in the period 2006-2007 was tested with a Mann-Whitney test ($p<0.05$). Kruskal-Wallis was used to test differences between more than 2 groups (3 different years in Nijmegen). Mann-Whitney was used to test differences between 2 groups (2 different years in Arnhem). In Nijmegen a distinction was made between locations that where dredged in the winter of 2005-2006 and locations where no dredging took place. In Arnhem all locations where dredged between 2003 and 2005.

Spearman's Rho correlation between the environmental factors and taxon richness, Shannon index, and rareness were calculated for all urban water locations sampled in the period 2005-2007.

Life-history strategies

Seventy-four taxa were assigned a life-history strategy using biological traits, extracted from Moog (2002), Vallenduuk & Moller Pillot (2007), Moller Pillot (2009), and personal communication with H.K.M. Moller Pillot. Eight taxa could not be assigned a strategy, because there was too little information on their traits. Vermonden et al. (2009a) showed

Table 1 Chironomid life-history strategies (LHS) and their associated traits in urban water systems. HS, H, S, V and HV are explained in Figure 1. No data on traits was available for taxa in U (unknown). Average number of generations and average maximum length of larvae was calculated from table 3. Data on life traits was extracted from Moog (2002), Vallenduuk & Moller Pillot (2007), Moller Pillot (2009) and personal communication with H.K.M. Moller Pillot.

LHS	Number of taxa	Haemoglobin	Spring synchronisation	Average number of generations	Average maximum length larvae (mm)
HS	38	+	+	2.2	13.0
H	15	+	-	2.2	7.6
S	6	-	+	2.2	8.0
V	13	-	-	2.5	7.2
HV	2	-	-	5.0	4.0
U	8				

that nutrients, sediment composition, transparency, and abundance of vegetation were important environmental factors for macroinvertebrate species diversity and composition. The following traits were therefore expected to be most relevant for chironomids in urban waters and were used to assign life-history strategies to species: haemoglobin, spring synchronisation, voltinism, and larval size (Table 1, Figure 1).

Chironomid larvae can be tolerant to poorly oxidized conditions, because of the possession of haemoglobin (Pinder, 1986). Haemoglobin in chironomids has a high affinity for oxygen and allows rapid release of oxygen when needed. Spring synchronisation allows species to emerge early in the season, and to reproduce and develop into the final instar at the end of spring, thus avoiding oxygen stress during the summer. Voltinism refers to the number of generations in one year (Armitage et al., 1995). Chironomids can be uni-, bi- or multivoltine, corresponding with 1, 2, 3 or more generations per year, respectively. Larval size and voltinism often form a trade-off, because multivoltine species do not have enough time to grow to a large size. From these trades and trade-offs we assigned chironomids to life-history strategies.

Species of life-history strategy HS have haemoglobin and spring synchronisation (Table 1, Figure 1). This allows them to remain active during harsh periods and grow to a

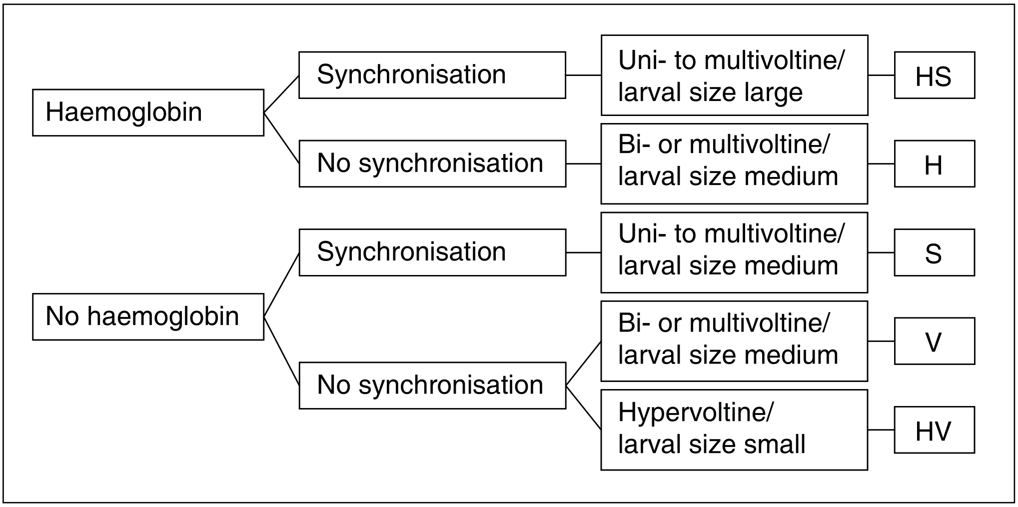

Figure 1 Overview of the different life-history strategies and their defining traits.

large size. Larval size corresponds with adult body size, which in turn is positively correlated with reproductive effort. These traits and phenology make these species especially resistant to oxygen stress. Species of life-history strategy H also have haemoglobin, but lack spring synchronisation and are medium-size. In life-history strategy S, species have spring synchronisation, resulting in fewer generations per year. These species lack haemoglobin. In life-history strategy V, species have no haemoglobin or spring synchronisation. The absence of spring synchronisation and a relatively small larval size allow species to produce more generations than species of most other strategies. Species of life-history strategy HV minimize the age of reproduction and consequently produce many generations per year. Their small body size allows a rapid larval development. They do not invest time and energy in the construction of tubes. Species in this group can also reproduce asexually, which does not require time investment in mate searching or mating.

The relative abundance (abundance of a strategy divided by the total abundance of all strategies multiplied by 100%) of the strategies was compared between urban chironomid associations and between chironomid assemblages in urban and rural areas with Bray-Curtis similarity (Bray & Curtis, 1957). ANOSIM was used to test significant differences between the urban chironomid associations and between the urban and rural assemblages (Clarke & Warwick, 1994). Differences of strategies in the period 2005-2007 were tested with a Kruskal-Wallis test (differences between more than 2 groups, $p<0.05$) for locations in Nijmegen and with a Mann-Whitney test (differences between 2 groups, $p<0.05$) for locations in Arnhem in the period 2006-2007.

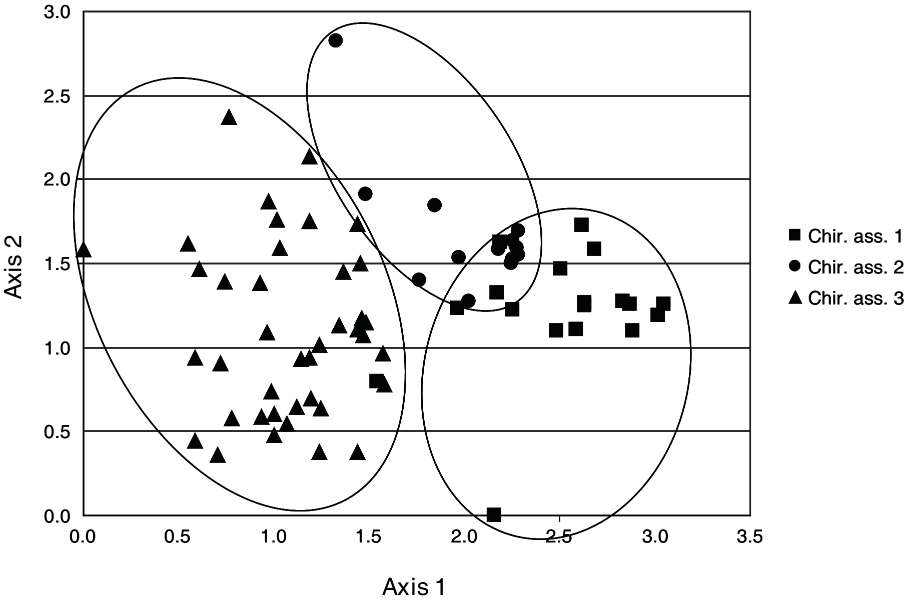

Figure 2a Detrended Correspondence Analysis of chironomid taxa in the period 2005-2007. Symbols represent different chironomid associations, distinguished with TWINSPAN (Hill, 1979).

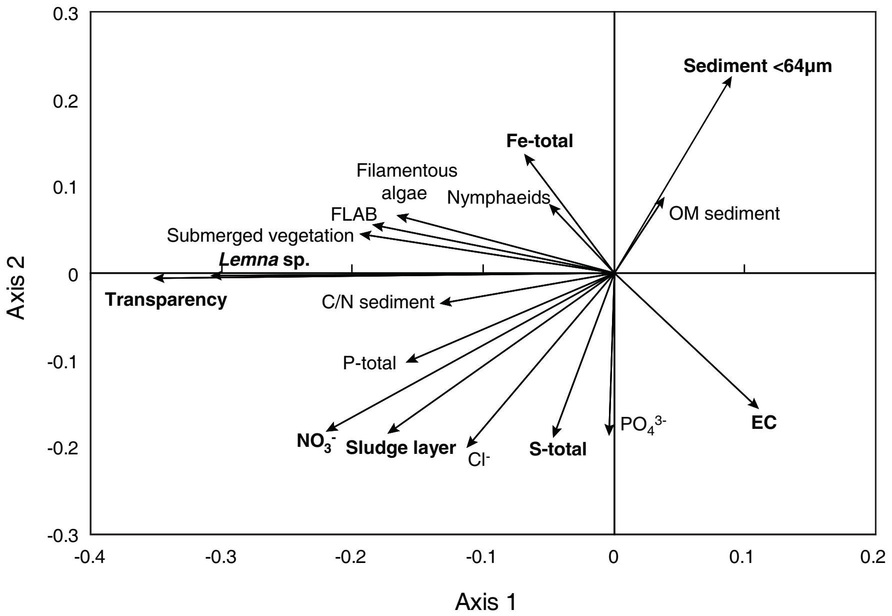

Figure 2b Detrended Correspondence biplot of environmental factors related to chironomid assemblages. Environmental factors in bold significantly explain variation in chironomid assemblages (Canonical Correspondence Analysis, 500 Monte Carlo permutations, p<0.05).

Results

Chironomid associations and assemblages

Three different chironomid associations could be distinguished, based on taxon composition and abundances (Figure 2a). CCA showed that nitrate, iron, sulfur, electrical conductivity, grain size (sediment composition), sludge layer, cover by lemnids, and transparency explained a significant proportion of the variation within and between the chironomid associations in urban water systems (Figure 2b).

The first chironomid association was typical for relatively low nutrient levels, sand sediment (>50% of grain size >64μm), relatively turbid surface water, and poorly developed vegetation (Table 2). Based on frequency and abundance of chironomids we found that the following species were characteristic of the first chironomid association: *Cryptochironomus defectus*, *Glyptotendipes paripes*, *Microtendipes pedellus* agg., *Chironomus muratensis*, *Cladopelma* gr. *viridulum*, *Tribelos intextum*, and *Cladotanytarsus* sp. (Table 3). The second chironomid association was typical for relatively low nutrient levels, clay sediment (>50% of grain size <64μm), and relatively turbid surface water. Characteristic species of the second chironomid association were: *Polypedilum sordens*, *Nanocladius bicolor*, *Parachironomus biannulatus*, *Parachironomus* gr. *arcuatus*, *Dicrotendipes lobiger*, *Endochironomus tendens*, and *Polypedilum tritum/ uncinatum*. The third chironomid association was typical for water bodies with the highest nutrient and sulfur levels, sand sediment, clear surface water, high abundance of *Lemna* sp. and filamentous algae, and a relatively thick sludge layer. Characteristic species of the third chironomid association were: *Dicrotendipes notatus*, *Acricotopus lucens*, and *Psectrotanypus varius*.

Table 2 Average values of environmental factors (minimum and maximum values in brackets) of chironomid associations. a and b indicate statistical significance of differences (bold, Kruskal-Wallis followed by a Mann-Whitney test p<0.05), using the same letters for factors of water systems that do not significantly differ and different letters for factors that are significantly different.

	Chir. ass. 1 (n=18)	Chir. ass. 2 (n=13)	Chir. ass. 3 (n=40)
NO_3^- (mg l^{-1})	**2.4 (0.3-12.6)[a]**	**2.0 (0.4-4.1)[a]**	**7.2 (0.4-23.4)[b]**
NH_4^+ (mg l^{-1})	0.15 (0.04-0.53)[a]	0.22 (0.06-0.54)[a]	0.15 (0.03-0.39)[a]
PO_4^{3-} (mg l^{-1})	**0.06 (0.02-0.23)[a]**	**0.06 (0.02-0.22)[a]**	**0.13 (0.02-0.45)[b]**
Total-P (mg l^{-1})	**0.03 (0.01-0.09)[a]**	**0.03 (0.01-0.08)[a]**	**0.05 (0.01-0.16)[b]**
Total-S (mg l^{-1})	**13.7 (6.9-18.7)[a]**	**12.6 (2.4-16.8)[a]**	**15.4 (3.3-20.3)[b]**
Cl^- (mg l^{-1})	**43.1 (28.1-53.6)[a]**	**38.4 (18.4-52.5)[a]**	**48.1 (30.2-76.4)[b]**
EC (µS m^{-1})	534 (440-631)[a]	542 (213-629)[a]	523 (301-704)[a]
Total-Fe (mg l^{-1})	0.06 (0.03-0.16)[a]	0.09 (0.03-0.26)[a]	0.08 (0.01-0.24)[a]
Sediment <64 µm (%)	**21.3 (2.7-88.9)[a]**	**85.0 (33.4-97.0)[b]**	**19.7 (0.2-95.4)[a]**
C/N content sediment	12.0 (6.5-25.4)[a]	13.9 (9.9-17.1)[a]	14.2 (7.7-35.1)[a]
Organic content sediment (%)	**3.0 (0.5-10.8)[a]**	**4.9 (1.6-7.6)[b]**	**2.7 (0.5-12.9)[a]**
Sludge layer (thickness in cm)	**0.4 (0.0-2.5)[a]**	**0.1 (0.0-1.0)[a]**	**1.0 (0.0-3.0)[b]**
Transparency (m)	**0.6 (0.3-1.0)[a]**	**0.6 (0.3-1.0)[a]**	**0.9 (0.6-1.0)[b]**
Submerged vegetation[c]	**0.3 (0.0-2.0)[a]**	**1.5 (0.5-3.0)[b]**	**1.3 (0.0-3.0)[b]**
Nymphaeid vegetation[d]	**0.0 (0.0-0.0)[a]**	**0.3 (0.0-1.0)[ab]**	**0.2 (0.0-1.0)[b]**
***Lemna* sp.[c]**	**0.0 (0.0-0.5)[a]**	**0.2 (0.0-1.0)[a]**	**0.7 (0.0-3.0)[b]**
Filamentous algae[c]	**0.1 (0.0-1.0)[a]**	**0.9 (0.0-2.5)[b]**	**0.8 (0.0-2.5)[b]**
FLAB[c]	**0.0 (0.0-0.5)[a]**	**0.3 (0.0-1.0)[ab]**	**0.4 (0.0-2.5)[b]**

[c] Category scale: 0=absent, 1=<10%, 2=10-50%, 3=>50% cover;
[d] Binary scale: 0=absent, 1=present

Table 3 Frequency of occurrence and average relative abundance of chironomid species within the 3 chironomid associations. H: haemoglobin, S: synchronization, V: number of generations, L: maximum length of larvae and LHS: life-history strategy, according to figure 1 (U = unknown). Data on life traits was extracted from Moog (2002), Vallenduuk & Moller Pillot (2007), Moller Pillot (2009) and personal communication with H.K.M. Moller Pillot.

