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the challenges of controlling water quality**

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Sustainable urban stormwater management
**THE CHALLENGES OF
CONTROLLING WATER QUALITY**

Ph.D. thesis · 2011 · Simon Toft Ingvertsen

Sustainable urban stormwater management

The challenges of controlling water quality.

Ph.D. thesis – 2011
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Title

Sustainable urban stormwater management
- The challenges of controlling water quality.

Ph.D. thesis 2011

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Preface

This Ph.D. project was carried out in the period from February 2008 – September 2011 at the Section of Plant and Soil, Department of Agriculture and Ecology, Faculty of Life Sciences, University of Copenhagen. It was partly funded by the 2BG (Black, Blue, and Green) research project as well as by the faculty and the department.

The project was intended to address topical end-user challenges regarding urban stormwater quality control in the integrated planning of sustainable urban water systems. During the course of the project there were many interactions with end-users and other professionals in the field of urban stormwater management. This was reflected in many of the decisions made during the period and most of the project was therefore an iterative process ultimately leading to the current product. Thus, in addition to discussing and concluding on the conducted research, this thesis also gives an account of the current state of the art knowledge and the challenges faced by decision makers in relation to urban stormwater quality. Two published research papers as well as two submitted manuscripts are appended in this thesis.

Acknowledgements

I would like to express my utmost gratitude to my principal supervisor Jakob Magid as well as my project supervisor Marina Bergen Jensen. You complimented each other very well in the process and consistently found the time to respond seriously to any of my requests. Together, you have been able to find a good balance between supervision and informal talks as well as between help and the promotion of self-help. I have learned a lot from you and I look forward to continued collaboration with the both of you.

During the course of the project I have had the pleasure of spending time and collaborating with fellow Ph.D. students and colleagues within the 2BG research project. I am thankful for the good times spent, the collaboration during the case studies, and valuable discussions at various gatherings and other occasions.

Thanks to my fellow Ph.D. student and friend, Karin Cederkvist, for nice teamwork and good times in the field, in the laboratory, and on the e-mail.

I would also like to express to my fellow Ph.D. students and colleagues at the department of Agriculture and Ecology that I sincerely enjoy your company and the atmosphere that you create. My wish to keep working at this department for another period can to a great extent be ascribed to you. A special thanks to my office and coffee mate Simon Mundus, who can actually say some funny as well as clever things once in a while.

Thanks to Harald Sommer and his colleagues at the *Ingenieurgesellschaft Prof. Dr. Heiko Sieker mbH* in Hoppegarten, Berlin, whose collaboration and great help in establishing contact to the relevant authorities and obtaining permissions for collecting samples was invaluable to this project.

Finally, and most importantly, my family: Julie and Laurits. I hope you know how much I mean it when I say, from the bottom of my heart, that I'm eternally grateful for your exceptional patience and support, especially during the final months of the project. Det er ekte kjærighet.

Summary

Management of urban water needs to be adapted to more dynamic precipitation patterns. This may be approached through integrated planning including aspects such as sustainability, climate adaptation, the natural hydrological freshwater cycle, and the qualities of urban life. Around the world this has promoted novel ways of managing urban stormwater runoff, namely by implementing decentralised sustainable urban drainage systems (SUDS). Based primarily on local or on-site retention and infiltration these systems take many forms in order to manage the quantities of urban stormwater runoff, and while the hydraulic performance of most SUDS is generally well understood, controlling the quality of the water is still a major challenge. This is linked with highly variable pollutant profiles of urban stormwater, rigorous water quality demands, and lack of knowledge and decision support concerning the treatment efficiency and lifetime expectancy of various SUDS.

Hence, the aim of this Ph.D. project was to improve the basis for future decision making with regard to water quality aspects in the integrated planning of sustainable urban water systems. During the course of the project, three central challenges for future decision making regarding the quality of urban stormwater were identified: (#1) being able to predict pollutant concentrations in urban stormwater, (#2) defining emission limit values for stormwater discharges, and (#3) selecting appropriate treatment options. These challenges are addressed and analysed individually in this thesis. The primary research focus was placed within challenge #3, more specifically on improving the conditions for documenting and benchmarking the treatment performance of existing SUDS, while thorough understanding of challenge #1 and #2 was considered a prerequisite for conducting this research successfully. The objective was approached on three different levels: A general level involving review and interpretation of existing literature and data sets on urban stormwater quality, case studies at the catchment level, and experimental assessments of methods and treatment efficiency on single facility level.

In order to address challenge #1 available literature and data sets on urban stormwater quality from around the world was thoroughly investigated. Each of the five contaminant groups, ‘suspended solids (SS)’, ‘heavy metals’, ‘xenobiotic organic compounds (XOC)’, ‘nutrients, organic matter, and salt’, and ‘pathogens’ are reviewed with respect to basic knowledge and recent findings. Several attempts have been made to simplify methods for predicting pollutant concentrations in urban runoff, i.e. based on land use or urban typologies, development of surrogate parameters to limit the number of relevant parameters, and software modelling of pollutant build-up and wash-off. However, while it may be possible to develop acceptable catchment specific predictive models based on local monitoring, the portability of such results often turns out poor. An account is given of the factors known to frequently influence the pollutant profile of urban runoff. The sum of uncertainties imposed by the numerous factors yields substantial local and regional variations, making accurate predictions of individual pollutant profiles virtually impossible unless detailed catchment specific monitoring programmes are carried out in advance. These indications were confirmed in case studies carried out within this project on specific catchments in the Danish cities Odense and Copenhagen. In terms of monitoring, sampling methods and frequency are shown

to induce significant variations in observed pollutant concentrations limiting the reliability of monitoring results. In order to make this aspect more transparent to decision makers a reliability classification system could be developed, i.e. based on sampling method as well as number and nature of events sampled.

Challenge #2 was only addressed briefly as this was not directly part of the research objectives of the project. However, a look into the legal framework concerning water quality revealed that little national as well as international regulation specifically address urban stormwater discharges. Many decisions are left to local authorities in order to comply with international framework directives. Thus, emission limit values will need to consider each receiving water body individually, in order to reach appropriate management standards. Currently there is lack of knowledge concerning the presence and behaviour of a range of the EU priority pollutants in urban runoff. For some of the critical pollutants the sensitivity in standard analysis packages does not yet allow for valid assessment of compliance with water quality standards. This is namely the case for tributyltin (TBT) and some of the high molecular weight PAH compounds. However, recently published national water quality standards for the heavy metals copper and zinc may also be difficult to comply with if urban stormwater discharges constitute a significant part of the input to a water body. Thus, if national and international water quality standards are enforced to the letter they could seriously limit the implementation of SUDSs in Danish cities. The overall challenge seems to be taking all the prevailing uncertainties into consideration without limiting the possibilities for implementing SUDS. Concerning emission limit values, it is suggested that for a yet undefined period of time (i.e. 5 to 10 years) authorities need to define case specific emission limit values for each SUDS employed. However, in this time period we should aim at gaining sufficient experience and documentation regarding the treatment efficiency of a range of SUDS to enable implementation of design criteria (best available technology) rather than emission limit values.

The main work load in the project was placed within challenge #3: Selecting appropriate treatment options. Possible approaches to selection tools and comparison of treatment performance among SUDS are reviewed and discussed and an account is given of the numerous factors that influence treatment performance. It was found that, among other things, inconsistent use of water quality parameters in monitoring programmes hampered the potential for valid comparison of treatment efficiency. Through a review of available data sets on urban stormwater quality a minimum data set of water quality parameters is suggested for consistent use in future monitoring programmes to ensure broad-spectrum testing and comparable data sets at low costs. The proposed minimum data set includes: (i) fine fraction of suspended solids ($< 63 \mu\text{m}$), (ii) total concentrations of zinc (Zn) and copper (Cu), (iii) total concentrations of phenanthrene, fluoranthene and benzo(b,k)fluoranthene, and (iv) total concentration of phosphorus (P) and nitrogen (N). Indicator pathogens and other specific contaminants (i.e. chromium, pesticides, phenols) may be added if recreational or certain catchment scale objectives are to be met.