	Chir.ass. 1 (n=18)		Chir. ass. 2 (n=13)		Chir. ass. 3 (n=40)		H	S	V	L (mm)	LHS
	Freq.	Abun.	Freq.	Abun.	Freq.	Abun.					
Cladotanytarsus sp.	72.2	20.0	7.7	1.0	12.5	2.2	+	-	2.5	6	H
Tribelos intextum	11.1	1.0	0	0	0	0	+	+	1.0	10	HS
Microtendipes pedellus agg.	16.7	1.0	0	0	0	0	+	+	2.0	9	HS
Glyptotendipes paripes	55.6	6.3	0	0	0	0	+	+	2.5	U	HS
Glyptotendipes ospeli	5.6	1.0	0	0	0	0	+	+	2.0	10	HS
Endochironomus lepidus	5.6	1.0	0	0	0	0	+	+	2.0	14	HS
Cryptochironomus obreptans	5.6	1.0	0	0	0	0	+	+	2.0	12	HS
Cladopelma gr. *viridulum*	16.7	5.7	0	0	0	0	+	-	2.0	5	H
Chironomus muratensis	50.0	6.8	0	0	0	0	+	+	3.0	20	HS
Paracladius conversus	5.6	1.0	0	0	0	0	U	U	U	9	U
Cryptochironomus defectus	55.6	1.7	0	0	0	0	+	+	2.0	11	HS
Cricotopus intersectus agg.	22.2	11.0	0	0	5.0	3.5	-	-	2.5	5	V
Polypedilum bicrenatum	22.2	5.3	23.1	2.0	10.0	1.3	+	-	2.0	6	H
Chironomus plumosus agg.	44.4	3.5	23.1	7.0	12.5	3.4	+	+	2.5	24	HS
Polypedilum sordens	44.4	7.5	69.2	57.0	5.0	2.5	+	-	2.5	8	H
Glyptotendipes pallens agg.	88.9	6.1	92.3	21.7	7.5	2.7	+	+	2.5	14	HS
Endochironomus albipennis	83.3	9.4	76.9	130.3	5.0	1.0	-	+	2.5	11	HS
Chironomus acutiventris	5.6	1.0	7.7	1.0	0	0	+	+	2.5	U	HS
Nanocladius bicolor agg.	16.7	3.3	53.8	5.7	2.5	1.0	U	U	U	5	U
Parachironomus biannulatus	5.6	1.0	23.1	2.7	2.5	2.0	-	-	2.0	7	V
Glyptotendipes gripekoveni	0	0	7.7	2.0	0	0	+	+	2.0	11	HS
Cryptochironomus supplicans	5.6	1.0	15.4	1.0	0	0	+	+	2.5	12	HS
Psectrocladius obvius	5.6	2.0	7.7	3.0	2.5	2.0	-	+	2.5	12	S
Parachironomus gr. *arcuatus*	50.0	1.8	84.6	13.6	25.0	3.5	-	-	2.0	7	V

Table 3 Continued

	Chir.ass. 1 (n=18)		Chir. ass. 2 (n=13)		Chir. ass. 3 (n=40)		H	S	V	L (mm)	LHS
	Freq.	Abun.	Freq.	Abun.	Freq.	Abun.					
Dicrotendipes nervosus	50.0	2.9	76.9	7.7	25.0	8.2	+	-	2.5	9	H
Dicrotendipes lobiger	5.6	1.0	30.8	2.3	7.5	1.3	+	-	2.0	8	H
Polypedilum tritum/ uncinatum	0	0	53.8	6.1	17.5	4.7	+	+	2.5	8	HS
Endochironomus tendens	16.7	1.7	69.2	22.3	17.5	11.1	+	+	2.0	12	HS
Dicrotendipes pulsus	5.6	2.0	23.1	11.0	2.5	17.0	+	-	2.0	8	H
Guttipelopia guttipennis	5.6	1.0	23.1	2.7	15.0	1.8	U	U	U	10	U
Zavreliella marmorata	0	0	7.7	1.0	2.5	1.0	-	+	2.5	4	S
Glyptotendipes signatus	0	0	7.7	1.0	2.5	1.0	+	-	1.5	8	H
Clinotanypus nervosus	27.8	2.6	38.5	1.4	15.0	2.3	+	+	1.0	13	HS
Polypedilum nubeculosum	66.7	6.8	61.5	18.6	42.5	16.7	+	+	2.5	8	HS
Cladopelma gr. *goetghebueri*	27.8	1.2	7.7	1.0	10.0	2.0	-	-	2.0	7	V
Chironomus obtusidens	5.6	1.0	0	0	2.5	1.0	+	+	2.0	15	HS
Cricotopus gr. *sylvestris*	100.0	88.2	100.0	50.5	87.5	65.6	-	-	3.5	7	V
Tanytarsus sp.	83.3	4.8	69.2	3.2	67.5	10.2	+	-	2.5	8	H
Kiefferulus tendipediformis	0	0	7.7	9.0	5.0	1.0	+	+	2.0	8	HS
Chironomus commutatus	27.8	15.8	38.5	3.8	30.0	5.3	+	+	2.0	13	HS
Procladius sp.	66.7	5.8	61.5	5.0	65.0	8.3	-	+	2.0	10	S
Prodiamesa olivacea	5.6	1.0	0	0	5.0	1.0	U	U	U	15	U
Stictochironomus sp.	11.1	1.5	0	0	2.5	3.0	U	U	U	14	U
Psectrocladius gr. *limb/sordidellus*	44.4	13.9	61.5	5.9	52.5	14.5	-	-	3.5	8	V
Paramerina cingulata	0	0	38.5	3.6	22.5	11.1	-	-	3.0	6	V
Ablabesmyia sp.	22.2	1.0	53.8	7.6	40.0	3.3	-	-	2.0	11	V
Microtendipes chloris agg.	44.4	2.8	53.8	5.3	45.0	6.4	+	+	2.5	9	HS
Chironomus nuditarsis	16.7	5.7	30.8	8.3	27.5	8.3	+	+	2.5	20	HS
Orthocladius sp.	5.6	1.0	7.7	3.0	7.5	12.0	U	U	U	8	U
Glyptotendipes imbecillis	0	0	7.7	1.0	2.5	9.0	+	+	2.0	8	HS
Dicrotendipes modestus	0	0	7.7	2.0	2.5	6.0	+	-	2.0	10	H
Xenopelopia sp.	0	0	15.4	2.5	17.5	2.9	-	+	2.5	8	S
Paratanytarsus sp.	44.4	7.4	61.5	37.9	75.0	35.1	-	+	3.0	7	S
Polypedilum cultellatum	0	0	7.7	4.0	22.5	3.9	+	-	2.5	U	H
Phaenospectra sp.	5.6	1.0	7.7	1.0	17.5	1.9	+	-	2.0	8	H
Paratendipes albimanus	11.1	2.0	0	0	22.5	2.0	+	-	1.5	6	H
Dicrotendipes notatus	11.1	4.5	23.1	1.7	52.5	10.8	+	-	2.5	8	H
Chironomus luridus agg.	22.2	6.0	15.4	7.0	37.5	18.1	+	+	2.0	15	HS
Monopelopia tennicalcar	5.6	1.0	7.7	2.0	12.5	5.8	-	-	2.0	6	V
Corynoneura scutellata agg.	33.3	3.0	38.5	2.0	45.0	21.2	-	-	4.5	4	HV
Chironomus riparius agg.	0	0	7.7	1.0	7.5	5.3	+	+	4.0	15	HS
Chironomus annularius agg.	22.2	2.3	23.1	6.0	45.0	13.9	+	+	3.0	13	HS
Chironomus tentans	0	0	0	0	5.0	1.5	+	+	3.0	22	HS
Psectrotanypus varius	0	0	15.4	1.5	35.0	7.2	-	-	3.0	11	V
Cricotopus gr. *fuscus*	16.7	1.3	0	0	20.0	21.3	-	-	3.0	8	V
Tanypus sp.	5.6	2.0	7.7	3.0	35.0	3.0	-	-	2.0	11	V
Acricotopus lucens	0	0	15.4	1.0	40.0	8.5	-	-	3.0	7	V
Anatopynia plumipes	0	0	0	0	5.0	3.0	+	+	1.0	18	HS
Macropelopia nebulosa	0	0	0	0	2.5	1.0	+	+	2.0	12	HS
Chaetocladius piger agg.	0	0	0	0	2.5	3.0	-	+	1.0	7	S
Pseudochironomus prasinatus	0	0	0	0	2.5	1.0	+	+	1.5	8	HS
Einfeldia pagana	0	0	0	0	5.0	10.0	+	+	U	12	HS
Microspectra sp.	0	0	0	0	10.0	3.0	+	-	3.0	10	H
Stenochironomus sp.	0	0	0	0	2.5	1.0	+	+	2.0	18	HS
Chironomus tentans/pallidivittatus	0	0	0	0	2.5	5.0	+	+	3.0	22	HS
Chironomus longipes	0	0	0	0	5.0	2.0	+	+	2.0	11	HS
Chironomus cingulatus	0	0	0	0	5.0	5.5	+	+	2.5	15	HS
Limnophyes sp.	0	0	0	0	5.0	1.0	-	-	5.5	4	HV
Stempellinella sp.	0	0	0	0	2.5	1.0	U	U	U	4	U
Einfeldia dissidens	0	0	0	0	5.0	1.0	+	+	1.5	12	HS
Conchapelopia agg.	0	0	0	0	2.5	1.0	U	U	U	7	U
Endochironomus gr. *dispar*	0	0	0	0	2.5	1.0	+	+	2.0	15	HS

Spearman rank correlations of environmental factors (BIOENV procedure) showed that lemnids, transparency, total sulfur concentration, submerged vegetation, and FLAB in combination correlated significantly with chironomid assemblages (R=0.44). Lemnids was the variable best explaining the species data alone (R=0.36).

Biodiversity indices

Taxon richness was highest in the locations with clay sediment (chironomid association 2), intermediate in the vegetation-poor locations (chironomid association 1), and lowest in the nutrient-rich locations (chironomid association 3, Figure 3a). Values for Shannon-index and rareness did not differ significantly between chironomid associations (Figure 3b-c). Values for taxon richness and Shannon-index were not significantly different between sites in Nijmegen, Arnhem or rural areas (Figure 3d-e). Chironomid rareness was significantly higher in the urban sites, compared to the rural sites (Figure 3f). Out of 18 environmental factors only sludge layer and lemnid abundance showed significant Spearman's Rho correlations with taxon richness ($R=-0.35$ and $R=-0.36$, respectively, $p<0.01$). Shannon-index was also negatively related to lemnid abundance ($R=-0.26$, $p<0.05$). Chironomid rareness was not significantly related to any environmental factors.

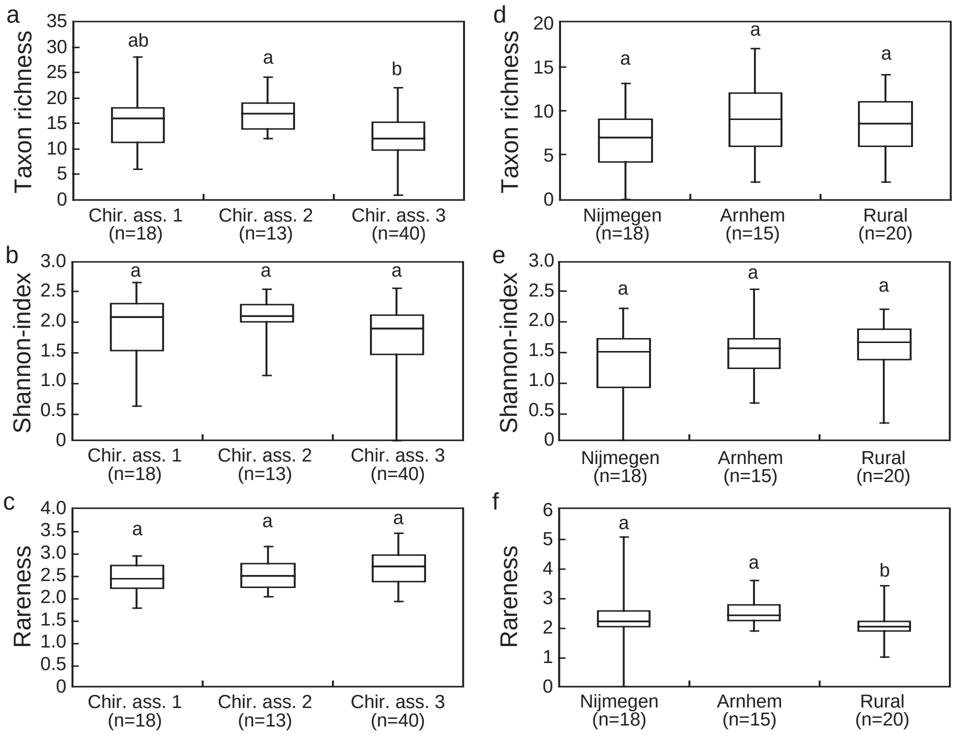

Figure 3 Box plots of biodiversity indices for chironomid associations (a-c) and sampling sites in urban (data net samples Nijmegen and Arnhem, August-September 2007) and rural areas (data STOWA, 2006) (d-f). The line in the middle of the box represents the median, the lower and upper edge of the box the 25th and 75th percentile, and the error bars indicate the minimum and maximum values. Significant differences between water body types are indicated by a and b (Kruskal-Wallis test, followed by a Mann-Whitney test, $p<0.05$), using the same letters for water systems that are not significant different and different letters for systems that are significantly different.

Short-term changes and effects of dredging

Six out of 18 locations shifted from one chironomid association to another after dredging, while only one location out of 12 locations shifted, where no measures took place (Figure 4). Taxon richness was significantly higher in 2007 for locations that were dredged in previous years (Figure 5). Shannon-Index gave the same trends, but years were not significantly different. Rareness did not change significantly over the period 2005-2007, except for the chironomid assemblages in the dredged locations in Nijmegen, where assemblages had a lower score for rareness in 2006, than in 2005 and 2007.

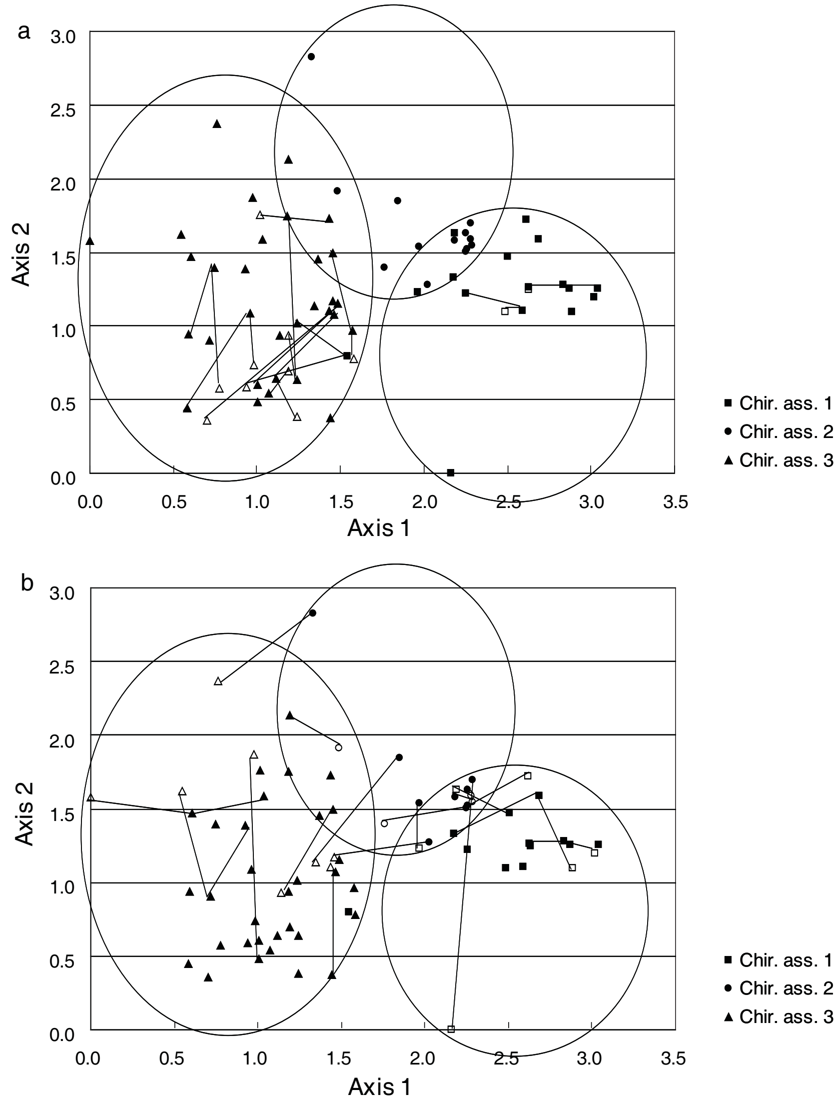

Figure 4 Detrended Correspondence Analysis of chironomid taxa in the period 2005-2007 in the locations of Nijmegen, where no rehabilitation measures took place (a), the dredged locations of Nijmegen and Arnhem (b). Lines indicate the shifts from 2005 (open symbols) to 2006 and 2007 (closed symbols) or shifts from 2006 to 2007 (Arnhem locations).