Recognising that trafficked areas such as roads and parking lots frequently make up a significant proportion of the impervious areas in cities there is a need for SUDS with sufficient treatment efficiency to ensure the quality of discharges. Roadside infiltration systems employing an

engineered top soil layer (filter soil) designed for both infiltration and treatment purposes have been in operation for several years in Germany and may constitute a promising drainage option in a Danish context. However, their long-term treatment efficiency and expected lifetime is not yet well investigated. Consequently, a number of existing German infiltration systems were assessed with respect to basic characteristics, heavy metal and P content, leaching potential of dissolved organic matter (DOC), heavy metals, and P, as well as treatment efficiency towards fine SS and dissolved heavy metals. The two latter tasks were achieved by laboratory experiments performed on intact soil columns collected from existing roadside swales. As a novel approach to the testing of treatment efficiency, fluorescent microspheres were used as surrogates for fine suspended solids which are considered a crucial parameter for the overall treatment efficiency of SUDS. The method proved to work well and thus, allowed for a distinction between added and in-situ mobilised solids. Overall, the filter soils were barely polluted with respect to heavy metals in spite of many years of operation. The treatment efficiency of the tested soils proved to be high for fine suspended solids, Cd, Cu, and Zn, but not for chromium (Cr) which appears to pass through the soil as chromate. On the other hand, in-situ mobilisation of DOC and possibly inorganic colloids resulted in critical effluent concentrations of Cu, Zn, and Pb (lead). Thus, in order to improve the overall efficiency of filter soils for infiltration of polluted urban runoff, it is suggested to conduct further research on the interactions between soil pH, base cation saturation, and content of iron and aluminium oxides which are found to be the major controlling parameters for DOC mobilisation.

This Ph.D. project has led to rising national interest in the possibilities for using engineered filter soil for infiltration of runoff from trafficked areas, and under a new national partnership for climate adaptation and innovation (www.vandibyer.dk/english/) a project has been initiated with the aim of testing and developing filter soil solutions in a Danish context. Furthermore, many of the findings of the project, including the suggested minimum data set of water quality parameters for assessing and comparing treatment efficiency of SUDS may feed into planned research within an innovation consortium on water quality assurance with regard to urban stormwater management.

List of publications

Appended publications:

- I. **Ingvertsen, S.T.**, M.B. Jensen, and J. Magid. 2011a. A Minimum Data Set of Water Quality Parameters to Assess and Compare Treatment Efficiency of Stormwater Facilities. *J. Environ. Qual.* 40(5):1488-1502.
- II. Fryd, O., M.B. Jensen, **S.T. Ingvertsen**, J. Jeppesen, and J. Magid. 2010. Doing the first loop of planning for sustainable urban drainage system retrofits: A case study from Odense, Denmark. *Urban Water J.* 7(6): 367-378.
- III. **Ingvertsen, S.T.**, K. Cederkvist, Y. Régent, H. Sommer, J. Magid, and M.B. Jensen. 2011. Assessment of Roadside Swales with Engineered Filter Soil. Part I. Characterisation and Lifetime Expectancy. Submitted manuscript (*J. Environ. Qual.*).
- IV. **Ingvertsen, S.T.**, K. Cederkvist, M.B. Jensen, and J. Magid. 2011b. Assessment of existing roadside swales with engineered filter soil. Part II. Treatment efficiency and in-situ mobilisation in intact soil columns. Submitted manuscript (*J. Environ. Qual.*).

Other publications:

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Cederkvist, K., **S.T. Ingvertsen**, and M.B. Jensen. 2010. Engineered filter soil for treatment of road runoff (Filterjord til rensning af vejvand). *Vand & Jord.* 17(4):142-144 (In Danish).

Ingvertsen, S.T., H. Sommer, K. Cederkvist, Y. Régent, M.B. Jensen, and J. Magid. 2010. Infiltration and treatment of urban stormwater: how well do swale-trench systems work? In: Proc. of 7th Novatech, 27th June – 1st July 2010, Lyon, France.

Fryd, O., A. Backhaus, J. Jeppesen, M. Bergman, **S.T. Ingvertsen**, H. Birch, T. Panduro, and C. Fratini. 2009. River Harrestrup Case Study: Connected disconnections – Conditions for landscape-based disconnections of stormwater from the Copenhagen sewer system in the catchment area for Harrestrup Å. Working report from the 2BG project, December 2009. Available online: http://www.2bg.dk/Publications/2BG_HarrestrupAa_Booklet_UK_web.pdf

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Introduction

Background

For several decades sewers have been playing an important part in ensuring a hygienic and safe environment in most urban areas. Conventional centralised sewer systems have conveyed both wastewater from households and industries as well as stormwater runoff from impervious surfaces to wastewater treatment plants or directly to nearby surface water bodies. However, during the latest decades, increasingly frequent sewer overflows and local flooding have occurred due to excessive volumes of stormwater exceeding the capacity of sewer systems. This is caused by a combination of changing rainfall patterns, growing urbanisation, and a tendency to extend the fraction of paved areas within cities. And there is no reason to believe that these trends are reversing in the near future.

According to reports from the Intergovernmental Panel on Climate Change (IPCC) the average annual precipitation during the period 1901 to 2005 has increased in several parts of the world (Fig. 1a), namely Northern Europe (Fig. 1b), most of North America, Northern and Central Asia, North-western Australia, and South-eastern South America. Most regions on the northern hemisphere have experienced an upward linear trend of between 6 and 8% in the period 1901 to 2005 (Trenberth et al., 2007). The projections of future annual precipitation in Northern Europe (Fig. 2a) indicate that in Denmark average precipitation during winter months will increase by 10 to 15% from the period 1980-1999 to the period 2080-2099 (Christensen et al., 2007). On the contrary, the average precipitation during summer will decrease slightly. Thus, on annual basis, the amounts of precipitation in Denmark are not expected to increase more than 5% by the year 2080. However, the annual precipitation has little to say in regard to sewer system capacity which is based on rainfall statistics, more specifically the return period of rain events of a certain intensity (volume per time unit). Arnbjerg-Nielsen (2006) has demonstrated that during the 20-year period from 1979 to 2000 there has been a statistically significant increase in the maximum intensity and frequency of 10 minute rain storms. It should be noted that this trend was most pronounced for the eastern part of the country. It is fair to say that this trend has been further underlined in recent years with three major cloud-bursts within the area of greater Copenhagen since 2007 causing severe flooding and damage in large areas. And climate models continue to predict that in a future climate with rising

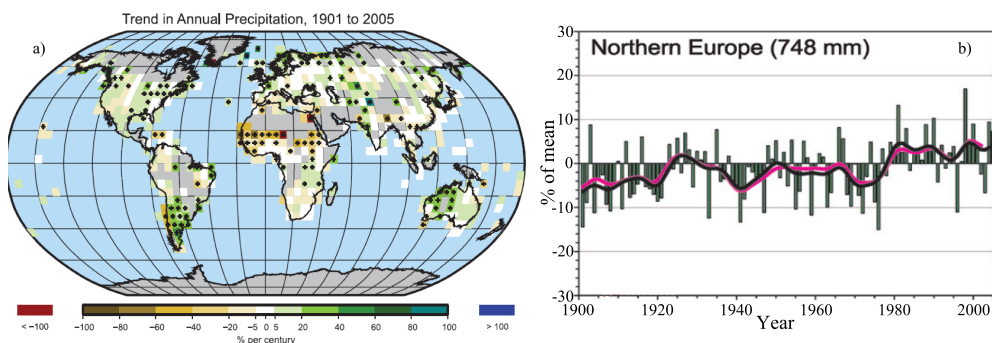


Fig.1 Observed changes in precipitation from 1901 to 2005. Adapted from Trenberth et al., (2007). (a) Changes in annual precipitation in percent per century. Grey areas signify insufficient data to identify reliable trends. (b) Time series of annual precipitation for Northern Europe shown as the percent of the mean value from 1961-1990 (748 mm)

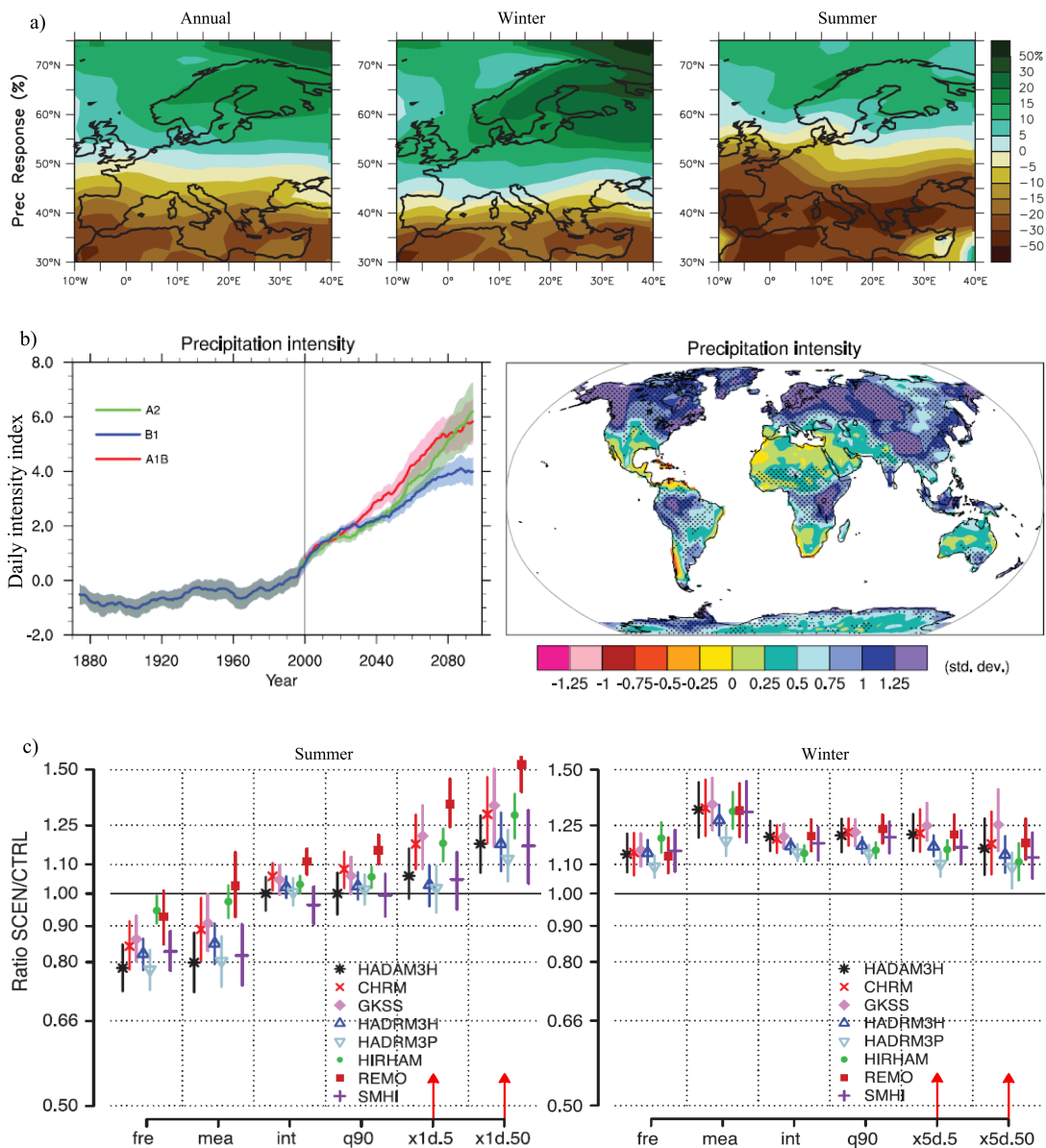


Fig. 2. Projected changes in precipitation patterns. (a) Precipitation changes (%) over Europe between the periods 1980-1999 and 2080-2099 for a medium emission scenario (A1B). Averaged over 21 models. From Christensen et al. (2007). (b) Adapted from Tebaldi et al. (2006). Left: average global changes in precipitation intensity for three future emission scenarios (B1: low, A1B: medium, A2: high) based on 9 individual models. The shaded areas signify the standard deviation. Right: changes in average spatial patterns of multi-model simulated precipitation intensity represented by the difference between two twenty-year averages (2080-2099 minus 1980-1999). Changes are given in units of standard deviations of an ensemble ($n=9$) of normalised models. (c) Changes in mean (with 95% confidence interval) summer and winter precipitation in Southern Scandinavia for the A2 emission scenario (ratio between 2071-2100 "SCEN" and 1961-1990 "CTRL") as predicted by eight individual models. fre = wet-day freq., mea = mean seasonal precip., int = mean wet-day precip., q90 = 90th percentile of wet-day precip., x5d.5 and -.50 = 5- and 50-year return values of five-day precip., x1d.5 and -.50 = 5- and 50-year return values of one-day precip. Adapted from Christensen et al. (2007).

temperatures the precipitation intensity will also increase over most regions, particularly on the northern hemisphere (Fig. 2b) (Meehl et al., 2007). In the context of these models, precipitation intensity is defined as the annual total precipitation divided by the annual number of rainy days. In Denmark the most intense rain events are expected to occur during summer, and multi-model simulations predict that the intensity of summer rain events with a 50-year return period will be between 12 and 50% higher in the period 2071-2100 compared to the period 1961-1990 (Fig. 2c). Specifically for Denmark, this range was confirmed by van Roosmalen et al. (2010). This has led to recommendations for introducing climate factors into the existing dimensioning practises of urban drainage systems (SVK 2008). In addition to the increasing storm intensities and urban growth, the tendency to increase the amount of paved surfaces in urban areas contributes to the drainage problems by increasing stormwater runoff peak flows.

The changing hydrological conditions constitute a major driver which has led to much debate regarding future practices for urban stormwater management, and ideas about a paradigm shift towards more sustainable systems have advanced. Such ideas are further strengthened by a number of additional drivers: (i) Undersized sewer systems results in an increased number of sewer overflows which discharge untreated storm- and wastewater directly into the aquatic environment deteriorating water and sediment quality in streams, lakes, and coastal waters (Hvidtved-Jakobsen, 1982; Eganhouse and Sherblom, 2001; Suárez and Puertas, 2005), (ii) With conventional sewer systems, potentially valuable freshwater is being conveyed out of the city without consideration to recharge of groundwater reservoirs and the potential benefits which could be gained with respect to recreational assets or reuse of the water, i.e. van Roon (2007) (iii) Stormwater in combined sewer systems may compromise some of the vital processes necessary for efficient treatment at centralised wastewater treatment plants, namely in the case of prolonged rain events where the high hydraulic loadings can affect the performance of the secondary clarifier resulting in loss of sludge to the receiving water body (Rauch and Harremoes, 1996; Jack and Ashley, 2002), (iv) In terms of energy consumption, the large quantities of stormwater entering wastewater treatment plants are a costly affair which could be reduced, i.e. by increased local treatment (Clauson-Kaas et al., 2009), and (v) In the long run, there is a risk that the overall costs of the current approach to urban drainage may be substantial for forthcoming generations as the here-and-now investments in conventional sewer systems are massive and often do not consider whole life services (Ashley and Hopkinson, 2002). In combination, these drivers have promoted the implementation of sustainable urban drainage systems (SUDS) all across the world, namely in the United States, Australia, United Kingdom, The Netherlands, France, and Germany. The ideas and implementation of SUDS are currently advancing in Denmark.