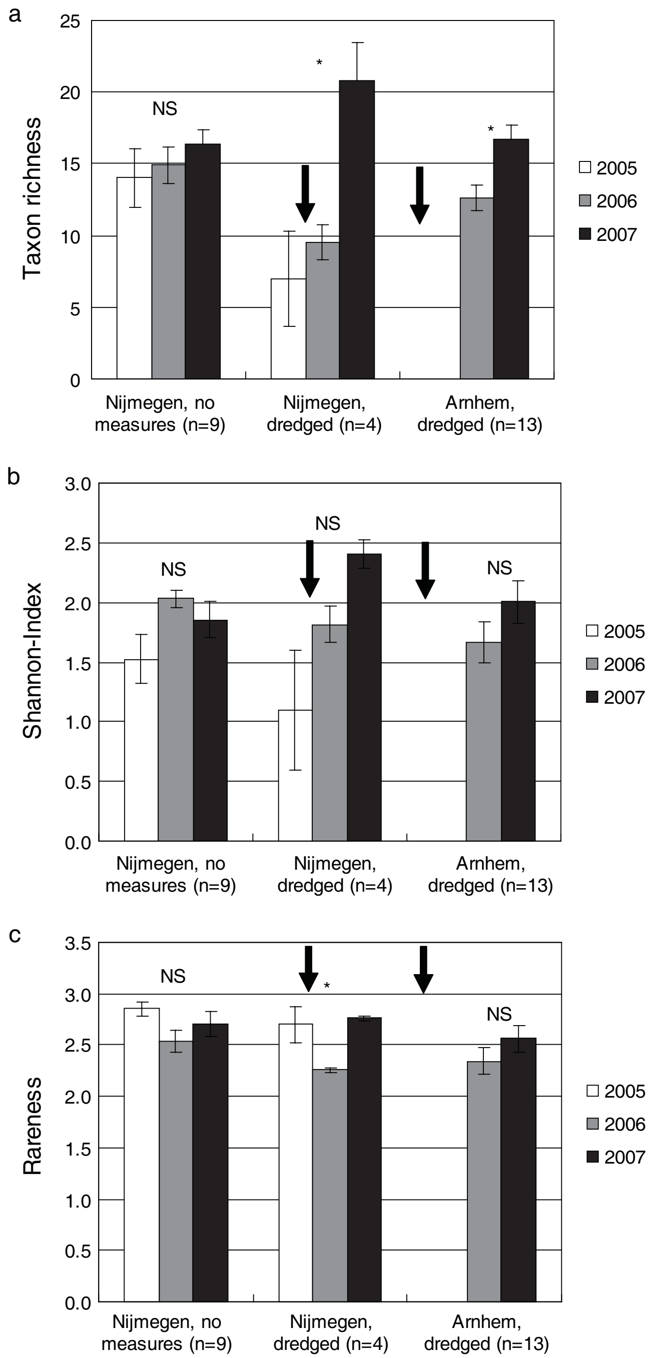

Figure 5 Average values of taxon richness (a), Shannon-index (b) and rareness (c) in Arnhem and Nijmegen in the period 2005-2007. Error bars represent the standard error. Arrows indicate dredging. Asterisks indicate significant differences (Kruskal-Wallis test; Nijmegen, Mann-Whitney test; Arnhem: NS: p>0.05, *:p<0.05).

Life-history strategies

Relative abundances of life-history strategies were not significantly different between urban and rural sites or between chironomid associations within the urban area (Figure 6). In Nijmegen at locations where no dredging measures took place, life-history strategies H (haemoglobin, no synchronisation) increased in the chironomid assemblages from 2005 to 2006 and decreased from 2006 to 2007 (Table 4). In the chironomid assemblages of the dredged locations of Nijmegen an increase was found of the life-history strategy V (bi-/multivoltine, medium sized). In the chironomid assemblages of the dredged locations of Arnhem life-history strategy HS (haemoglobin, spring synchronisation) became more abundant and strategy V (bi-/multivoltine, medium sized) less abundant.

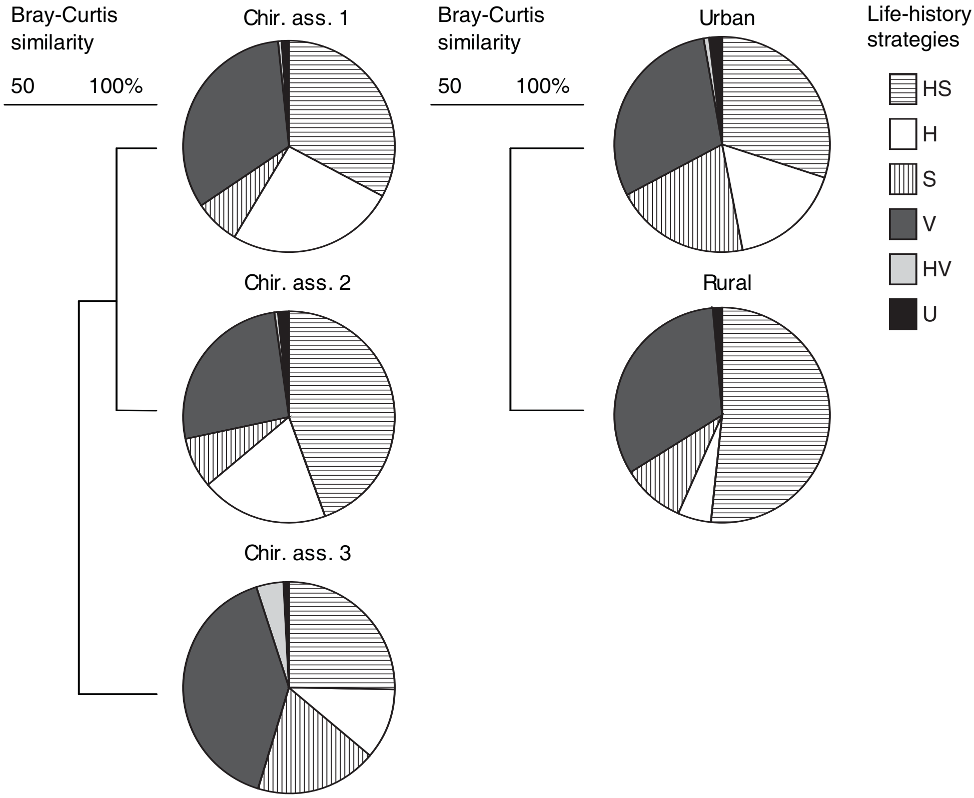

Figure 6 Average relative abundance and Bray-Curtis similarity of the life-history strategies of taxa in the 3 urban chironomid associations and urban (data net samples Nijmegen and Arnhem, August-September 2007) and rural areas (data STOWA, 2006). Life-history strategies HS, H, S, V and HV are explained in Figure 1. No data on biological traits was available for taxa in U (unknown).

Table 4 Average relative abundance of the life-history strategy in Nijmegen and Arnhem in the period 2005-2007. Numbers in bold indicate significant differences between years (Kruskal-Wallis test; Nijmegen, Mann-Whitney test; Arnhem): p<0.05. Life-history strategies M, HS, H, S, V and HV are explained in Figure 1. No data on traits was available for taxa in U (unknown).

		Life-history strategies					
Relative abundance (%)		HS	H	S	V	HV	U
Nijmegen, no dredging	2005	17.9	**11.3**	20.3	49.7	0.4	0.4
	2006	20.4	**32.5**	22.3	22.8	1.6	0.5
	2007	17.9	**9.0**	18.4	48.4	5.4	0.9
Nijmegen, dredged	2005	54.1	28.3	0.8	**4.3**	12.5	0.0
	2006	51.4	23.3	7.4	**17.9**	0.0	0.0
	2007	50.2	9.3	9.4	**29.8**	1.0	0.3
Arnhem, dredged	2006	**20.8**	13.9	6.7	**56.6**	1.0	1.0
	2007	**53.9**	13.3	7.0	**23.6**	0.6	1.6

Discussion

Chironomid associations and relationships with environmental factors

Three different chironomid associations were distinguished based on taxa and their abundances (Figure 2a). Analyses indicate that substrate (sludge layer and substrate type: sand vs. clay), concentration of sulfur, nitrate, total iron, electrical conductivity, lemnid abundance, and transparency are key factors for structuring chironomid assemblages (Figure 2b). Similar results were found in other water systems. Syrovátka et al. (2009) found that particulate organic matter, aquatic vegetation, near-bottom current velocity, and substrate roughness were the key factors determining the chironomid associations in a simple, channeled stream in the Czech Republic. Substrate type, current velocity, and aquatic vegetation were the major explanatory factors structuring chironomid associations in a river basin in Argentina (Principe et al., 2008). Carew et al. (2007) found that salinity, substrate type, submerged, and riparian vegetation were particularly important for the distribution of specific chironomid taxa in urban wetlands. Current velocity (on average 3-14 cm s^{-1}) and salinity varied very little in our study.

Environmental factors are interrelated and these factors in combination influence the chironomids directly or indirectly. For example, high concentrations of nutrients stimulate the growth of algae and lemnids, increasing organic loading and the accumulation of sludge (Parr & Mason, 2004). Decomposition of the organic matter in the sludge layer might subsequently result in higher levels of nutrients and sulfur in the water layer (Wetzel, 2001). Chironomids are probably not directly influenced by nutrients, but the latter influences the availability and quality of food from the organic matter, which are important for determining the chironomid associations (Armitage et al., 1995).

Secchi-depth (transparency) is positively related to submerged vegetation (Scheffer et al., 1993), because vegetation reduces resuspension, and suppresses algal growth due to a reduction of nutrient availability. Transparency also stimulates growth of submerged vegetation. Submerged vegetation can be used by chironomids as a food source (decaying plant tissue, plant derived detritus, epiphyton), substrate or for mining (Van der Velde & Hiddink, 1987). Vegetation can stabilize a clear-water state up to relatively high nutrient levels (Scheffer et al., 1993).

Life-history strategies can be used to provide insight into how aquatic macroinvertebrates deal with the prevailing environmental conditions. There was little variation of the average relative abundance of life-history strategies between urban and rural areas, and between urban chironomid assemblages (Figure 6). To gain complete insight in the causal relationships between biodiversity and environmental characteristics of urban water systems, the life-history approach has to be applied to other taxonomic groups, too.

Biodiversity indices

Values for chironomid taxon richness and Shannon-index were similar in urban drainage systems and man-made drainage systems in rural areas. Value for rareness was higher in the urban drainage systems, meaning that chironomid species found in urban areas were more unique compared to species found in rural areas. These results do not correspond with other studies. Chironomid taxon richness decreased with total impervious area in the lowland Yarra River in Australia (Walsh et al., 2007). Furthermore, chironomid taxon richness was lowest in the urban streams, when comparing with rural and natural areas in North Carolina Piedmont (Lenat & Crawford, 1994). However, other macroinvertebrate groups (e.g. Ephemeroptera, Gastropoda, Hirudinae, Coleoptera, and Trichoptera) in our study area also had similar values for biodiversity indices in urban and rural areas (Vermonden et al., 2009a).

Taxon richness was negatively correlated to thickness of the sludge layer and percentage cover by lemnids, while Shannon-index was negatively correlated to lemnid abundance. Thickness of sludge layer and cover by lemnids were also negatively related to chironomid taxon richness (Spearman's Rho correlation, $p<0.05$). Growth of lemnids inhibits submerged vegetation by shading and die-off of algae and plant material result in the accumulation of sludge (Parr & Mason, 2004). Oxygen consumption is high due to decomposition of organic material and oxygen production is low due to the lack of submerged vegetation. Chironomids might be limited by the lack of oxygen in the waters with a high abundance of lemnids, and find little suitable habitat by the lack of submerged vegetation.

Short-term changes and effects of dredging

Chironomid assemblages changed significantly in approximately 33% of the dredged locations, while only 8% of the locations shifted where no dredging took place (Figure 4). Chironomid taxon richness increased significantly after dredging. There was also a negative correlation between thickness of the sludge layer and taxon richness. Shannon-index and rareness did not change significantly after dredging.

Increased relative abundance of life-history strategy V (bi-/multivoltine, medium sized) after dredging in Nijmegen indicates reduced anoxic stress. Van Kleef et al. (unpubl. data) also found an increase in this strategy after dredging of softwater lakes. Dredging not only removes the sludge layer, but also the cover by lemnids. The combination of removal of lemnids and sludge should significantly improve oxygen conditions, because both factors play an important role for the oxygen content of the surface water (Parr & Mason, 2004). Changes in the abundance of life-history strategy V (bi-/multivoltine, medium sized) were also positively related to changes in transparency (Van Kleef et al., unpubl. data).

In Arnhem relative abundance of life-history strategy V decreased and HS increased. Sampling sites in Arnhem could only be monitored after dredging and reference values were lacking. Possibly life-history strategy V also increased directly after dredging and decreased again in the years thereafter. This could indicate that the positive effects of dredging on oxygen conditions were only temporarily. The effects of dredging might be temporary, because other factors, such as catchment-scale effects of impervious area (Walsh et al., 2005b), nutrient input of other sources (Vermonden et al., 2009b), and vegetation are not yet optimally managed. The increase of the abundance of species with haemoglobin and synchronization in Arnhem could indicate that vegetation was recovering after dredging, as most of these species filterfeed, while using submerged vegetation as substrate.

Nutrients and sludge layer played a key role for chironomids in urban water systems. Dredging resulted in different chironomid species composition, higher taxon richness, and more species indicative for good oxygen conditions on the short term. Chironomids react rapidly on changes, because they colonize new habitat fast and have several generations per year. The effects of dredging on other taxonomic groups of urban water systems in the middle-long term still need to be investigated, together with the effects of other rehabilitation measures, such as reduction of nutrient input on the catchment scale, optimizing the mowing regime, and development of natural banks.

Implications for urban stream ecology

Previous studies showed that effects of rehabilitation measures in urban streams on biological communities and ecosystem functioning were very limited (Larson et al., 2001, Booth, 2005, Suren & McMurtrie, 2005). It was suggested that these local-scale measures are probably ineffective, because of the catchment-scale effects of impervious area (Walsh et al., 2005b). Catchment scale measures are often expensive, due to the large scale of the measures. Our study showed that local-scale measures such as dredging can be effective, resulting in higher chironomid taxon richness and more species sensitive to anoxic stress. In the study area extensive ecological monitoring had taken place, resulting in clear insights on the key environmental factors for ecology in urban water systems (Vermonden et al., 2009a). Ecological monitoring of local urban water systems is an essential instrument for determining the most optimal rehabilitation possibilities of urban water systems.

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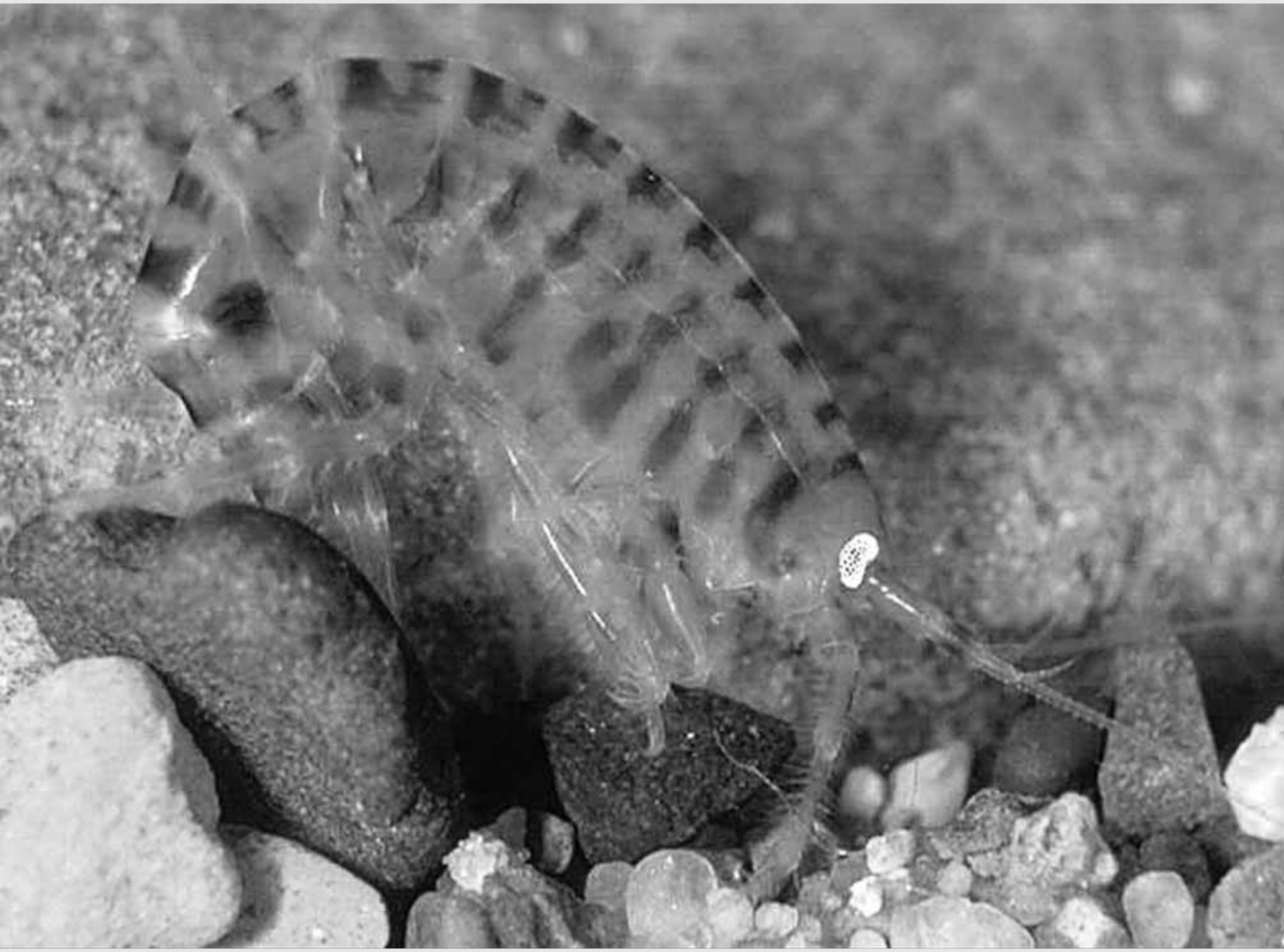

Gammarus tigrinus from the river Waal. Photo: Mariëlle van Riel

Chapter 6

Environmental factors determining invasibility of urban waters for exotic macroinvertebrates

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Abstract

Urbanization usually leads to biotic homogenization with a decrease of native species and increase of exotic species. We investigated whether local environmental factors in urban water bodies, such as water quality, habitat structure and biotic interactions, influenced the invasion of these systems by exotic macroinvertebrate species.