Sustainable Urban Drainage Systems (SUDS)

Contrary to sewers which rely on centralised systems and typically have a lifetime of approximately 100 years, SUDS are primarily based on local or on-site management of urban stormwater runoff and can be implemented according to potentially changing rain patterns with relatively short notice. The systems may basically employ four physical mechanisms to control the quantities of water:

- (i) Storage in ponds or basins. The water can be slowly released back to the sewer system or to other SUDS.
- (ii) Infiltration into the subsoil. The water percolates to groundwater reservoirs or drain pipes leading to nearby surface water bodies.
- (iii) Evaporation. A fraction of the water from a variety of SUDS will potentially leave the system as vapour, but no SUDS rely entirely on evaporation. Plants typically enhance evaporation through evapotranspiration.
- (iv) Conveyance. Transport the runoff between impervious surfaces and SUDS or between individual SUDS.

A range of common SUDS employing storage and infiltration are sketched in Fig. 3. Storage ponds are commonly divided into *dry/wet* or *detention/retention* ponds, the former referring to ponds with the outlet at the bottom, i.e. an outlet pipe or infiltration; while the latter refers to ponds that are continuously water-filled with a minimum water level determined by the height of a membrane or outlet pipe. Infiltration systems come in various sizes and shapes, but commonly one distinguishes between *surface* and *subsurface* infiltration systems. Surface systems such as infiltration swales and -basins, rain gardens, and permeable pavements allow for more flexibility in that the top soil layer can be monitored, adjusted, and replaced in case of poor performance. Similar operations are harder to perform in subsurface systems like soakaways and trenches, but on the other hand, such systems do not occupy potentially useful surface area. All the SUDS displayed in Fig. 3 exert some level of evaporation which varies according to temporal conditions. In addition to relatively low technology SUDS such as those in Fig. 3 there are a number of more technical and typically more costly installations available, i.e. a range of different separators, technical basins (Vollertsen et al., 2009), and dual porosity filtration (Jensen et al., 2011). For a more comprehensive list of the most common SUDS, see The SUDS Manual (Woods-Ballard et al., 2007) or in the Danish method catalogue for

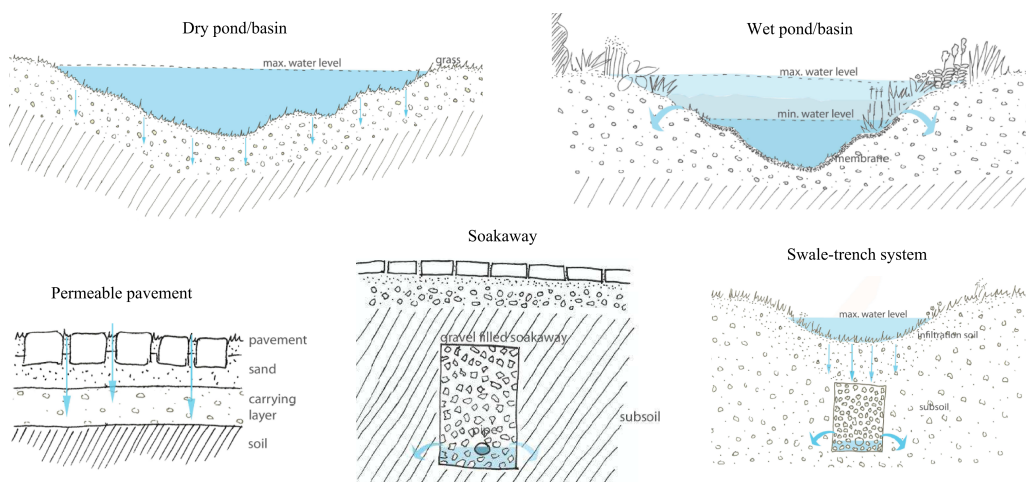


Fig. 3. Examples of common sustainable urban drainage systems employing storage and infiltration. Sketches by Antje Backhaus. Adapted from Jensen et al. (2010b).

SUDS published by the Municipality of Copenhagen (KK, 2009). In terms of hydrology, there are standard procedures for how to size and design SUDS (i.e. Woods-Ballard et al., 2007; Jensen et al., 2010), but they almost all require some degree of regular inspection and maintenance in order to sustain adequate hydraulic performance.

While the hydraulic performance of most SUDS is generally well understood and can be adjusted to a certain service level based on rain statistics and fairly simple calculations, understanding and controlling the quality of the water is still a major challenge in the field of sustainable urban stormwater management. There are a number of reasons for this: (i) Urban stormwater falling on impervious surfaces such as roofs, parking lots, roads, public squares, etc., potentially picks up a wide range of pollutants commonly found in the urban matrix, but the pollutant profile is highly variable among surfaces and urban typologies (Duncan, 1999; Göbel et al., 2007), (ii) Urban pollutants differ substantially with respect to occurrence, partitioning, inherent properties, and toxicity, rendering their potential to be efficiently removed in SUDS extremely variable (Eriksson et al., 2007; **paper I**), (iii) Most SUDS, regardless of their intended function, exert some level of stormwater treatment through common physical, chemical, and biological processes such as filtration, sedimentation, volatilisation, adsorption, biotic and abiotic degradation, and plant uptake, but the effectiveness of these processes depends widely on local conditions; hence, the treatment efficiency of analogous SUDS towards individual pollutants may vary substantially among locations (Barret, 2008; Fassman, 2011), (iv) The rising interest for using stormwater for recreational purposes in urban areas as well as the obligation of member states in the European Union to achieve good ecological and chemical state in surface and groundwaters (Water Framework Directive, 2000/60/EC) through rigorous international water quality standards (2008/105/EC) further underlines the challenge of controlling the quality of urban stormwater discharges.

Research project: Black, Blue, and Green (2BG)

A Danish five-year research project was initiated in 2007 under the title *Black, Blue, and Green – Integrated infrastructure planning as key to sustainable urban water systems (2BG)*. The overall aim of the project was to contribute to a sustainable global development by demonstrating innovative methods for urban water management. The project was built on the hypothesis that a paradigm shift may be needed in Denmark towards more sustainable urban water systems in order to adapt cities to more frequent high-intensity thunderstorms and increasing urbanisation with emphasis on decentralised systems (SUDS), integrated interdisciplinary planning, water quality control, and local hydrological premises. Taking urban stormwater as point of departure, the consortium consisted primarily of universities, private companies and end users represented by municipalities and water utility companies.

The present PhD project, which is concerned with treatment options for urban stormwater, is one of eight PhD projects in total, all together exploring the hydrology, water quality as well as the social, economical and political aspects of SUDS.

Research objectives

In light of the prevailing uncertainties related to urban stormwater quality and the treatment potential of decentralised sustainable urban drainage systems (SUDS) it is the overall aim of this PhD thesis to improve the basis for future decision making with regard to water quality aspects in the integrated planning of sustainable urban water systems. During the course of the project, three central challenges for future decision making regarding the quality of urban stormwater were identified: (#1) being able to predict pollutant concentrations in urban stormwater, (#2) defining emission limit values for stormwater discharges, and (#3) selecting appropriate treatment options. These challenges are addressed and analysed individually in this thesis. The primary research focus was placed within challenge #3, more specifically on improving the conditions for documenting and benchmarking the treatment performance of existing SUDS, while thorough understanding of challenge #1 and #2 was considered a prerequisite for conducting this research successfully. The following four research objectives were defined:

- (i) Understand the dynamics of urban stormwater pollutant profiles and the interactions with urban typologies.
- (ii) Analyse and improve the conditions for benchmarking the treatment performance of existing SUDS
- (iii) Develop and test methods to document the treatment performance of SUDS.
- (iv) Develop tools for planning and decision support regarding urban stormwater quality.

The first objective was based on the hypothesis that pollutant profiles of urban stormwater could be grouped according to a set of different urban surfaces or typologies. This was addressed in a literature review and a case study (**paper I and II**), and since much effort was put into this subject during the entire project period, it is also comprehensively emphasised in the thesis. The second objective was approached in an iterative manner resulting in the proposal of a minimised data set of water quality parameters for future performance assessments and comparison among SUDS (**paper I**). Existing roadside infiltration swales employing engineered filter soil was selected as a SUDS to meet the third research objective which is addressed in **paper III and IV**. The fourth and final objective was only partially met in that a specific tool has not yet been developed. However, during the project a number of interactions with local authorities being in the process of making decisions have taken place, and lessons learned from that are reflected in some of the non-peer reviewed papers. Furthermore, given the nature and approach of this project, the present thesis may provide much useful information for decision makers and other professionals in the field of urban stormwater management.

Challenge #1: Predicting the pollutant profile

Information about the occurrence and concentrations of pollutants in urban stormwater runoff is essential in the planning of decentralised urban water systems, i.e. what type and level of treatment is required? This can either be obtained by systematic sampling of runoff, or by relying on predictions that are based on experience or extrapolation from other available data. However, while monitoring runoff samples would be the preferred approach in terms of validity it can be a rather costly affair. Thus, a predictive approach would often be preferred by authorities with a limited budget, although it may result in uncertain predictions. A frequent consequence of this approach is employment of the precautionary principle which can ultimately lead to overly priced treatment facilities or, in extreme cases, rejection of projects. This section seeks to provide insight in the current understanding and development of predicting urban stormwater pollutant profiles.

Pollutants and their sources

A first step towards predicting the pollutant profile of urban runoff is obtaining thorough knowledge about the potentially occurring pollutants and their sources. In the antecedent dry periods between rain events a variety of pollutants continuously accumulate on urban surfaces such as roofs, building facades, parking lots, and roads. But as rain falls and the water flows across the surfaces most of the accumulated pollutants are picked up and the quality of the seemingly unpolluted stormwater may be drastically diminished. How much of the accumulated pollutant load is actually mobilised depends largely on the amount and intensity of rain. Table 1 displays a range of pollutants that are commonly found in urban stormwater runoff from various surfaces as well as their sources. A description of each of the pollutant groups is provided in the text below. The occurrence and behaviour of pollutants is largely covered in **paper I**, but to the extent it is found reasonable, additional information is accounted for here, i.e. newly published data or studies beyond the scope of **paper I**.

Table 1. Overview of potential urban stormwater pollutants and their sources in the urban environment.

Pollutant groups	Examples of common parameters	Examples of common sources
Suspended solids / particles	Size can vary between <0.45 and 10,000 µm.	Wear and corrosion of materials, combustion, wind deposited soil and dust, plant debris (i.e. leaves).
Heavy metals	Cadmium, chromium, copper, lead, zinc.	Wear of tires and break linings, engine oil, corrosion of car parts, crash barriers, and signs, metal roofs and down pipes, industries (i.e. spillage).
Xenobiotic organic compounds	Polycyclic aromatic hydrocarbons (PAH), phenols, pesticides, phtalates.	Combustion, industries (i.e. spillage), wear of materials, volatilisation and release from car parts and building materials, weed and algae control.
Nutrients, organic matter, and salts	Phosphorus, nitrogen, organic matter, salt (electric conductivity).	Plant debris, animal droppings, combustion (NO _x), fertilisation, road salts.
Pathogens	E. coli, enterococci	Pets, birds, seldom humans (except for leaking septic tanks).

Suspended solids

Suspended solids (SS) are practically always present in urban stormwater runoff (Duncan, 1999; Göbel et al. 2007) and often at sufficiently high loadings to have a direct impact on receiving waters and limit the use of stormwater for recreational purposes. The total concentrations of SS vary substantially among urban catchment types, from a few mg L⁻¹ to thousands of mg L⁻¹, and with trafficked areas frequently contributing the highest concentrations (Maestre and Pitt. 2005). Typical median concentrations reported in the literature are in the range of 150 – 200 mg L⁻¹ (Duncan, 1999; Göbel et al., 2007). SS can play a vital role for the dynamics of the pollutant profile as they often serve as carriers of a wide range of adsorbed or occluded pollutants, i.e. heavy metals and hydrophobic organic compounds. Thus, the fate of many pollutants depends on the mobilisation of solids from urban surfaces as well as the retention of SS in SUDS. The degree of particle association is highly variable among pollutants, but large inter-study differences are also observed for the particle association of individual pollutants. The distribution of pollutants over particle sizes in urban runoff is often skewed towards the fine fraction solids, i.e. below 50 or 100 µm in diameter. These issues are addressed in further detail for heavy metals and PAH in **paper I**.

Heavy metals

Heavy metals are considered to be of primary concern due to their ubiquity in urban stormwater runoff (Duncan et al. 1999; Göbel et al. 2007; Eriksson et al. 2007), high toxicity, and tendency to accumulate in organisms as well as aquatic and terrestrial environments. Compared to most xenobiotic organic pollutants they are uncomplicated and cheap to analyse, and the five most studied heavy metals, Cd, Cr, Cu, Pb, and Zn exert quite different behaviours with respect to speciation and partitioning among solid and liquid phases. Thus, they may serve as good indicators of the general pollution level of urban stormwater as well as be useful in terms of documenting the treatment performance of SUDS (**paper I**). In **paper I** the median runoff concentrations of heavy metals from roofs and roads are shown in a table (Table 3) for three major review studies on observed stormwater concentrations around the world. In Table 2 below the entire concentration ranges of Danish and international studies are shown in addition to the results of a newly published study on micropollutants in stormwater runoff in the Copenhagen area (Birch et al., 2011). There are few Danish data sets available and the studies mentioned in Table 2 represent rather few measurements. Overall, the concentrations from both Danish and international data are highly

Table 2. Overview of heavy metal concentration ranges in urban stormwater runoff from Danish and international studies. All concentrations are in µg L⁻¹.

Element	Danish studies ^a		International studies ^b		Birch et al. (2011) ^c Separate sewer
	Separate sewer	Road runoff	Roof runoff	Road runoff	
Cd	0,0 – 1,6	<0,1 – 1,5	0,07 – 1,3	0,67 – 63	0.3 (0.0–0.6)
Cr	0,0 – 27	7,6 – 56	2,0 – 6,0	5,0 – 50	14 (0.4–41)
Cu	0,0 – 140	18 – 720	4,8 – 3416	21 – 7031	71 (22–155)
Pb	3,8 – 210	<0,4 – 190	0,5 – 493	10 – 2028	32 (9.8–72)
Zn	0,0 – 790	47 – 700	24 – 48800	15 – 2000	166 (74–244)

^a Kjølholt et al. (1997), Danish EPA (2006), Lehmann et al. (2001), Jensen et al. (2011).

^b Duncan et al. (1999), Pitt et al. (2004), Göbel et al. (2007).

^c Mean value (minimum–maximum), n = 3-5.

variable which is also the case for runoff from seemingly similar urban typologies. The reasons for this variability are discussed later. The recent study by Birch et al. (2011) was mostly based on grab samples from 6 different storm sewers including several urban typologies such as road, residential and industrial areas. Depending on the metal the mean value is based on 3 to 5 sample concentrations. The concentrations from Birch et al. (2011) are within the ranges of the previous Danish as well as international data. The mean values are comparable to the median values from the international data gathered in **paper I**, except for Cd and Zn concentrations which are generally lower. However, the amount of data and information is too scarce to explain this (positive) difference.