Urban surface water systems were located in lowlands of the Rhine-Meuse delta. Presence and abundance of native and exotic macroinvertebrate species were compared between different urban water types and related to environmental variables with multivariate analysis and Spearman's correlations. Moreover, co-existence of related native and exotic species was studied.

In total nine exotic species were found in the following taxa: Tricladida (1), Crustacea (5), Bivalvia (1) and Gastropoda (2). Taxonomically related native and exotic crustacean species did not seem to be influenced by competition; most species showed high abundances in nutrient-rich waters. Nevertheless, two exotic crustacean species were much more abundant in waters where other crustaceans were absent, indicating that these species filled empty niches. Native species richness and abundance was positively related to environmental heterogeneity in the form of submerged and nymphaeid vegetation. The occurrence and abundance of most exotic species were positively related to several eutrophication indicators, such as nitrate, sludge layer and lemnid vegetation.

Exotic species in urban waters were mostly detritivorous or omnivorous and therefore dependent on leaf breakdown. In the eutrophic water systems detritus was a rich food resource. Invasibility of urban water systems could be reduced by decreasing nutrient levels and stimulating the development of submerged and nymphaeid vegetation in order to increase environmental heterogeneity. Nutrient levels can be lowered by regular dredging, avoiding the inlet of nutrient-rich water and decreasing the (excessive) feeding of water birds and fish

Introduction

In urban areas the abundance of native species is often reduced and the abundance of exotic species is increased, probably due to the predatory impact of exotic species on native species (e.g. Marchetti et al., 2006, McKinney, 2006). Less native amphibians and more exotic crayfish and fish were found in urbanized areas of southern California streams (Riley et al., 2005). Pyšek (1998) found that plant species richness was greater for exotics when compared to native species in central European cities. King & Buckney (2000) observed more exotic plants in the urbanized parts of northern Sydney streams, although native plant diversity was not lower in urban areas than in non-urban, minimally disturbed areas. Exotic species accounted for a greater proportion of total plant species in densely built-up areas of Brussels, Belgium (Godefroid & Koedam, 2007).

Several factors can play an important role in the increase of exotic species in urban areas: high propagule pressure due to human-mediated introduction (Lonsdale, 1999), more favourable habitat created for the establishment of exotic species (Daehler, 2003), various disturbance levels (D'Antonio & Meyerson, 2002), fluctuating resources (Davis et al., 2000, Chytrý et al., 2008), vacant niches due to reduced native species richness (Elton, 1958, Stachowicz et al., 1999) and low habitat heterogeneity (Kestrup & Ricciardi, 2009). Human-mediated introduction can be intentional, for example in the case of ornamental species, or unintentional as by-product of human activities (Van der Velde et al., 2006, Leuven et al., 2009). Roads, streams and connected water networks also enhance exotic species invasions by acting as corridors of agents for dispersal (Parendes & Jones, 2000, Leuven et al., 2009). Exotic species generally perform better under high resources conditions such as high light and nutrient availability (Daehler, 2003). Urban areas are generally enriched in nutrients (e.g. Paul & Meyer, 2001). In addition, exotic species might be the first to recolonize after disturbance (D'Antonio & Meyerson, 2002, Van der Velde et al., 2002), due to invasive species characteristics, such as high dispersal capacity and fecundity (Richardson et al., 2000, Leuven et al., 2009). According to Davis et al. (2000) ecosystems are especially vulnerable to invasions when resource availability fluctuates and invading propagules are readily available. The biotic resistance model states that on the local scale native species richness could prevent invasion, by reducing the available resources (Elton, 1958, Stachowicz et al., 1999, Van der Velde et al., 2006).

Urban water systems could function as potential corridors for the dispersal of exotic species. Previous studies on exotic species in urban areas were mostly focussed on plant species (e.g. Pyšek, 1998, King & Buckney, 2000, Celesti-Grapow et al., 2006, Ehrenfeld, 2008, Botham et al., 2009), fish (Marchetti et al., 2006), fish and amphibians (Riley et al., 2005) or terrestrial arthropods (Plowes et al., 2007). This study focuses on native and exotic aquatic macroinvertebrates in urban water systems. Macroinvertebrates are a key element in water systems, as they process organic material from both autochthonous and terrestrial sources (Vannote et al., 1980) and they form the basic food chain for many species of fish, birds and other wildlife present in towns and cities. Moreover, macroinvertebrate species diversity and abundance is generally high in water systems and species have a varying sensitivity to changing environmental conditions. They are relatively easy to catch (Metcalfe, 1989) and sampling can be standardized.

Vermonden et al. (2009a) distinguished urban water types based on macroinvertebrate species composition, which differed in nutrient levels, vegetation development, transparency and sediment composition. In this study we investigated the invasibility of water bodies belonging to various water types and the relation with local environmental factors, such as water quality and biotic interactions on the establishment of exotic species in these urban water systems.

The following research questions are addressed in this case study:

(1) What is the frequency and abundance of exotic species in various urban water types?

(2) What is the relationship between presence and abundance of exotic macroinvertebrate species and local environmental conditions in urban water systems?

(3) Are richness and abundance of exotic macroinvertebrates related to native macroinvertebrate species richness and abundance in urban water systems?

(4) Do exotic species replace, compete or co-exist with related native species?

Methods

Study area

The towns of Arnhem and Nijmegen are situated in the eastern part of The Netherlands along two distributaries of the River Rhine (Vermonden et al., 2009a). Municipalities of Nijmegen and Arnhem have approximately 2,800 and 1,400 inhabitants per km^2, respectively (Statistics Netherlands, 2009). In the 1970s, urban water systems were designed to regulate groundwater levels and to manage the discharge of urban storm water run-off into the rivers. Approximately 4% of the surface area of these towns consists of watercourses that are connected via culverts. The slow-flowing, permanent watercourses range from small linear ditches to large ponds, generally with a width of between 5 and 40 m and a depth of up to 3 m (Vermonden et al., 2009a). Land use in the study area is predominantly residential, with an impervious area of approximately 30% (roads, buildings, parking lots), while 66% is pervious area (gardens, parks and other green areas) and 4% is surface waters. Separate sanitary sewer systems are constructed in the study area to transport sewage alone (Vermonden et al., 2009b). Sewage is pumped directly to the sewage treatment plant and is not discharged into the urban drainage systems. Another pipe system is designed to convey storm water runoff from impervious areas (e.g. roads and roofs) directly to surface waters.

Sampling

For this study, we selected 11 water bodies in the southern part of Arnhem and 25 water bodies in the western part of Nijmegen. The Arnhem water bodies were monitored in September 2005 and May 2006. The bodies in Nijmegen were monitored in the April-May 2005 period (spring) and a second time in the August-September 2005 period (summer). Locations were chosen so as to include variety in morphology, nutrient status and sediment quality (Vermonden et al., 2009a,b). Macrophyte cover and species composition also varied between water bodies (Vermonden et al., 2010). Macrophyte species characteristic of the urban waters included: *Ceratophyllum demersum*, *Lemna minuta*, and *Hydrocharis morsus-ranae*.

The percentage of locations where native and exotic macroinvertebrate crustaceans co-occurred was calculated based on the monitoring in 2005 and 2006 with additional data on 2006 and 2007 (16 of the locations in Nijmegen monitored in 2005, were also monitored in spring and summer 2006 and 2007; the locations in Arnhem, as monitored in summer 2005 and spring 2006, were also monitored in summer 2006 and spring and summer 2007; in 2007 two new locations in Arnhem and two new locations in Nijmegen were additionally monitored in spring and summer). These data were included for analyzing interactions between crustacean species, because some exotic crustacean species were present in very few locations, during 2005 and spring 2006.

Macroinvertebrates

For each water body, aquatic and benthic macroinvertebrates were collected on each sampling date. Aquatic macroinvertebrates were sampled using a 20 by 30 cm pond net with 0.5 mm mesh size (Vermonden et al., 2009a). A sample consisted of two sweeps over a length of approximately 2 m in open water just above the sediment, one sweep starting from the open water towards the bank, and one sweep parallel to the bank. Benthic macroinvertebrates were sampled from the top layer of the sediment using a core sampler (diameter 7 cm, height 9 cm) that was pushed fully into the sediment once at distances of approximately 75, 150, and 225 cm from the bank. In the laboratory, all samples were washed over three sieves with 2, 1 and 0.5 mm mesh size to facilitate sorting. Individuals were identified to the lowest taxonomical level possible (usually species level). Oligochaeta and Diptera were excluded from the analysis.

Environmental variables

The following parameters were measured in the field: electrical conductivity (EC, Hanna Combo meter), stream velocity (SENSA-RC2 water velocity meter), dimensions of the water body, percentage of shade, transparency (Secchi depth) and percentage cover by submerged vegetation (e.g. *Elodea nuttallii*, *Ceratophyllum demersum*), nymphaeid vegetation (e.g. *Nuphar lutea*, *Nymphaea alba*) and lemnid vegetation (*Lemna* sp.). Water bodies that were transparent down to the bottom were assigned a Secchi depth of 1 m, because some water bodies were not deeper than 1 m and Secchi depth could not be measured beyond the bottom. Abundance of water birds was recorded for 30 min. Dominant species were always sedentary (e.g., *Anas platyrhynchos* and *Fulica atra*). Four water bird abundance classes were distinguished: 0 = absent, 1= 1-20 individuals ha^{-1}, 2 = 20-50 individuals ha^{-1}, 3 = > 50 individuals ha^{-1}. Two surface water samples and three sediment samples were taken for further analysis. The pH and alkalinity of surface water samples were measured the following day, after storage overnight at 4°C. The pH measured in the laboratory was very consistent with that measured in the field (Hanna Combo meter). The pH measured in the laboratory was used in the analysis. Water samples were stored at -20°C until further analysis after adding citric acid (125 mg l^{-1}). The following ions were measured colorimetrically (Auto Analyzer 3, Digital colorimeter, Bran + Luebbe, Norderstedt, Germany): NO_3^- according to Kamphake et al. (1967), NH_4^+ according to Grasshoff & Johannsen (1972), Cl^- according to O'Brien (1962) and PO_4^{3-} according to Henriksen (1965). Metals, Ca^{2+} and Mg^{2+} were measured by inductively coupled plasma mass spectrometry (Thermo Electron Corporation, United Kingdom). Sediment samples were dried for 24 h at 100°C and grain size was determined with a Coulter LS 230 laser diffraction device (Beckman Coulter, Inc, Fullerton CA, USA). All physico-chemical factors were measured at least in duplicate and average values were used for the data analysis.

Data analysis

Macroinvertebrates from sediment and water samples were pooled in the analysis. Species were defined as exotic species when they have established themselves after intentional or unintentional introduction by human activities far from their original biogeographic area and classified as such based on the list in Van der Velde et al. (2002). Spearman's Rho correlations were used to determine relationships between environmental variables, native and exotic macroinvertebrate species richness and abundances, and relative abundances of exotic species (abundance of exotic species divided by abundance of all species, multiplied by 100%).

Canoco for Windows Version 4.0 (Ter Braak & Šmilauer, 1998) was used to perform Canonical Correspondence Analysis (CCA) in order to relate presence and abundance of individual macroinvertebrate species to environmental variables. To gain more insight in differences between exotic species and related native species, analysis was limited to taxonomic groups in which exotic species were found: Bivalvia, Gastropoda, Tricladida and Crustacea and the two seasons were pooled. A direct method was used to identify the environmental variables significantly contributing to the variation in macroinvertebrate assemblages (Jongman et al., 1995). A unimodal response model was selected, because there was a wide range in environmental variables. Before analysis, macroinvertebrate abundances were transformed according to Preston (1962): Preston class $= \log_2$ (abundance + 1) and rare species (occurring only once or twice in the data-set) were deleted prior to the analysis. Environmental variables with wide ranges (NH_4^+, PO_4^{3-}, total

P, Li^+, total Fe, Cu^{2+}, Pb^{2+}) were log-transformed; Al^{3+} and stream velocity were transformed to an ordinal scale. An ordinal scale was also used for the abundance of water birds and the cover by submerged, nymphaeid and lemnid vegetation (for definitions of macrophyte growth forms see Den Hartog & Van der Velde, 1988). The percentage of clay and silt fraction (<64 μm) was used as a measure of sediment composition. Significance of environmental variables was tested with CCA, using 500 Monte Carlo permutations under full model conditions (Jongman et al., 1995).

The average abundance of native and exotic species was determined in four different urban water types as distinguished by Vermonden et al. (2009a). The first urban water type was characterized by relatively low nutrient levels, sand sediment, turbid water and poorly developed vegetation (Table 1). The second urban water type also had relatively low nutrient levels, but a clay sediment and vegetation was always present. The third urban water type was characterized by high nutrient levels, high transparency, and a high cover of submerged vegetation. The last urban water type showed the highest nutrient levels, submerged vegetation was almost absent and lemnids or filamentous macroalgae were dominant.

Spearman's Rho correlations were calculated between the abundance of native crustacean species and the maximum abundance of exotic crustacean species. The analysis only included locations, where the native and exotic species co-occurred.

Table 1 Average values of environmental variables of the urban water types (minimum and maximum values in brackets). a and b indicate significant differences between types (bold, ANOVA, post-hoc Gabriel: $p<0.05$).