Xenobiotic organic compounds

Xenobiotics are defined as ‘chemicals found where they are not naturally produced or expected to be present, or chemicals that are present at unnaturally high concentrations’. There are thousands of xenobiotic organic compounds (XOC) in the urban water cycle (Donner et al., 2010), and many of these are also found in urban stormwater (Eriksson et al., 2005; Zgheib et al., 2011). Among the vast number of potentially occurring XOCs, the PAHs constitute the most ubiquitous group of pollutants in the urban environment and have therefore been suggested as priority and indicator pollutants in urban runoff (Eriksson et al., 2007; **paper I**). In addition, the water quality standards set out by the European Commission (2008/105/EC) may easily, for some of the PAH compounds, be 10 – 500 times lower than what is frequently observed in urban runoff, namely from trafficked areas (**paper I**; Birch et al., 2011; Zgheib et al., 2011). Although we have been aware of the potential presence of a vast number of XOCs in urban runoff, very few measurements have been made to improve our knowledge basis. There may be several reasons for that. First, analysis of most XOCs is rather difficult and expensive. Second, there has been little consensus on which XOCs to measure (except for PAH) – which were the crucial parameters? And third, general reluctance towards opening what could turn out to be *Pandora’s Box*. However, the European Commission’s appointment of priority pollutants (2455/2001/EC) and concomitant water quality standards (2008/105/EC) for surface waters has provided a framework for monitoring schemes, and subsequent to the submission of **paper I**, at least two studies have been published concerning occurrence and partitioning of priority pollutants in urban stormwater. Birch et al. (2011) collected grab samples from a number of storm sewers in the Copenhagen area and measured the total concentrations of 37 organic contaminants including those on the EU list of priority pollutants. Zgheib et al., (2011) collected flow proportional stormwater samples from 6 precipitation events at the storm sewer outlet from a densely populated Parisian suburb catchment (reduced area: ~1.5 km²) consisting of commercial and residential areas. They analysed for 80 organic compounds including all the priority pollutants and found that 39 out of the 80 compounds were not present in any of the six event mean samples while 34 were recurrent (Table 3). Seven compounds occurred only in some of the samples. All the 16 US EPA PAH compounds (**paper I**) were among the recurrent pollutants, as was tributyltin (TBT) and its derivatives, six polychlorinated biphenyls (PCB), nonylphenol, DEHP (Di(2-ethylhexyl)phthalate, as well as a few of the pesticides included in the

Table 3. Occurrence of the pollutants observed in stormwater (percentage, n = 6). From Zgheib et al. (2011).

Never		Rarely	Fairly	Often	Reccurent	
0		[0–25]	[25–50]	[50–75]	[75–100]	
Cadmium ^a	4-chloro-3-methylphenol	Pentachlorophenol ^b	Chromium	Isoproturon ^b	Tributyltin ^a	Benzo[g,h,i]perylene ^a
Mercury ^a	Pentabromodiphenylether ^a	4-(para)-nonylphenol ^b	Desethylatrazine	Glyphosate	Dibutyltin	Indeno[1,2,3-cd]pyrene ^a
Nickel ^b	Octabromodiphenylether	4-n-octylphenol			Monobutyltin [MBT]	PCB28
Platinum	Decabromodiphenylether	Metaldehyde			Lead ^b	PCB52
PCB194	Alachlor ^b				Copper	PCB101
Hexachlorobenzene ^a	Aldrin				Zinc	PCB118
Pentachlorobenzene ^a	Endrin				Naphthalene ^b	PCB138
1,2,4-trichlorobenzene ^b	Dieldrin				Acenaphthene	PCB153
1,2,3-trichlorobenzene ^b	DDT-2,4'				Acenaphthylene	PCB180
1,3,5-trichlorobenzene ^b	DDT-4,4'				Fluorene	methylene chloride ^b
Benzene ^b	Isodrin				Phenanthrene	Nonylphenols ^a
Ethylbenzene	Endosulfan alpha ^b				Anthracene ^b	para-tert-octylphenol ^b
Isopropylbenzene	Endosulfan beta ^b				Fluoranthene ^b	4-ter-butylphenol
Toluene	Lindane ^a				Pyrene	Diuron ^b
Xylenes (Sum o,m,p)	alpha Hexachlorocyclohexane ^a				Benzo[a]anthracene	Aminotriazole
1,2-dichloroethane ^b	Chlorfenvinphos ^b				Chrysene	AMPA
Hexachlorobutadiene ^a	Chlorpyrifos ^b				Benzo[a]pyrene ^a	Di(2-ethylhexyl)phthalate ^b
Chloroform ^b	Trifluralin ^b				Benzo [k]fluoranthene ^a	
Carbon tetrachloride ^b	Atrazine ^b				Benzo [b]fluoranthene ^a	
Tetrachloroethylene ^b	Desethylsimazine				Dibenzo[a,h]anthracene	
Trichloroethylene ^b	Simazine ^b					
C10–C13 chloroalkanes ^a						

a Priority hazardous substance.
b Priority substance.

 Table 4. Observed mean concentrations (min – max) of xenobiotic organic compounds (XOC) in storm sewer discharges in two newly published papers. EU water quality standards from 2008/105/EC. All concentrations in ng L⁻¹, except nonylphenol and DEHP (µg L⁻¹). Particle associated fraction was calculated by using the mean concentrations alone.

Compound	Birch et al. (2011) ^c		Zgheib et al. (2011) ^d		WQS	
	Total		Total	Dissolved		% part.
PAH						
Naphthalene	34 (<10 – 72)		131 (88 – 175)	66 (50 – 94)	49	2400
Phenanthrene	139 (17 – 290)		370 (90 – 712)	73 (25 – 110)	80	1300 ^a
Anthracene	44 (12 – 84)		50 (16 – 96)	<10	>80	300
Fluoranthene	252 (25 – 550)		425 (98 – 832)	16 (13 – 18)	96	100
Pyrene	220 (34 – 560)		575 (100–1223)	20 (15 – 20)	97	
Benzo[a]pyrene	91 (<10 – 310)		159 (41 – 315)	<10	>94	50
Benzo[b+k]-fluoranthene	287 (<10 – 100)		445 (110 – 876)	<10	>98	30
Benzo[g,h,i]-perylene	150 (<10 – 470)		279 (71 – 569)	<10	>96	2
PCB						
PCB ₂₈	n.d.		35 (32 – 39)	<30	~100	
PCB ₅₂	n.d.		27 (<30 – 39)	<30	~100	
PCB ₁₀₁	n.d.		29 (<30 – 43)	<30	~100	
PCB ₁₁₈	n.d.		29 (<30 – 43)	<30	~100	
PCB ₁₃₈	n.d.		41 (32 – 52)	<30	~100	
PCB ₁₅₃	n.d.		42 (33 – 52)	<30	~100	
PCB ₁₈₀	n.d.		37 (32 – 43)	<30	~100	
Pesticides						
Diuron	25 (<10 – 55)		513 (394 – 647)	498 (360 – 640)	3	200
Isoproturon	21 (<10 – 44)		43 (4 – 82)	33 (<50 – 600)	23	300
Glyphosate	589 (43 – 1200)		992 (50 – 1922)	983 (<50– 1900)	<1	
AMPA	181 (60 – 330)		571 (479 – 731)	455 (320 – 660)	20	
Misc.						
Nonylphenol (µg L ⁻¹)	0.26 (0.10 – 0.43)		5.0 (1.6 – 9.2)	0.94 (0.59 – 1.5)	81	0.3 ^b
Pentachlorophenol	n.d.		72 (<100– 286)	50 (<100 – 200)	31	400
DEHP (µg L ⁻¹)	10 (3 – 29)		30 (15 – 61)	3.4 (<5 – 8.3)	89	1.3 ^b
Tributyltin	<4		60 (50 – 78)	<50	>17	0.2