	Water type 1, turbid (n=8)	Water type 2 nutrient-poor (n=11)	Water type 3 vegetation-rich (n=8)	Water type 4 nutrient-rich (n=9)
NO_3^- (mg l^{-1})	**2.0^a (0.5-6.8)**	**1.3^a (0.0-4.7)**	**5.1^b (0.4-7.8)**	**5.7^b (0.7-10.3)**
NH_4^+ (mg $^{-1}$)	0.18 (0.05-0.81)	0.15 (0.02-0.30)	0.11 (0.03-0.33)	0.12 (0.04-0.16)
PO_4^{3-} (mg l^{-1})	0.17 (0.03-0.93)	0.06 (0.02-0.12)	0.10 (0.02-0.37)	0.12 (0.03-0.19)
Cl^- (mg l^{-1})	**42^{ab} (33-57)**	**40^a (29-55)**	**48^{ab} (42-55)**	**50^b (46-58)**
pH	7.8 (7.5-8.1)	7.7 (7.4-9.2)	7.6 (7.1-8.6)	7.4 (7.1-7.9)
Alkalinity (meq l^{-1})	3.0 (2.1-4.8)	3.2 (0.9-5.1)	2.5 (1.9-4.2)	2.5 (2.0-4.3)
EC (μS m^{-1})	520 (393-647)	533 (288-667)	520 (477-632)	531 (482-656)
Ca^{2+} (mg l^{-1})	67 (49-87)	63 (14-88)	57 (47-73)	58 (49-79)
Mg^{2+} (mg l^{-1})	9.3 (7.9-11.2)	9.2 (3.0-13.3)	8.1 (7.2-9.6)	8.7 (7.4-13.0)
Total-Fe (μg l^{-1})	67 (35-119)	168 (50-653)	146 (55-354)	68 (25-188)
Al^{3+} (μg l^{-1})	8.4 (4.6-15.5)	7.6 (4.2-13.3)	4.7 (2.3-7.1)	15.0 (2.3-90.2)
Cu^{2+} (μg l^{-1})	1.5 (0.7-3.3)	1.9 (0.8-6.3)	1.9 (0.7-3.9)	2.3 (0.7-5.1)
Zn^{2+} (μg l^{-1})	26 (19-40)	29 (10-55)	31 (23-48)	38 (26-68)
Pb^{2+} (μg l^{-1})	0.2 (0.1-0.3)	0.4 (0.1-1.1)	0.2 (0.1-0.5)	1.0 (0.1-3.1)
% Grain size < 64 μm	**12^a (2-63)**	**54^b (1-97)**	**21^{ab} (3-69)**	**11^a (1-60)**
Thickness of sludge layer (cm)	**0.7^a (0-3.5)**	**0.6^a (0-3)**	**5.8^{ab} (0-17.5)**	**6.2^b (0-12.5)**
Transparency (Secchi (m))	**0.6^a (0.4-1)**	**0.8^{ab} (0.4-1)**	**0.9^b (0.5-1)**	**0.9^b (0.4-1)**
Stream velocity (cm s^{-1})	4.1 (2.8-10.5)	4.8 (2.8-9.7)	13.8 (3.6-60.4)	3.1 (2.6-4.0)
Width (m)	**22^a (12-30)**	**14^{ab} (5-30)**	**11^b (6-25)**	**12^b (8-20)**
Shade (%)	58 (30-80)	55 (25-80)	44 (0-80)	58 (5-95)
Nymphaeid vegetation[c]	**0.2^{ab} (0-1)**	**0.5^a (0-1)**	**0^b (0-0)**	**0^b (0-0)**
Submerged vegetation[d]	**0.7^a (0-3)**	**1.8^{ab} (0.5-3)**	**2.4^b (0.5-3)**	**0.8^a (0-2)**
Lemnid vegetation[d]	0 (0-0)	0.2 (0-1)	0.6 (0-1.5)	1.1 (0-3)
Water birds[e]	1.9 (1-3)	1.5 (0-3)	2.2 (1-3)	2.4 (1-3)

[c] 0=absent, 1=present, [d] 0=absent, 1=<10%, 2=10-50%, 3=>50% cover, [e] 0=0 ha^{-1}, 1=1-20 ha^{-1}, 2=20-50 ha^{-1}, 3=>50 ha^{-1}

Results

In the urban water systems of Arnhem and Nijmegen, a total of nine exotic and 30 native macroinvertebrate taxa were found within the taxonomic groups Bivalvia, Gastropoda, Tricladida and Crustacea (Table 2). Within the taxon Tricladida one out of seven species

Table 2 Percentage of locations (frequency of occurrence) where native and exotic macroinvertebrate species were present in urban water systems (overview limited to taxonomic groups with exotic species; exotic species in bold). Data for the two seasons were pooled prior to analysis.

Macroinvertebrate species	Presence (%) all water types (n=36)	Presence (%) water type 1, turbid (n=8)	Presence (%) water type 2, nutrient-poor (n=11)	Presence (%) water type 3, vegetation-rich (n=8)	Presence (%) water type 4, nutrient-rich (n=9)
Tricladida					
Dugesia tigrina	27.8	25	45.5	25	11.1
Bothromesostoma personatum	2.8	0	9.1	0	0
Dendrocoelum lacteum	19.4	0	9.1	0	0
Dugesia polychroa	55.6	0	27.3	37.5	11.1
Mesostoma sp.	2.8	12.5	63.6	87.5	55.6
Planaria torva	2.8	0	9.1	0	0
Polycelis sp.	36.1	0	27.3	87.5	33.3
Crustacea					
Asellus aquaticus	88.9	50	100	100	100
Crangonyx pseudogracilis	36.1	0	9.1	62.5	77.8
Gammarus pulex	66.7	50	54.5	100	66.7
Gammarus tigrinus	30.6	75	27.3	12.5	11.1
Limnomysis benedeni	30.6	62.5	27.3	0	33.3
Proasellus coxalis	36.1	0	36.4	25	77.8
Proasellus meridianus	50	25	27.3	75	77.8
Mollusca, Bivalvia					
Dreissena polymorpha	2.8	12.5	0	0	0
Musculium lacustre	19.4	25	45.5	0	0
Pisidium sp.	25	25	45.5	25	0
Sphaerium corneum	25	0	36.4	37.5	22.2
Mollusca, Gastropoda					
Acroloxus lacustris	16.7	12.5	45.5	0	0
Anisus vortex	47.2	0	45.5	100	44.4
Bathyomphalus contortus	19.4	50	9.1	12.5	11.1
Bithynia leachii	27.8	12.5	72.7	12.5	0
Bithynia tentaculata	80.6	25	100	100	88.9
*Ferrissia wautieri**	11.1	12.5	27.3	0	0
Gyraulus albus	66.7	25	100	100	33.3
Gyraulus crista	22.2	0	27.3	62.5	0
Hippeutis complanatus	33.3	12.5	72.7	12.5	22.2
Lymnaea stagnalis	13.9	12.5	27.3	12.5	0
Physa fontinalis	55.6	12.5	81.8	100	22.2
Physella acuta	5.6	0	9.1	0	11.1
Planorbarius corneus	2.8	0	0	12.5	0
Planorbis carinatus	36.1	0	45.5	87.5	11.1
Potamopyrgus antipodarum	27.8	12.5	27.3	25	44.4
Radix auricularia	19.4	12.5	45.5	0	11.1
Radix balthica	36.1	0	27.3	100	22.2
Stagnicola palustris s.l.	8.3	0	18.2	12.5	0
Valvata cristata	13.9	12.5	36.4	0	0
Valvata piscinalis	55.6	12.5	63.6	87.5	55.6
Viviparus contectus	2.8	0	9.1	0	0

* taxonomic status uncertain, possibly *F. gracilis* (see text)

was non-native. Five out of seven species of Crustacea (Peracarida) were non-native. One non-native species was found, out of four species of Bivalvia. Within Gastropoda two out of 21 species was non-native. Exotic species (origin between brackets) were *Dugesia tigrina* (North America), *Crangonyx pseudogracilis* (North America), *Gammarus tigrinus* (North America), *Limnomysis benedeni* (Ponto-Caspian), *Proasellus coxalis* (Eastern Europe), *Proasellus meridianus* (Southern Europe), *Dreissena polymorpha* (Ponto-Caspian), *Physella acuta* (North America), and *Potamopyrgus antipodarum* (New Zealand).

During the spring, native species richness and abundance were negatively related to surface water NO_3^- and PO_4^{3-} concentrations and the thickness of the sludge layer, and positively related to NH_4^+ levels (Table 3). Exotic species richness and abundance were positively related to thickness of the sludge layer (mainly leaf litter). Relative abundance of exotic species was positively related to nitrate concentrations, cover by lemnids and thickness of the sludge layer. During the summer, native species richness and abundance were positively related to the submerged vegetation cover. Exotic species richness and relative abundance of exotic species were positively related to nitrate levels. Relative abundance of exotic species was negatively related to cover by submerged vegetation. The relative abundance of exotic species was negatively related to native species richness and abundance during the summer.

Table 3 Spearman's Rho correlations between environmental variables, native and exotic macroinvertebrate species richness and abundance and relative abundance of exotic species (abundance of exotic species divided by abundance of all species) in urban waters. All taxonomic groups (except for Oligochaeta and Diptera) were included in the analysis.

	Spring					Summer				
	Native species richness	Exotic species richness	Abundance native species	Abundance exotic species	Relative abundance exotic species	Native species richness	Exotic species richness	Abundance native species	Abundance exotic species	Relative abundance exotic species
Native species richness	-	–0.01	0.74***	–0.11	–0.25	-	0.16	0.79***	0.17	–0.44**
Exotic species richness		-	0.16	0.88***	0.84***		-	0.25	0.64***	0.40**
Abundance native species			-	0.17	–0.12			-	0.32*	–0.39**
Abundance exotic species				-	0.91***				-	0.62***
NO_3^- (mg l^{-1})	–0.57***	0.27	–0.37*	0.26	0.30*	0.03	0.29*	0.00	0.14	0.29*
NH_4^+ (mg l^{-1})	0.54***	0.09	0.58***	–0.02	–0.16	–0.09	–0.15	–0.10	–0.11	–0.03
PO_4^{3-} (mg l^{-1})	–0.36*	0.14	–0.37*	0.01	0.10	0.02	0.19	0.03	0.14	0.18
Submerged vegetation[a]	0.17	–0.07	0.15	–0.02	–0.04	0.66***	0.03	0.58***	–0.02	–0.42**
Nymphaeid vegetation[b]	0.38*	–0.10	0.24	–0.16	–0.16	0.19	–0.13	0.09	0.10	–0.04
Lemnid vegetation[a]	–0.13	0.34*	–0.02	0.26	0.29*	0.12	–0.18	0.08	–0.04	0.05
Thickness of sludge layer	–0.45**	0.30*	–0.39**	0.30*	0.40**	0.06	0.17	0.05	0.08	0.10

[a] 0=absent, 1=<10%, 2=10-50%, 3=>50% cover, [b] 0=absent, 1=present.

*: p<0.05, **: p<0.01, ***: p<0.001

Macroinvertebrate assemblages in urban water systems were significantly related to nitrate concentration in the surface water, transparency, submerged vegetation, sediment composition (clay or sand), nymphaeid vegetation and pH (Figure 1). Exotic species were similarly related to environmental variables as native species. Only *G. tigrinus* and *L. benedeni* were plotted higher on the second axis of the CCA biplot than the other

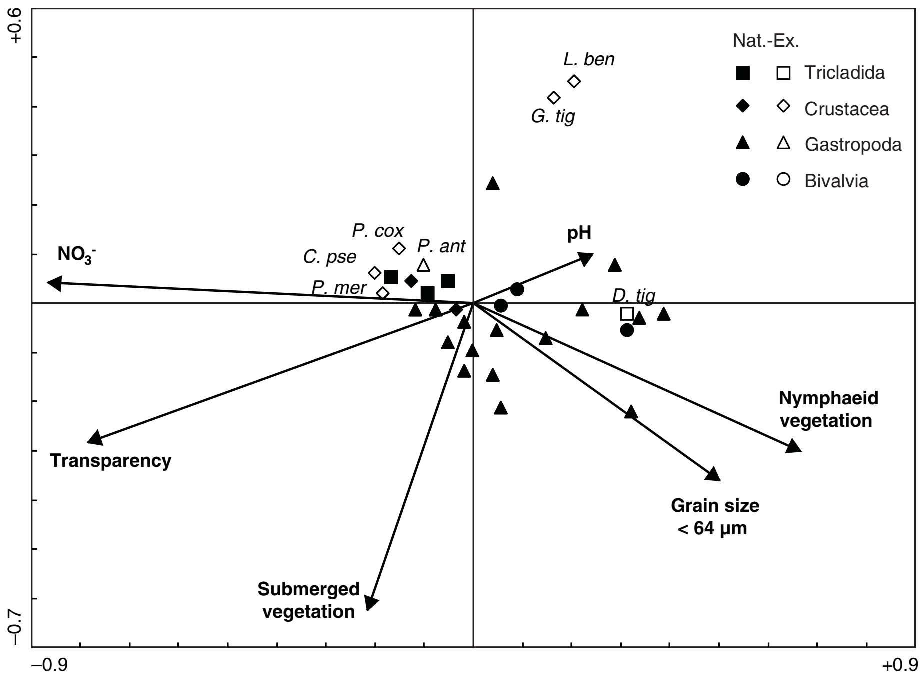

Figure 1 Canonical Correspondence analysis biplot of native and exotic macroinvertebrates and environmental variables significantly explaining variation in species composition. Rare species (occurring only once or twice in the data-set) were deleted prior to the analysis. *C. pse* = *Crangonyx pseudogracilis*, *D. tig* = *Dugesia tigrina*, *G. tig* = *Gammarus tigrinus*, *L. ben* = *Limnomysis benedeni*, *P. ant* = *Potamopyrgus antipodarum*, *P. cox* = *Proasellus coxalis*, *P. mer* = *Proasellus meridianus*.

species, meaning that these species were present in waters with very low coverage of submerged vegetation, which was negatively related to the second axis of the CCA biplot.

The average abundance of the triclad *D. tigrina* was highest in the nutrient-poor water bodies (Type 2, Figure 2). Crustacean abundance (both native and exotic) was highest in the nutrient-rich water systems of type 3 and 4. Asellidae was the dominating family within the Crustacea. *P. coxalis* was the most abundant exotic species of the Asellidae during spring, while *P. meridianus* was the most abundant one during summer. Within the Amphipoda, *C. pseudogracilis* was more abundant than *G. tigrinus* in nutrient-rich waters (Type 3 and 4) during both seasons. *G. tigrinus* was more abundant than *C. pseudogracilis* in turbid waters (Type 1). The mysid *L. benedeni* was mostly found within vegetation-poor water bodies (Type 1 and 4). Exotic Mollusca species were rare or absent in most locations. The bivalve *D. polymorpha* was only present in the turbid waters (Table 2, Figure 2). The abundances of the gastropod species *P. acuta* and *P. antipodarum* were very low in comparison to the abundance of native gastropods. *P. acuta* was found in the nutrient-poor as well as nutrient-rich waters (Type 2 and 4), while *P. antipodarum* was found in all urban water types.

The exotic species *C. pseudogracilis*, *P. coxalis* and *P. meridianus* co-existed with the native *Asellus aquaticus* in more than 90% of the locations where the exotic species occurred (Table 4). *G. tigrinus* and *L. benedeni* co-existed with *A. aquaticus* in 39 and 53% of the locations, respectively. Co-existence of exotic crustaceans with the native *G. pulex*

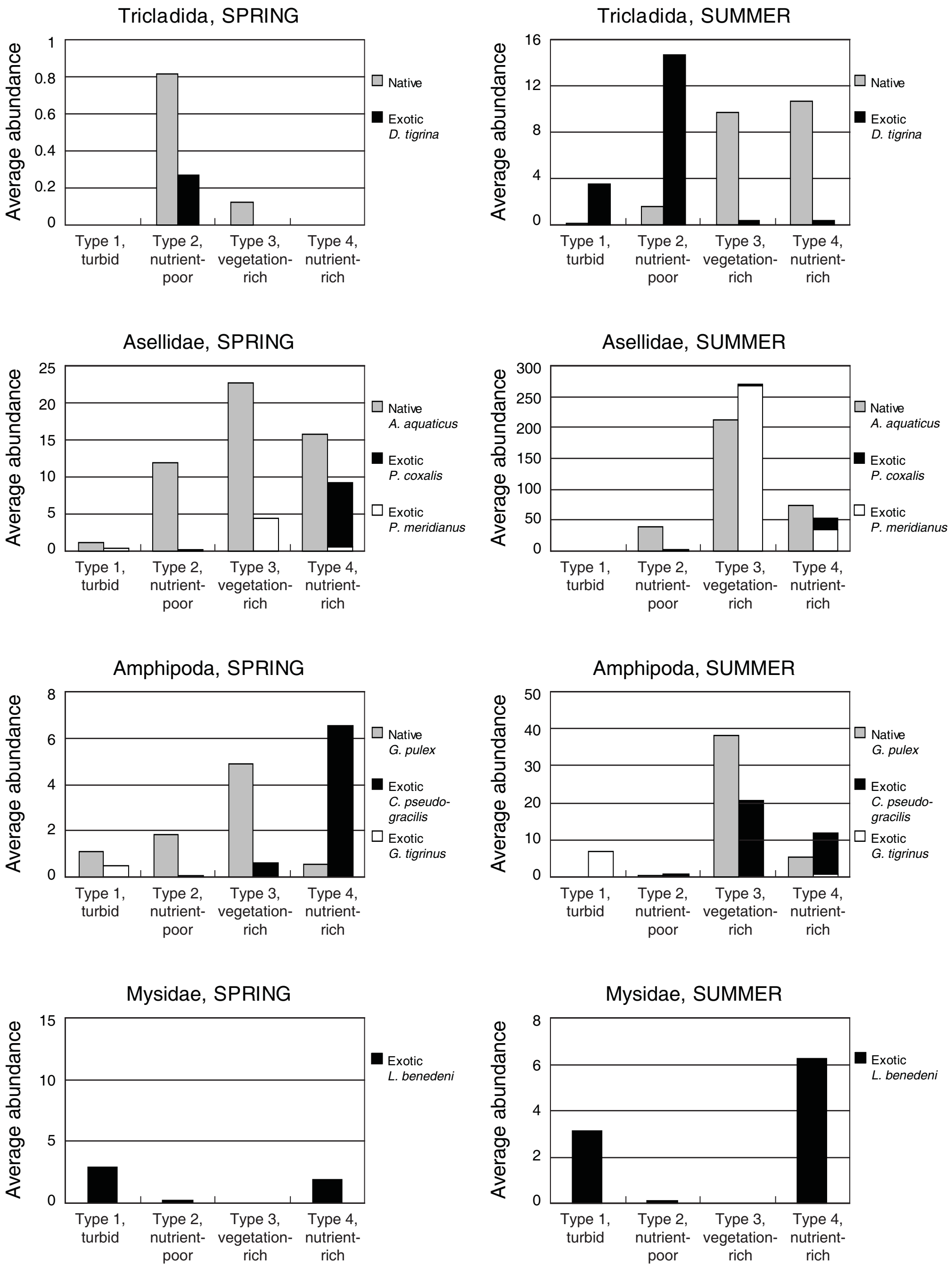

Figure 2 Average abundance of native and exotic macroinvertebrate taxa per water type, in spring and summer. Abundances of native species within taxa Tricladida, Bivalvia and Gastropoda are summed. Abundances of exotic species are presented in stacked columns. Detailed figures of exotic Gastropoda abundances are shown as well.