^a Danish WQS (water quality standard) (MIM, 2010), ^b Note unit: µg L⁻¹, ^c n = 3-5, ^d n = 6
n.d.: not determined

study. In addition to the concentration levels, Zgheib et al. (2011) also reported the partitioning between the liquid and the particulate phase for each compound. The results of the two newly published studies are presented in Table 4. In general, the total concentrations measured in the Copenhagen area are significantly lower than those observed in the Parisian suburb. There could be many reasons for that, i.e. different pollutant sources in the catchments, different sewer constructions, or different sampling techniques. Nonetheless, a continued conductance of such monitoring studies is crucial to establish a solid fundament for appointing the XOCs that are most relevant for urban stormwater. In time, this could lead to a significant reduction in the necessary pollutant parameters and concomitantly a reduction in the costs. For several of the included XOCs the concentrations observed in both studies require substantial treatment or dilution in the receiving surface water in order to comply with the quality standards, namely the high molecular weight PAHs and tributyltin. In the case of tributyltin the WQS is considerably lower than for the other pollutants. In fact, this value is about 2 orders of magnitude lower than what commonly used analytical techniques can manage (Lepom et al., 2009). The partitioning observed by Zgheib et al. (2011) supports the hypothesis put forward in **paper I**, that adequate removal of fine suspended solids and the PAH compounds phenanthrene, fluoroanthene and benzo[b+k]fluoranthene in a SUDS would indicate high treatment efficiency towards PCBs, nonylphenol, and DEHP as well. However, all the priority pesticides seem to be more dissolved than particulate, suggesting that other treatment processes than filtration and sedimentation should be employed in order to obtain high water quality. The partitioning observed for pentachlorophenol and tributyltin is encumbered with uncertainty as both the total and the dissolved concentrations were determined around or below the limit of quantification. The same could be the case for the PCB congeners, but Zgheib et al. (2011) specifically stated in the paper that the particular fraction constituted ~100% of the total concentration.

Nutrients, organic matter, and salts

The most important nutrients for surface water quality, phosphorus (P) and nitrogen (N), do not exhibit directly toxic effects. The primary adverse environmental impact resulting from excessive discharge of nutrients is eutrophication of surface waters leading to algal blooms and subsequent oxygen depletion. In Denmark, and possibly many other member states of the European Union, stormwater discharges are not considered to be of primary concern when it comes to nutrients. Undoubtedly, however, there are sensitive water bodies which may develop eutrophic conditions as a result of urban stormwater discharges (Carpenter et al. 1998), but in most cases the contributions from agriculture are targeted first, i.e. Jeppesen et al., (2007) and Egemose et al. (2009). In other parts of the world, i.e. the U.S. and Australia, nutrient discharges from urban stormwater are considered key to the protection of coastal waters from impairment, and great effort is often put into controlling the discharges. This topic is discussed more thoroughly in **paper I**.

Organic matter also contributes directly to the consumption of oxygen in receiving surface waters as microorganisms decompose the compounds in their effort to gain energy, but the concentrations of organic matter (measured as either biochemical or chemical oxygen demand; BOD, COD) in urban stormwater are low compared to effluents from wastewater treatment plants and combined sewer overflows. However, the presence of dissolved organic matter (DOM) in urban runoff may have a direct influence on the solubility of other pollutants such as heavy metals which are potentially kept in solution by complexation with dissolved organic structures (Herngren et al., 2005).

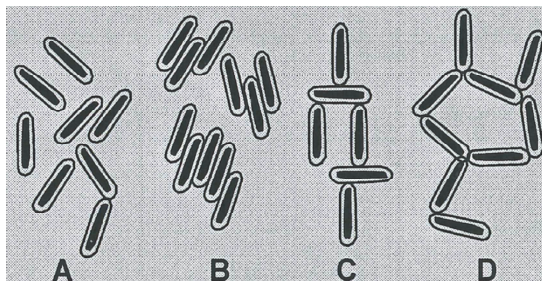


Fig. 4. Examples of clay particle arrangements. (A) dispersed, (B) plane-to-plane promoted, (C) edge-to-plane, (D), edge-to-edge. B is promoted by monovalent cations while C and D are promoted by di- and trivalent cations. From Borggaard and Elberling (2004).

Salts can cause problems in groundwater used for drinking water as well as in sensitive surface freshwaters. In an urban context, the major source of salts is the use of de-icing agents to improve traffic safety in cold climate countries like Denmark. The cheapest and most widely used road salt, sodium chloride (NaCl), has the potential to contaminate both groundwaters and surface waters as a result of intense road salting, i.e. Williams et al. (1999) and Godwin et al. (2003). Moreover, there can be direct effects on roadside soils and vegetation. The soils may be influenced by high concentrations of monovalent cations such as Na^+ which promote plane-to-plane arrangements of soil particles, i.e. clay particles (Fig. 4), resulting in a more compacted structure. Plants may suffer from poor water permeability in the soils as well as the osmotic gradient and toxic effects induced by the high concentrations of Na^+ and Cl^- in the soil pore water (Paludan-Müller et al., 2002; Ramakrishna and Viraraghavan, 2005). Furthermore, high concentrations of Cl^- increase the mobility of heavy metals such as Cd and Zn through complexation (Bäckström et al., 2004; Norrström, 2005; Linde et al., 2007), while saturating the soil with Na^+ may subsequently enhance the potential for dispersion of inorganic and organic colloids as low ionic strength runoff percolate the soil (Amrheln et al., 1992; Norrström, 2005; **paper IV**). Alternative products such as calcium magnesium acetate (CMA) and potassium formiate (PF) show less toxic effects on plants and may be beneficial for maintaining an open structure of the soil. However, the anions are organic corresponding bases which may also increase the mobility of heavy metals through complexation (Herngren et al., 2005), and while they are probably mineralised in the soil within a few days they may contribute to rapid oxygen consumption if discharged into nearby surface waters (Ramakrishna and Viraraghavan, 2005). The economical costs of using CMA and PF are considerably higher than using NaCl.

Pathogens

The risks from pathogens seldom pose a threat to aquatic ecosystems, but may restrict the use of urban runoff for recreational purposes where humans will be in direct physical contact with the

water. However, few risk assessments exist which consider different scenarios of human contact with urban runoff. Recently a Danish report concerning the health aspects of using urban stormwater for recreational purposes was published by the Danish Nature Agency (Clauson-Kaas et al., 2011). Obviously the degree of human contact is important for the risk of catching water borne pathogens, but the origin of the runoff (i.e. from roofs, squares, and roads) also plays a vital role for the content of pathogens. It was concluded that the major risk concerning pathogens is imposed by other humans in contact with the water (i.e. children) rather than the nature of the runoff itself. It is suggested that roof and square runoff generally can be used for recreational purposes unless the surface serves as resting place for large amounts of birds or pets. In terms of risks from pathogens, road runoff could also be utilised, but would in most cases need to be cleaned for aesthetic and maintenance reasons. Further information about the behaviour and removal potential of pathogens in urban stormwater can be seen in **paper I**.

Pollutant profile variation

Taking a look at Table 2 and Table 4 in this thesis as well as the median concentrations reported in Table 3 in **paper I**, it is clear that the pollutant concentrations observed in urban stormwater runoff are highly variable. Some of these variations can be explained by differences in land use, but even among very similar urban typologies, i.e. roads of similar traffic intensity, the concentrations can vary up to several orders of magnitude. Obviously, as some pollutants originate from a few specific sources while others may come from multiple sources, the relative variations are not equal for all pollutants. Overall, the pollutant profile of urban runoff is a product of two processes: Pollutant build-up and subsequent wash-off. But each of these processes is subject to significant variations due to a number of influencing factors (Table 5). Of the factors mentioned in Table 5, land use, dry period, traffic intensity, rain depth, and intensity are considered to be the most significant ones (Kayhanian et al., 2007). However, none of these factors have been observed to singlehandedly describe the pollutant concentrations in urban runoff (Charbenau and Barret, 1998; Kayhanian et al., 2003; Maestre and Pitt, 2005; Francey et al., 2010). Rather, all the mentioned factors potentially exert their influence simultaneously yielding a complex system of interactions that is difficult to navigate (Zoppou et al., 2001).

Land use

In this section specific attention is paid to land use, as it represents a convenient parameter on which to base predictions and planning regarding the quality of urban runoff. There are many approaches to the classification of *land use* types, and often it depends on the purpose of the given project. Small scale studies or planning of individual SUDS may utilise very specific subdivisions of a catchment area, i.e. roofs and roads, while whole watershed projects often distinguish between land use types such as residential, commercial, and industrial areas that can then be further subdivided according to population density, activity, degree of impervious cover, or other parameters. Overall, it is the number and types of pollutant sources within land use types that determine the differences or similarities among them. Hence, land use is a wide concept and the classification in one

Table 5. Examples of factors possibly influencing the build-up and wash-off of pollutants on urban surfaces.