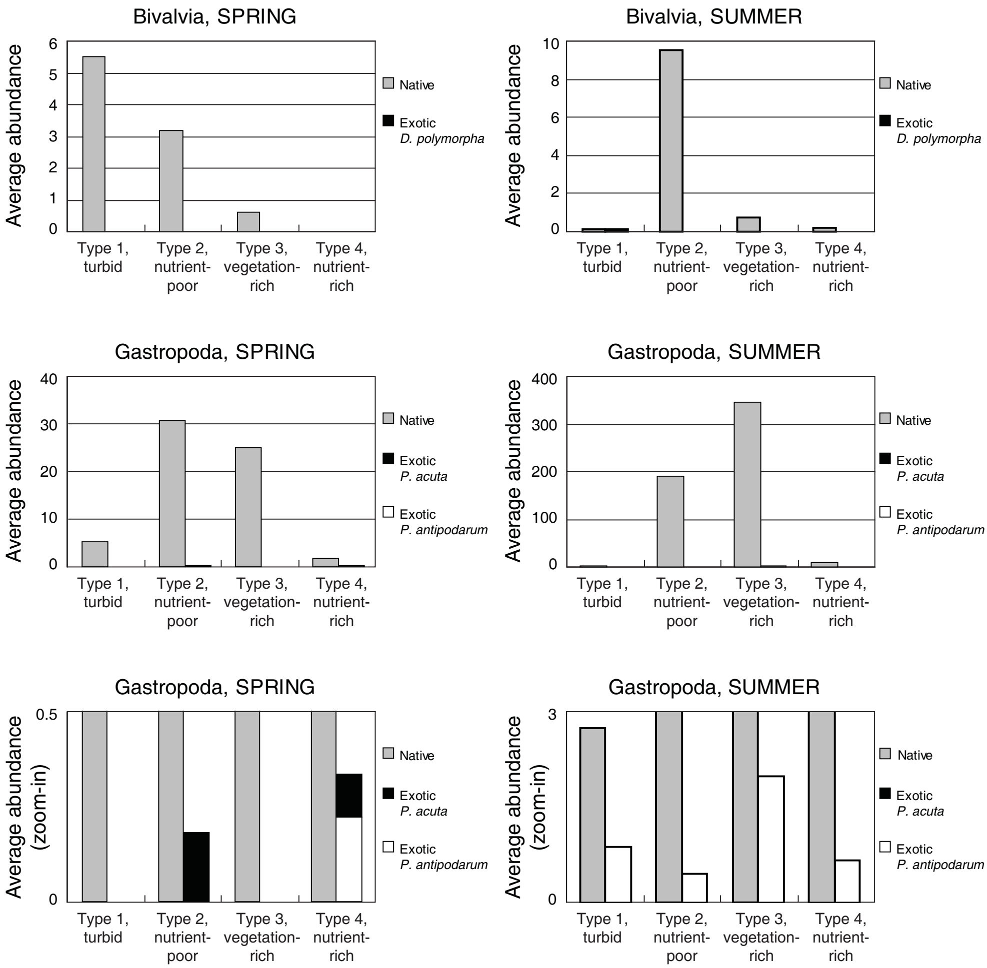

Figure 2 *Continued*

Table 4 Percentage of samples where exotic species co-occur with native species in urban waters. Percentage was calculated as number of samples where both species occur, divided by number of samples where the exotic species occurs, multiplied by 100%. In brackets the number of samples are given, where exotic species occur exclusively, where both species coexist (in bold) and where the native species occur exclusively, respectively.

	Native species			
Exotic species	*Asellus aquaticus*		*Gammarus pulex*	
Crangonyx pseudogracilis	91.3%	(4, **42**, 96)	67.4%	(15, **31**, 44)
Gammarus tigrinus	39.1%	(14, **9**, 129)	21.7%	(18, **5**, 70)
Limnomysis benedeni	53.3%	(21, **24**, 114)	28.9%	(32, **13**, 62)
Proasellus coxalis	91.2%	(3, **31**, 107)	52.9%	(16, **18**, 57)
Proasellus meridianus	96.2%	(2, **50**, 88)	75.0%	(13, **39**, 36)

gave similar figures, but percentages were lower. The maximum abundances of *C. pseudogracilis*, *P. coxalis* and *P. meridianus* were positively related to the abundance of the native *A. aquaticus* (Table 5). The maximum abundance of *C. pseudogracilis* was also positively related to the abundance of the native *G. pulex*.

Table 5 Spearman's Rho correlation coefficient on maximum abundance values of exotic species in urban waters.

	Native species	
Exotic species	*Asellus aquaticus*	*Gammarus pulex*
Crangonyx pseudogracilis	0.42*	0.41*
Gammarus tigrinus	NS	NS
Limnomysis benedeni	NS	NS
Proasellus coxalis	0.56**	NS
Proasellus meridianus	0.67***	NS

*: $p<0.05$, **: $p<0.01$, ***: $p<0.001$

Discussion

The exotic species found in this study were well established species in the River Rhine and its delta (Bij de Vaate et al., 2002, Van der Velde et al., 2002, Leuven et al., 2009). *D. tigrina* is an asexually reproducing predatory flatworm which can co-exist with other triclads in larger waters by opportunistic feeding on dominant prey (Van der Velde, 1975). *D. tigrina* is present in all different urban water types. *D. polymorpha* is a sessile bivalve filter feeder occurring only in the turbid urban waters. *P. antipodarum* is a parthenogenetic reproducing snail, deposit feeding on detritus and diatoms and was found in all the water types investigated. *P. acuta* is an omnivorous snail, feeding on dead animals, algae, macrophytes, fungi, and detritus (Gittenberger et al., 1998 and literature therein). In the urban waters examined, *P. acuta* occurred in the nutrient-poor as well as in the nutrient-rich water types.

Ferrissia wautieri (alias *F. clessiniana*) is a limpet species sometimes considered an exotic species from North America, in which case it should be *F. gracilis*, which is also found in Europe (Walther et al., 2006). However, in The Netherlands, *F. wautieri* was found in Holocene and Pleistocene deposits and in France even in Pliocene deposits (Wautier, 1975, Meijer, 1987, Gittenberger et al., 1998), which indicates that *F. wautieri* is a native European species. As the taxonomy of this species is in question, we do not consider it in further detail here.

Crustaceans were the most abundant exotic species. Karatayev et al. (2009) found that invaders were overrepresented by crustraceans and molluscs in North America and Europe. In our study, crustaceans were especially abundant in nutrient-rich urban water bodies, with the exception of the exotic species *L. benedeni* and *G. tigrinus*. Increasing nutrient levels usually result in a decrease of macroinvertebrate richness, but an increase of macroinvertebrate species associated with decaying leaves and other detritus (Bergfur et al., 2007). At higher nutrient levels, the breakdown of leaves by microbial processing is enhanced (Pascoal & Cássio, 2004) and more food becomes available for the detritivores. Many crustaceans are omnivores (amphipods) or detritivores-herbivores (asellids) (Monakov, 2003), which may explain why both native and exotic crustaceans were most abundant in the nutrient-rich waters. The positive correlations between several crustacean species indicates that all crustacean species can profit from increased food availability and there might be little competition between these crustacean species. When provided with abundant food, *A. aquaticus* and *P. coxalis* were able to co-exist in a series of experiments (see Monakov, 2003 and literature therein). Karatayev et al. (2009) argued that species with broad, generalized diets, such as crustaceans, are more likely to be successful in new environments and may even be facilitated by moderate levels of eutrophication.

L. benedeni was in most cases more abundant than other exotic crustaceans in waters where native *A. aquaticus* and *G. pulex* were not recorded (Table 4). *L. benedeni* is usually associated with lentic water bodies with aquatic vegetation, or other types of hard substrate such as roots, stones or zebra mussels (Wittmann, 1995, Kelleher et al., 1999, Gergs et al., 2008). *L. benedeni* has a non-specific food preference and feeds mainly on small particles, such as detritus, epilithon and phytoplankton (Bij de Vaate et al., 2002, Gergs et al., 2008). Assman et al. (2009) showed that leaf litter can be an important part of the diet of *L. benedeni*. Moreover, he suggested that *L. benedeni* could substitute a missing shredder. This could be the case here, because *L. benedeni* was often present when other crustaceans were absent. Moreover mysids protect themselves against predation by forming swarms (Lindén, 2007) with turbidity as additional shelter factor.

G. tigrinus usually occurred where *G. pulex* was absent. The native *G. pulex* usually dominates over the exotic *G. tigrinus* at conductivities between 390-430 $\mu S\ cm^{-1}$, while *G. tigrinus* is favoured at conductivities between 1500-1600 $\mu S\ cm^{-1}$ (Dick & Platvoet, 1996). In our study, conductivity was relatively low (average 526 $\mu S\ cm^{-1}$) and *G. pulex* was the dominant species in nutrient-poor, vegetation-rich and nutrient-rich urban waters. Orav-Kotta et al. (2009) showed that competition between two gammarid species (*G. tigrinus* and *G. salinus*) depends on habitat type. *G. tigrinus* is able to shelter in soft sediment without vegetation (Platvoet et al., 2009). *G. pulex* might be more dependent on vegetation as habitat and therefore less abundant in turbid waters with poorly developed vegetation. No direct correlation could be demonstrated between *G. tigrinus* and *G. pulex* abundances due to the low number of locations (n=5), where these species coexisted.

In the urban waters during spring, native species richness and abundance were negatively related to several parameters indicating eutrophication (nitrate, phosphate, sludge layer). On the contrary, most exotic species in urban waters were positively related to eutrophication indicators (sludge layer and lemnid vegetation).

Native species richness and abundance increased with the abundance of submerged vegetation during summer. During summer, submerged vegetation developed and native vegetation-dwelling grazing species, such as gastropods, became much more abundant. Vegetation is important for macroinvertebrates, as a habitat, food source and shelter to avoid predation (Crowder & Cooper, 1982, Dvořak & Best, 1982, Newman, 1991). Habitat heterogeneity promotes the coexistence of native and exotic species (Palmer & Ricciardi, 2004, Kestrup & Ricciardi, 2009). High environmental heterogeneity in the urban water locations with submerged and nymphaeid vegetation may have led to high native species richness and abundance. Although absolute abundance of exotic species also increased during summer, the relative abundance decreased, due to much higher abundances of native species richness. Native and exotic species co-existed and did not out-compete each other (Table 4, 5); both profited from habitat and food availability. Native species depended more on environmental heterogeneity in the form of submerged and nymphaeid vegetation, while exotic, mostly detritivorous-herbivorous and omnivorous species depended on leaf breakdown, which was higher at increased nutrient levels.

Invasibility of urban water systems by exotic species could be decreased by supporting development of submerged and nymphaeid vegetation in order to increase environmental heterogeneity supporting native species. Development of these types of vegetation can be stimulated by lowering of nutrient levels, optimizing the mowing regime and the development of natural banks (Vermonden et al., 2009a). With increasing coverage of vegetation, nutrient levels also decrease. Nutrient levels can additionally be lowered by regular dredging, avoiding the inlet of nutrient-rich water and decreasing the (excessive) feeding of water birds and fish. However, dredging can also act as a disturbance stimulating invasion by exotic macrophytes (Van der Velde et al., 2002).

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Dredging in urban water systems of Nijmegen. Photo: Kim Vermonden

Chapter 7

Synthesis

Kim Vermonden

General discussion

Historically, conservation biologists focused on natural areas (Miller & Hobbs, 2002). It is usually assumed that increasing human density threatens biodiversity (Luck, 2007). Nevertheless, biodiversity hotspots rich in endemic species are often found at high human population density (Cincotta et al., 2000). Miller & Hobbs (2002) therefore advocated that biologists should focus more on areas where people live and work, to conserve and to restore native habitats in densely settled areas.

Urban surface waters are considered attractive for citizens and are therefore given a more prominent place in new suburbs and urban renewal. It can be considered a novel opportunity to design them as ecological solutions to provide vital services such as biodiversity (Palmer et al., 2004, Wang et al., 2006). Restoration of ecosystems, conservation and enhancement of biodiversity in urban areas are therefore becoming more and more important (Savard et al., 2000). Unfortunately, ecological rehabilitation of urban water systems is not always successful (Grayson et al., 1999, Larson et al., 2001, Booth, 2005, Suren & McMurthrie, 2005). Paul and Meyer (2001) stressed the importance of gaining more knowledge on the ecology of urban streams and the challenge to integrate knowledge on physical, chemical and biological processes in impaired systems. This thesis aims at determining the potential of urban surface water systems for biodiversity and at understanding key factors to optimize their design and management for biological conservation.

The present research shows that urban surface waters in areas with a relatively high acreage of impervious area but a separate sewage system can still obtain relatively good water quality and high biodiversity values (Chapter 2-4). For macrophyte species, rehabilitation success of these waters is mainly related to trophic status and appeared not to be limited by the regional species pool (Chapter 3). Lowering nutrient levels and stimulating vegetation structure in urban waters are important keys for rehabilitation of native biodiversity in these systems and decreasing invasibility by exotic species (Chapter 4-7). Dredging of polluted sediment and organic debris is effectively improving oxygen conditions on the short term and resulted in higher chironomid diversity (Chapter 6). Nevertheless, other rehabilitation measures need to be included to reduce nutrient input from other sources (e.g. upward seepage of polluted groundwater, feeding of ducks and fishes) and to stimulate vegetation development and hereby increase biodiversity. In this synthesis we discuss the research methods, biological assessment of urban waters and the consequences for urban water management.

Research methods

Multivariate analyses are useful to unravel how a multitude of species responds to environmental factors (Ter Braak & Verdonschot, 1995). In this study, multivariate analysis has been successfully applied to correlate water quality and the main factors influencing water quality, environmental variables and macrophytes, (native and exotic) macroinvertebrates and chironomids in particular (Chapters 2-6). Water types and species associations are distinguished and related to environmental variables (Chapters 4-6). The correlations between biodiversity indices and environmental variables facilitate the identification of parameters that influence biodiversity in urban waters, either positively or negatively.

Correlations give a clear indication of the key factors for biodiversity in urban waters. However, causality between species and their environment cannot be shown with these types of correlations. Instead, life-history strategy analysis of species assemblages can contribute to understanding the causal mechanisms between species and their environment (Verberk et al., 2008). In this study, life-history strategy analysis has been applied in a case study on chironomids. Chironomid species with life-history strategies indicative for improved oxygen conditions became more abundant after dredging. Dredging had direct positive effects on chironomid diversity and the abundance of sensitive chironomid species. Our study indeed showed that life-history strategies can contribute to understanding the causal relationships between species and their environment. Nevertheless, to gain complete insight in the causal relationships between biodiversity and environmental characteristics of urban water systems, the life-history approach also has to be applied to other taxonomic groups.

Biological assessment of urban waters

Water quality in urban waters: storm water runoff from impervious area versus local factors

Impervious area is often used as a proxy for urban land use and even as a globally uniform measure of human impact (e.g. McKinney, 2002, Meyer et al., 2005, Alberti et al., 2007, Rhandir & Ekness, 2009, Sutton et al., 2009). Scheuler (1994) proposed the use of impervious area as an indicator of the impact of land development on aquatic systems. Walsh et al. (2005) advocated that urban stream restoration should focus on reducing impervious areas by redesigning storm water systems at the catchment level.

Increase of impervious area typically leads to higher runoff and peak flows, bank erosion in drainage ditches and streams and flush of accumulated pollutants and subsequently a decrease in biodiversity (Scheuler, 1994). Generally, watersheds with less than 5-10% impervious area are predicted to be sensitive streams with full ecological functioning and good to excellent aquatic diversity (Scheuler, 2009). Increasing impervious area of more than 10% cover leads to degradation of stream quality. Watersheds with more than 25% impervious area are expected to be highly degraded with poor water quality and low biodiversity (Scheuler, 2009).

In our study sites the impervious area was approximately 30% and water systems were therefore expected to be severely degraded. In contrast to these expectations, concentrations of most chemical substances in urban surface waters were complying well with water quality standards (Vermonden et al., 2008). While a rain storm had a clear temporary impact in our study area, average water quality was not directly related to acreage of impervious area (Chapter 2). In our urban surface waters, groundwater seepage from polluted river water was a much more important determinant of water quality parameters such as nitrate, potassium, sodium, chloride, alkalinity, calcium, magnesium and iron.

Impervious cover is clearly a negative factor for a lot of urban water systems (e.g. Scheuler, 2009), but local circumstances should not be overlooked. Tavernia & Reed (2009) stress that no single set of urbanization metrics is universally applicable. Our study also showed that a watershed with approximately 30% impervious area can still obtain relatively good water quality. A possible explanation may be that in our study area sewer and storm water systems were separated, preventing overflow of sewers during extreme rain periods.

Contribution of urban waters to biological diversity

Some recent studies have shown that urban areas can be an important habitat for native species (Croci et al., 2008, Collier et al., 2009, Pryke & Samways, 2009, Stewart et al., 2009). Nevertheless, most studies showed considerable degradation of native flora and fauna in urban areas (e.g. Lenat & Crawford, 1994, Paul & Meyer, 2001, Moore & Palmer, 2005, Walsh et al., 2005, McKinney, 2006, Smith & Lamp, 2008).

In our study, values of several biodiversity indices for macroinvertebrates and macrophytes were similar or higher in urban than in rural waters (e.g., macroinvertebrate, chironomid and macrophyte species diversity, macroinvertebrate and chironomid Shannon-index, number of macroinvertebrate red list species, macroinvertebrate and chironomid rareness, Chapter 3-5). The urban water systems in our study area could be less degraded than elsewhere or our reference waters in rural areas could be much more deteriorated compared to other studies. The ecological status of many surface waters in rural areas of the Netherlands is poor, due to relatively high input of nitrogen and phosphorus (Gulati & Van Donk, 2002, Oenema et al., 2005). The high input of these nutrients can be related to surpluses in agriculture, wastewater discharge, nutrient imports via rivers, seepage of nutrient-rich groundwater and resuspension from sediments. This could indicate that rural waters used in our study might also be severely degraded and therefore scored relatively poor in comparison to the urban waters.

Macrophyte species composition and diversity in urban waters were significantly different from those in semi-natural water systems (Chapter 3). Helophyte species richness was significantly higher in semi-natural reference systems than in urban systems. The semi-natural water systems had significantly lower nutrient levels due to the input of local, relatively oligotrophic water.

Urban waters in our study area contributed significantly to macroinvertebrate species diversity in the Netherlands (Chapter 4). Approximately 50% of the triclad and gastropod species in the Netherlands were present in the urban waters. Furthermore two red list species (*Planaria torva* and *Leptocerus tineiformis*) were found. Spikmans (2006) also found two red listed vertebrates species in our study area, i.e. the amphibian *Rana lessonae* and the fish *Leucaspius delineatus*. We therefore conclude that the urban water systems in our study area are an important habitat for fauna species present in the Netherlands.

Biodiversity in a disturbance gradient

The results of the present study could be explained by the intermediate disturbance hypothesis, which suggests that maximum species richness is reached at moderate frequencies or intensities of disturbance (Hobbs & Heunneke, 1992) (Figure 1). The water systems in our study area were all under the influence of antropogenic disturbances, such as mowing, dredging and fluctuating nutrient levels. In the Netherlands nutrient levels are high and anthropogenic disturbance is inevitable. Therefore water systems with little or no disturbance cannot be expected at all in the Netherlands. Semi-natural water systems in our study were least disturbed, resulting in the highest diversity (Figure 1). Urban water systems were more disturbed than semi-natural water systems, resulting in lower diversity. Water systems in agricultural areas had lowest diversity and this could be related to higher disturbance levels due to high nutrient input from agriculture and possibly leaking of sewage water. Very high fluctuations in nutrient levels, due to fertilization of the land during the growing season could also degrade diversity. Nevertheless, urban waters could also have lower biodiversity than rural waters, for example, when urban water systems are

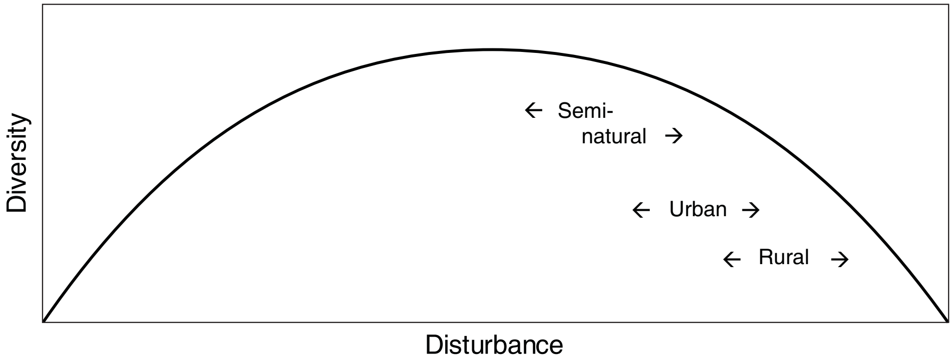

Figure 1 Possible theoretical relationship between values of diversity indices and antropogenic disturbance.

more disturbed or due to the discharge of waste water. Biodiversity in rural waters could also be higher than in urban waters when there is less disturbance in rural areas due to for example the development of buffer zones between agricultural land and water systems.

Diversity decreases when disturbance increases from semi-natural to urban and to rural areas. Species which are characteristic of late successional stages could be absent in the urban and rural areas, because disturbance frequency or intensity might be so high that systems can not develop beyond the pioneer stage (e.g. as a result of mowing and dredging). High nutrient levels stimulate the growth of algae and floating plants and inhibit the growth of submerged vegetation (Hough et al., 1989). This may decrease macrophyte diversity and reduce habitat complexity for fauna. Especially, macroinvertebrates depend on vegetation structure as a habitat, food source and shelter from predation (Crowder and Cooper, 1982, Dvořak and Best, 1982, Newman, 1991).

The results of our study confirmed the intermediate disturbance theory. Semi-natural waters with little disturbance had the highest diversity, followed by urban and rural water systems with lower diversity at higher disturbance levels.

Key environmental factors for biodiversity in urban water systems

Water quality is very important for macrophyte species composition and diversity, while species composition and diversity of macroinvertebrates are mainly related to differences in water quality, abundance of macrophytes, transparency and sediment composition (i.e. sandy vs. clay sediment, Chapter 3-6). Macrophyte species typical for eutrophic conditions were much more abundant in urban waters, than in semi-natural water systems. At increasing nutrient levels in small, shallow surface water submerged macrophytes rooting in the substrate disappear and non-rooting macrophytes such as lemnids become dominant (Roelofs, 1983). In larger and deeper waters epiphytic and planktonic algae become dominant and waters become turbid. Turbidity inhibits the growth of submerged vegetation (Scheffer et al., 1993). We found turbid water bodies with epiphytic and planktonic algae at low nutrient levels during spring and autumn. Probably, nutrient load was very high, but reduced by high nutrient uptake due to algal blooms. Furthermore at high nutrient levels part of the locations still sustained rooting submerged macrophytes. Only the locations with very high nutrient levels combined with lack of water flow showed a permanent cover with lemnids.

Nutrient levels, abundance of submerged, nymphaeid and lemnid vegetation, transparency and sediment composition played a key role for macroinvertebrate diversity and taxa richness in urban water systems (Chapter 4-6). Sediment composition probably did not influence most of the macroinvertebrates directly, because most species live in the water layer. Chironomid species can be bottom-dwelling and therefore be directly influenced by sediment characteristics, as was also found by Principe et al. (2008) and Syrovátka et al. (2009). Sediment composition varied from clay, sand to gravel. Clayey sediment was mostly found in the water bodies of Arnhem, while sandy sediment and gravel were mostly recorded in Nijmegen. Sediment composition influenced water quality in two ways. First of all, small clay particles easily suspended in the water layer when disturbed by rain fall, flow, waves or fish and increased turbidity of the water bodies. Sand and gravel are heavier and therefore less likely to be suspended. Secondly, sediment composition influenced soil permeability and therefore the amount of upward seepage entering the urban waters (Chapter 2).

Nutrient levels, growth of macrophytes and transparency are interrelated and the resulting environmental conditions determine macroinvertebrate species composition and diversity (Figure 2). High macroinvertebrate diversity was found when submerged

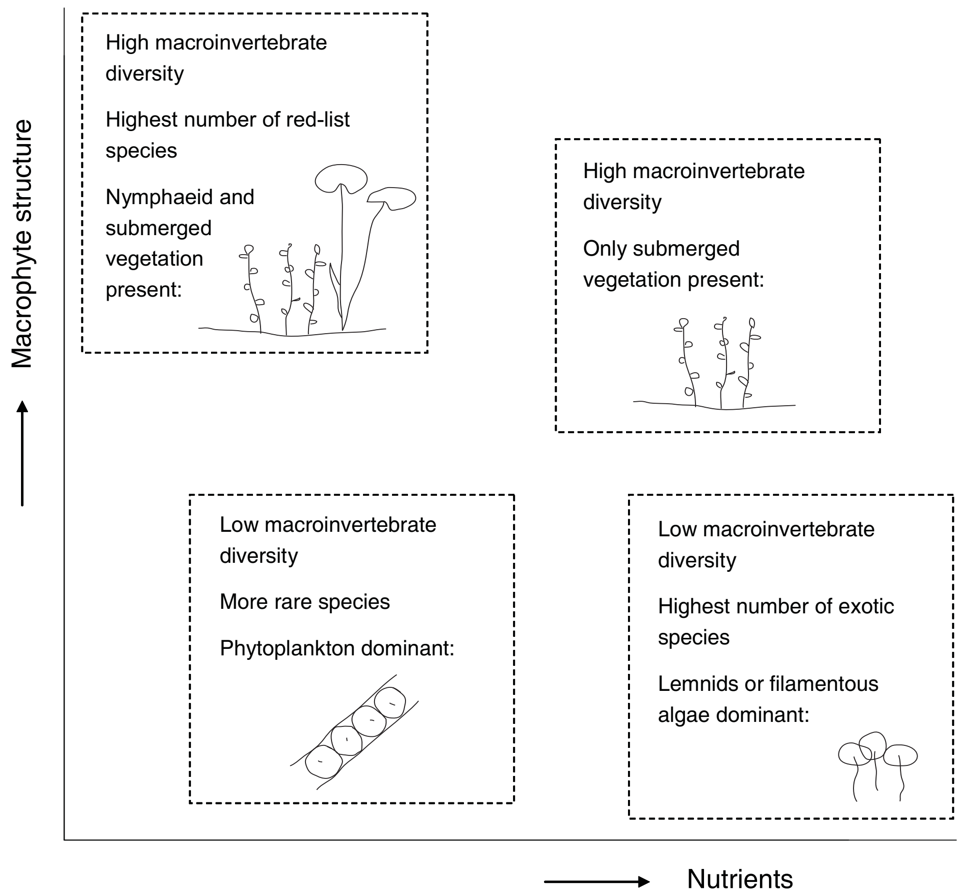

Figure 2 Relationship between macroinvertebrate diversity, cover and type of vegetation and nutrient levels.

vegetation was present and highest when also nymphaeid vegetation was present. Low macroinvertebrate diversity was recorded when submerged vegetation was absent, probably due to shading by floating filamentous algae (FLAB) or lemnids. Rare species were more often found in relatively nutrient poor waters, while exotic species were mostly found in nutrient-rich waters.

Nutrient availability was the main factor influencing macrophyte species composition, diversity and structure, while nutrients and macrophyte structure were key factors for macroinvertebrates in our study. As was discussed above, upward seepage is one of the main factors determining the nutrient levels of urban waters in our study area.

Invasibility of urban water systems by exotic species

Invasions of urban waters by exotic species of various taxonomic groups depend on different pathways for their introduction (Chapter 3, 6). Exotic macrophyte species are intentionally introduced directly as ornamental plants for ponds or aquaria and are often dumped by their owners or may disperse from gardens. Exotic macroinvertebrates are deliberately introduced, accidentally released (e.g. escapes and dumping) or dispersed via man-made waterways, either via natural migration and drift or shipping (Leuven et al., 2009). Both invasive macrophyte species and macroinvertebrates profit from eutrophication of urban water systems.

In order to become a successful invasive species, a species first has to be able to disperse into the new environment and then survive and reproduce under the local (a)biotic pressures (e.g. Van der Velde et al., 2006, Karatayev et al., 2009). Species with broad, generalized diets are more likely to survive in their new environment, while specialized species are more likely to extinct soon after introduction. Many crustaceans are omnivores (amphipods) or detritivores (asellids) and feed on a wide range of food sources. It is therefore not surprising that invasive species were mostly found among the crustaceans. They depend on leaf breakdown or macrophytes with relatively soft tissue, which occurred abundantly at the most eutrophicated urban surface water systems.
Invasive species usually perform better than native species when resource availability is high (Daehler, 2003, Karatayev et al., 2009). Our results confirmed that invasive species, both macrophytes and macroinvertebrates, are facilitated by eutrophication.

Managing key factors for biodiversity of urban waters

Nutrient content played a key role for macrophytes and macroinvertebrates in urban surface waters in our study area, as was stated above. Upward seepage of nutrient-rich groundwater was the most important factor for nutrient input. The influence of upward seepage could be reduced by draining the seepage water and pumping the drainage water directly back into the canals and rivers. The inlet of nutrient-poor regional upward seepage could also be enhanced. Furthermore water systems influenced by groundwater-river-surface water interactions would benefit from improvement of water quality on the regional, national and international level. This requires additional pollution control measures in the upstream parts of the Rhine and Meuse rivers.

A rain storm was also an important input of nutrients and other contaminants such as lead and zinc. Urban lawns and green roofs can effectively help manage stormwater runoff by intercepting and infiltrating runoff from impervious surfaces (VanWoert et al., 2005, Mueller & Thompson, 2009). In general, urban water managers should be critical about the use of catchment imperviousness as steering factor to enhance water quality. Liefting & Langeveld (2008) showed that small-scale filtration systems for storm water runoff contributed very little to the reduction of nutrients in urban surface waters. Nevertheless, a soil bank passage and sand filter did reduce the amounts of copper, lead and zinc in the receiving surface water significantly. The local circumstances and site specific nutrient balance and pollution levels should be assessed, before management decisions can be made.

In Arnhem some additional bottlenecks for surface water quality were found: the effluent of a communal water treatment plant, sewage overflows, inlet of nutrient-rich water, lack of stream flow, the (excessive) feeding of waterbirds and benthivorous fish, and accumulated sediment of organic matter (Vermonden et al., 2008). Management should also aim at reducing the nutrient input of these sources. Dredging can be used to remove the accumulated sediment of organic matter. This reduces the release of nutrients from the sediment and improves oxygen levels. In this way dredging also proved to be effective to increase chironomid species diversity in our study area (Chapter 5).

Maintaining and restoring watershed vegetation corridors in urban landscapes aid efforts for conservation of freshwater biodiversity (Urban et al., 2006). If nutrient levels in urban waters are lowered, it is very likely that lemnids and filamentous algae diminish and submerged and nymphaeid vegetation development is stimulated. Furthermore, vegetation growth can be stimulated by the development of natural banks and optimizing the mowing and dredging regime (e.g. Milson et al. 2003). The cost-efficiency of various rehabilitation measures for reducing nutrient levels and stimulating vegetation development (e.g., reducing upward seepage of polluted groundwater, development of natural banks, and infiltration of storm water runoff) still needs to be assessed.

Conclusions

- In contrast to expectations urban surface water systems with catchments consisting of impervious areas up to approximately 30% still showed moderate to good water quality and high biodiversity. Impervious area was not significantly related to water quality. Regional upward seepage of Rhine-Meuse water played a major role in local water quality during dry and wet weather conditions. A rain storm enhanced the concentration of ammonium, phosphorus, lead and zinc in the urban surface waters.
- The macrophyte species composition and diversity seemed to be influenced by the species pool present within 30 km of the study area. Nevertheless, species composition and diversity was significantly different between the urban and semi-natural area and differences could be related to eutrophication of the urban waters. Macrophyte species composition and diversity in the urban waters were not expected to be limited by the regional species pool.
- Biodiversity values in urban and rural water systems were similar, but biodiversity values in semi-natural reference systems were higher than in urban and rural areas. The urban water systems in our study area contributed significantly to fauna diversity in the Netherlands and included several red list species.
- Nutrient levels determined the macrophyte species composition. Cover by macrophytes and nutrient levels determined the macroinvertebrate species composition and diversity. Eutrophication of urban water systems resulted in lower biodiversity and more invasive species.

- Analysis of life-history strategies helped understanding the ecological processes important for chironomid species, such as the role of accumulated sediment of organic matter. Chironomid species indicative for improved oxygen conditions became more abundant after dredging of these sediments. The full potential of the life-history strategy analysis can only be assessed when other taxonomic groups are also included.
- Native macroinvertebrate biodiversity was mainly determined by environmental heterogeneity in the form of submerged and nymphaeid vegetation, while invasive, mostly detritivorous and omnivorous species were most abundant in nutrient-rich waters.
- Biodiversity in urban waters can be enhanced by lowering nutrient levels and stimulating vegetation development.

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Urban water system Nijmegen. Photo: Kim Vermonden

Summary

In our rapidly urbanizing world, water systems play an important role, providing vital services such as drinking water, irrigation, flood control, transportation, recreation and wildlife habitat (Chapter 1). Knowledge of the structure and function of urban water systems as habitat for flora and fauna species is needed to determine key factors for aquatic biodiversity in urban areas, necessary to optimize their design and management for biological conservation. The main objective of this study was to determine the key environmental factors for biodiversity in urban surface water systems. To achieve this objective we investigated how local and regional processes influenced macroinvertebrate and macrophyte species composition and species richness. For the scientific sound assessment of biodiversity in urban waters we used contemporary knowledge on ecology, such as the species-pool hypothesis, intermediate disturbance hypothesis, life-history strategies approach and invasion biology hypotheses. The urban water systems in our study were situated in the former floodplains of the rivers Meuse and Waal in the eastern part of the Netherlands. The study focussed mostly on the southern part of the city of Arnhem and the western part of the city of Nijmegen.

The main factors determining the water quality of urban water systems in the urban drainage systems of Nijmegen were storm water runoff and upward seepage from the Meuse-Waal canal fed by water from rivers (Rhine and Meuse, Chapter 2). Upward seepage was positively correlated with nitrate, potassium, sodium and chloride and negatively correlated with alkalinity, calcium, magnesium and iron in the urban surface waters. During a rain storm, ammonium, lead, zinc and phosphorus concentrations were significantly higher in the surface water than during a dry period. The acreage of impervious area had no significant influence on the water quality of this urban surface water system.

It was tested whether macrophyte species composition and diversity in urban waters is determined by local environmental conditions and/ or the regional species pool (Chapter 3). Macrophyte species composition and diversity of urban drainage systems were compared with those in rural and semi-natural surface water systems. In the urban and rural waters several exotic species characteristic of eutrophic conditions were present. In semi-natural waters exotic species were absent. The species present here are indicators for slightly acidic and oligotrophic conditions. In the semi-natural area macrophyte species richness was higher than could be expected based on species-area relationships in rural areas. Urban macrophyte species richness was similar to rural species richness. The local macrophyte species pool might be influenced by the size and composition of the regional macrophyte species pool, but local site conditions (e.g. water quality) eventually determined the macrophyte composition and diversity in urban water systems and semi-natural water systems.

The biodiversity value of urban surface waters was compared to the biodiversity value in similar water systems in rural areas and the total species pool in the Netherlands (Chapter 4). Macroinvertebrate taxa richness, Shannon-index, number of red list species, number of exotic species and rareness were very similar in urban and rural waters. Four types of urban water bodies were distinguished based on differences in macroinvertebrate species composition, abundance, diversity and environmental conditions. The turbid water bodies and the nutrient-rich water bodies with very poorly developed vegetation showed a low macroinvertebrate taxa richness. Nutrient-poor water bodies and water bodies with a high cover of submerged vegetation showed a high macroinvertebrate taxa richness. Key factors for the macroinvertebrate assemblages in urban water systems were

the nitrate content and transparency of the surface water, substrate type (clay/ sand), and abundance of nymphaeid and submerged vegetation. Urban water systems contributed significantly to macroinvertebrate diversity in the Netherlands.

Larvae of chironomids (non-biting midges), their relationship with environmental factors and the short-term effects of dredging of polluted sediment and organic debris have been studied (Chapter 5). Three different chironomid associations could be distinguished and related to the thickness of the sludge layer, substrate type (sand/ clay), the presence of lemnids, submerged vegetation and filamentous algae, and transparency of the water. Chironomid taxa richness, Shannon-index and rareness of urban waters were higher or similar to values of these indices in water systems in rural areas. Taxa richness of chironomids increased significantly after dredging. Temporal changes in abundances of species with particular life-history strategies indicated improved oxygen conditions after dredging.

Local factors such as environmental variables or biotic interactions could influence the invasibility of urban water systems for exotic macroinvertebrate species (Chapter 6). Native species diversity and abundance were positively related to the abundance of submerged and nymphaeid vegetation (i.e. measures for environmental heterogeneity), while exotic species diversity and abundance were positively related to nutrient levels and thickness of sludge layer (i.e. measures for eutrophication indicators). Taxonomically related native and exotic crustacean species did not seem to influence each other by competition; most species showed high abundances in nutrient-rich waters. The exotic species were mostly detritivorous or omnivorous and profited from eutrophic conditions where leaf-break down and subsequently food availability were high.

In contrast to other studies, the present study showed that urban surface water systems with an impervious catchment area of approximately 30% can still support high biodiversity, including some red-list species (Chapter 7). In the study area, the environmental pressure of rain water run off was rather low, while the influence of local upward seepage of river water with low water quality was quite high. Biodiversity in the urban waters investigated, was not limited by the regional species pool, but was influenced by disturbance level, nutrient status and the abundance of submerged and nymphaeid vegetation. Invasibility by exotic species was enhanced by eutrophication. Biodiversity in urban waters can be stimulated by lowering nutrient levels, by for example regular dredging, decrease the upward seepage of polluted groundwater, the inlet of nutrient-rich water and the feeding of fish and ducks and increase of stream flow. Biodiversity can also be stimulated by enhancing helophyte, submerged and nymphaeid vegetation development, for example via the development of natural banks and optimizing the mowing regime.

Urban water system Nijmegen. Photo: Kim Vermonden

Samenvatting

In onze snel verstedelijkende wereld spelen watersystemen een belangrijke rol. Watersystemen zorgen voor drinkwater, irrigatie, bescherming tegen overstroming, infrastructuur, recreatie en habitat voor flora en fauna (**Hoofdstuk 1**). Kennis over de structuur en het functioneren van stedelijke watersystemen als habitat voor flora en fauna is nodig om de sleutelfactoren voor biodiversiteit in steden te achterhalen, zodat het ontwerp en beheer van dit soort systemen kunnen worden geoptimaliseerd. De hoofddoelstelling van deze studie was om de belangrijkste stuurvariabelen voor biodiversiteit in stedelijke watersystemen te achterhalen. Daarom hebben we onderzocht hoe lokale en regionale processen de soortensamenstelling en diversiteit van ongewervelde dieren en water- en oeverplanten beïnvloeden. Voor de biologische beoordeling van biodiversiteit in stedelijke watersystemen is gebruik gemaakt van hedendaagse ecologische kennis, zoals de soortenpoelhypothese, matige verstoring hypothese, overlevingsstrategieën van soorten en biologie van uitheemse soorten. De onderzochte stedelijke watersystemen, de zuidelijke delen van de stad Arnhem en de westelijke delen van de stad Nijmegen, bevinden zich in de voormalige uiterwaarden van de rivieren Maas en Waal, in het oosten van Nederland.

De waterkwaliteit in de stedelijke watersystemen van Nijmegen werd beïnvloed door regenwaterafvoer en kwel vanuit het Maas-Waal kanaal. Dit kanaal wordt gevoed door water van de rivieren Rijn en Maas (**Hoofdstuk 2**). Kwel was significant positief gecorreleerd met de nitraat-, kalium-, natrium- en chlorideconcentraties en significant negatief gecorreleerd met de alkaliniteit en de calcium-, magnesium- en ijzerconcentraties in het stedelijke oppervlaktewater. Gedurende een grote regenbui waren de concentraties ammonium, lood, zink en fosfor veel hoger dan gedurende een droge periode. De hoeveelheid verhard oppervlak had geen significante invloed op de waterkwaliteit van stedelijke oppervlaktewatersystemen.

De vegetatiesamenstelling en -diversiteit van water- en oeverplanten in stedelijke watersystemen kunnen o.a. worden bepaald door lokale omgevingsfactoren of de regionale soortenpoel (**Hoofdstuk 3**). Vegetatiesamenstellling en -diversiteit in stedelijke watersystemen werden vergeleken met die in rurale en half-natuurlijke watersystemen. In de stedelijke en rurale watersystemen waren verschillende uitheemse soorten, karakteristiek voor voedselrijke omstandigheden, aanwezig. In de half-natuurlijke wateren waren uitheemse soorten afwezig. Hier komen vooral soorten voor die karakteristiek zijn voor lichtzure en voedselarme omstandigheden. In de half-natuurlijke wateren was de soortenrijkdom van water- en oeverplanten hoger dan verwacht kon worden op basis van relaties tussen soortenrijkdom en oppervlakte van watersystemen in rurale gebieden. De soortenrijkdom van water- en oeverplanten in stedelijke watersystemen was vergelijkbaar met die in rurale gebieden. Hoewel de lokale soortenrijkdom van water- en oeverplanten beïnvloed kan worden door de grootte en samenstelling van de regionale soortenpoel, bleken vooral lokale omgevingsfactoren (waaronder waterkwaliteit) bepalend te zijn voor de lokale soortensamenstelling en diversiteit in stedelijke en half-natuurlijke watersystemen.

De biodiversiteit van macroevertebraten in stedelijke watersystemen werd vergeleken met die in watersystemen in rurale gebieden en de totale soortenpoel in Nederland (**Hoofdstuk 4**). Soortenrijkdom, Shannon-index, aantal rode lijst soorten, aantal uitheemse soorten en zeldzaamheid van de macroevertebratenfauna waren ongeveer gelijk in stedelijke en rurale watersystemen. Vier stedelijke watersysteemtypen werden onderscheiden op basis van verschillen in soortensamenstelling, abundantie en diversiteit van de aquatische macroevertebratenfauna en omgevingsfactoren. De troebele wateren en de meest nutriëntenrijke wateren met zeer weinig vegetatie hadden een lage macroevertebratendiversiteit. De nutriëntenarmere systemen en de wateren met veel ondergedoken vegetatie hadden een hoge macroevertebratensoortenrijkdom. Sturende factoren voor de soortensamenstelling van de macroevertebratenfauna in stedelijke watersystemen waren de nitraatconcentratie, het doorzicht van het water, het soort substraat (klei vs. zand), en bedekkingspercentage van waterplanten met drijfbladeren (bijv. waterlelies) en ondergedoken vegetatie. Stedelijke watersystemen dragen significant bij aan de macroevertebratendiversiteit in Nederland.

Larven van chironomiden (dansmuggen), hun relatie met het milieu en de korte-termijn effecten van baggerwerkzaamheden op de chironomiden zijn in meer detail geanalyseerd (**Hoofdstuk 5**). Drie verschillende associaties van chironomiden konden worden onderscheiden en gerelateerd aan de dikte van de baggerlaag en substraattype (zand vs. klei), de abundantie van kroos, ondergedoken vegetatie, algen en het doorzicht van het water. Soortenrijkdom, Shannon-index en zeldzaamheid van de chironomiden waren ongeveer gelijk in stedelijke en rurale watersystemen. De soortenrijkdom van chironomiden ging significant omhoog na baggerwerkzaamheden. Veranderingen over de jaren in chironomiden met bepaalde overlevingsstrategieën indiceerde dat zuurstofcondities verbeterden na het baggeren.

De abundantie en diversiteit van uitheemse macroevertebratensoorten in stedelijke watersystemen werd onderzocht en gerelateerd aan de omgevingsfactoren (**Hoofdstuk 6**). Lokale factoren zoals abiotische factoren of biotische interacties kunnen de potenties van stedelijke watersystemen voor kolonisatie door uitheemse soorten beïnvloeden. De diversiteit en abundantie van inheemse macroevertebratenfauna werden positief beïnvloed door de hoeveelheid ondergedoken en drijfbladvegetatie (maat voor heterogeniteit van het milieu), terwijl de diversiteit en abundantie van uitheemse soorten positief werden beïnvloed door nutriëntenhoeveelheden en de dikte van de baggerlaag (een maat voor eutrofiëring). Taxonomisch verwante, inheemse en uitheemse kreeftachtigen leken elkaar niet te beïnvloeden door competitie. Beiden waren vooral aanwezig in nutriëntrijke wateren. De uitheemse soorten waren vooral detritivoor of omnivoor en profiteerden van nutriëntrijke omstandigheden waar de afbraak van organisch materiaal en daardoor de voedselbeschikbaarheid hoog waren.

In tegenstelling tot verwachtingen op basis van literatuur, liet dit onderzoek zien dat oppervlaktewatersystemen in steden met een verhard oppervlakte van ongeveer 30% nog steeds een hoge biodiversiteit kunnen herbergen (**Hoofdstuk 7**). In de onderzochte watersystemen bleek de invloed van vervuild regenwater relatief laag, terwijl de invloed van lokale kwel van vervuild rivierwater relatief groot was. De biodiversiteit in de onderzochte, stedelijke wateren werd niet beperkt door de regionale soortenpoel, maar wel door het verstoringsniveau, de nutriëntenrijkdom en de abundantie van ondergedoken en drijvende vegetatie. Eutrofiëring vergroot de kans op vestiging van uitheemse soorten. Een hogere biodiversiteit in stedelijke wateren kan worden gestimuleerd door het verlagen van nutriëntenconcentraties, door bijvoorbeeld regelmatig baggeren, verminderen van de kwel van vervuild riverwater, beperking van de inlaat van nutriëntenrijk water en het voederen van vissen en eenden en het verhogen van de stroomsnelheid van het water. Een hogere biodiversiteit kan ook gestimuleerd worden door de ontwikkeling van ondergedoken en drijfblad- en oevervegetatie, door bijvoorbeeld de aanleg van natuurvriendelijke oevers en het optimaliseren van het maaibeheer.

Urban water system Arnhem. Photo: Kim Vermonden

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Kim

Field monitoring in Nijmegen. Photo: Nico Vermonden

Curriculum vitae

Kim Vermonden was born on the 20th of November 1980 in Breda, The Netherlands. After finishing high school at the Newmancollege in Breda in 1999, she started studying Forest and Nature Management at Wageningen University, where she did the following master studies. In her first master project at Bargerveen Foundation she studied the aquatic food web in raised bogs in Estonia and The Netherlands with stable isotope analysis. For her second master project at the Department of Animal Ecology and Ecophysiology, Radboud University Nijmegen she travelled to Curaçao (Netherlands Antilles) to analyse changes in the condition of coral reefs and fish populations, comparing the situations between 1973 and 2003. The last part of her study existed of an internship at the Royal Netherlands Institute for Sea Research (Texel, The Netherlands), where she investigated the exploitation competition in differently-sized shore crab (*Carcinus maenas*).

After obtaining her MSc degree in November 2004 at Wageningen University, she started working as a junior researcher at the Department of Environmental Science, Radboud University Nijmegen in January 2005. The project involved identifying the key factors for biodiversity of urban surface water systems. The first three years were spent mainly in the field and laboratory, sampling and identifying macroinvertebrates, measuring water and sediment quality and supervising bachelor and master students. In January 2008 she moved to Denmark and continued her work for the Radboud University at the Freshwater Biological Laboratory of the Copenhagen University in Hillerød. During the first part of 2008 she analysed the data and communicated the results and implications for the management of urban water systems to the municipalities of Arnhem and Nijmegen and their partners in the EU Interreg IIIb programme urban water. In October 2008 she gave birth to a daughter, named Famke. After her maternity leave, she started working again in January 2009 on the remaining chapters of her PhD-thesis in collaboration with her Dutch and Danish colleagues. She is currently working at Biogen Idec in Hillerød at the department QC Bioanalytical.

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