Factor	Description	Significance
Build-up		
Land use	Distinction between different surface types (i.e. roof, road) and urban typologies (i.e. residential, commercial, industrial, etc.)	Mostly determined by the number and types of pollutant sources within a land use type (Maestre and Pitt, 2005). Typically, high density areas increase the pollutant accumulation on impervious surfaces (Duncan, 1999).
Dry period	The period of dry days between rain events.	In theory, the longer the dry period, the more pollutants accumulate on urban surfaces. (Maestre and Pitt, 2005)
Traffic intensity	The number of vehicles on a road within a time period. Often used: Average Annual Daily Traffic (AADT)	Combustion, engine oil, and wear of car parts are significant sources of a range of pollutants, i.e. heavy metals and PAHs. Thus, high AADT results in higher pollutant accumulation (Kayhanian et al., 2003).
Diffuse atmospheric deposition	The diffuse contribution of pollutants from dry or wet (rain) deposition.	I.e. in industrial areas, near densely trafficked roads, or in residential areas with active wood stoves there may be a contribution of air-borne pollutants to urban surfaces (Nielsen et al., 2010).
Soil type	Distinction between different soil types, i.e. sandy, loamy, clayey, etc.	The soil type in the area can influence the amount and size distribution of solids that accumulate on impervious surfaces. i.e. clay particles are much smaller than sand. (Roger et al., 1998)
Wind	Wind and turbulence conditions.	Wind and turbulence can have a strong influence on the amount and size distribution of solids on impervious surfaces. Atmospheric deposition is also diminished by turbulent conditions.
Season	The time of year.	Examples of parameters with seasonal variation: the use of studded tires and de-icing agents, wood stoves, flows of organic matter, the use of pesticides, presence of bacteria, etc. (Hallberg et al., 2007; Tiefenthaler et al., 2008)
Wash-off		
Rain depth	Millimetres (mm) of rain in an event.	More rain leads to increased dilution and, thus lower concentrations. However, above a certain level, the rain depth has little to say for the total mass of pollutants washed off the surface (Maestre and Pitt, (2005).
Flow intensity	The amount of water per time unit, i.e. $L\ min^{-1}$, $mm\ hour^{-1}$, etc.	The intensity is very important for the wash-off. Higher intensities tend to mobilise more pollutants. Namely SS may be strongly influenced. Intensity also determines if a first flush occurs, and thus, follows the hydrograph (Roger et al., 1998; Sansalone et al., 1998).
Runoff coefficient	Defines the fraction of the rain volume that actually runs off the surface. A factor between 0 and 1.	The runoff coefficient affects the amounts and intensities of runoff flow and thereby the wash-off of SS and other pollutants. Examples of differences: rough vs. smooth asphalt, slope vs. no slope (Charbeneau and Barret, 1998).
Particle size distribution	The distribution of solids among different size fractions, i.e. $<63\ \mu m$, $63-100\ \mu m$, etc.	Affects the mobility of total solids and thereby the large amount of potentially associated pollutants, i.e. heavy metals, PAHs, etc. (Paper I).

Table 6. Distribution of sampled events and sites among 11 defined land use types in the National Stormwater Quality Database (NSQD) (Maestre and Pitt, 2005).

Land use	Number of Sites	Percentage	Number of Events	Percentage
Residential	111	31	1042	28
Mixed Residential	44	12	611	16
Commercial	51	14	526	14
Mixed Commercial	29	8.1	325	8.6
Industrial	54	15	568	15
Mixed Industrial	22	6.1	249	6.6
Institutional	1	0.3	18	0.5
Open Space	10	2.8	49	1.3
Mixed Open Space	13	3.6	168	4.5
Freeways	22	6.1	185	4.9
Mixed Freeways	3	0.8	26	0.7

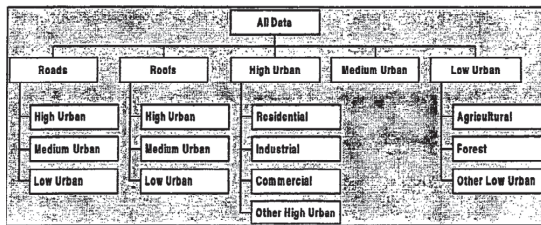


Fig. 5. Examples of surface and land use classification used in Duncan (1999).

catchment may not be readily transferred to other catchments. Duncan (1999) reviewed a large amount of international data on urban runoff quality reported in the literature. The review included the following water quality parameters: Total SS, total P, total N, COD, BOD, six heavy metals, coliforms, and streptococci. In order to perform a statistical analysis of the observations all included study sites had to be divided into different land use types. It was decided to group the studied sites according to surface use: roofs, roads, and other high, medium, and low urban catchments. It was further suggested that, regardless of the level of reported details, most of the study sites could be further subdivided into low, medium or high urban typology. Thus, each studied site

was allocated into the categories displayed in Fig. 5. The statistical overview revealed among other things, that (i) Roads are the major source of contaminants in urban areas, due to traffic as well as their lower elevation in the landscape, (ii) For all parameters, the concentrations from roofs are lower on average than concentrations from roads and all high urban typologies, except for Zn and Cu which appear in elevated concentrations in runoff from metal roofs and downpipes (Göbel et al., 2007), (iii) Residential typologies tend to have lower average concentrations of metals and organic matter, but higher concentrations of P and microbial parameters compared to other urban typologies, and (iv) Higher population density produce higher average runoff concentrations of total N, BOD, and fecal coliforms, but not for heavy metals.

The review and statistical overview by Duncan (1999) is among the most comprehensive compilations available, at least in an international context. In the United States, several large regional databases have been compiled. The most comprehensive one, The National Stormwater Quality Database (NSQD), is compiled of data collected during ten years from 1992 to 2002 and consists of 3765 rain events and 360 monitoring sites divided into 11 different land use types (Table 6) (Maestre and Pitt, 2005). Based on thorough data analysis it was suggested that land use type in combination with the level of impervious cover would reduce the variability of the concentrations observed, rather than when only one of these factors was considered. More information from the NSQD can be extracted in the future as new data is entered that comply with updated guidelines for the monitoring procedures.

On a more regional scale, Francey et al. (2009) monitored urban runoff concentrations in separate sewers from seven catchments in South Eastern Australia from 2003 to 2005. In spite of different urban typologies it was not possible to distinguish an impact of land use on observed average pollutant concentrations. Specifically for roads, Moores et al. (2009) carried out a

measuring and modelling programme at four sites in New Zealand and came to the conclusion that roads could be classified in order to the traffic flow. Thus, on road sections with frequent congestion, or at intersections where vehicles brake and accelerate, the concentrations of suspended solids, Cu, and Zn were significantly higher than on roads where traffic flowed freely.

In summary, it may be concluded that while there is no doubt that trafficked surfaces on average accumulate higher loads and runoff concentrations of most pollutants compared to roof surfaces, significant variations and deviations from this assumption prevail, and the results from one road may often not be transferable to another road. And although land use on a bigger scale, i.e. urban typologies such as ‘residential’ and ‘industrial’, seems to conveniently distinguish the pollutant profile from one catchment from that of another, there is simply not enough consistent data to support this hypothesis definitively. However, based on measurements of catchment storm sewers it seems that the degree of impervious cover is the single best estimator for the general pollution level.

Case studies

During the course of the Ph.D. project two case studies were carried out within the context of the 2BG research project in collaboration with fellow Ph.D. students, supervisors and representatives from the relevant municipalities and water companies. Both case studies were carried out for Danish urban catchments in Odense and Copenhagen, respectively, and approached in a transdisciplinary forum where hydrological aspects, water quality, and socio-cultural values were considered in the planning process. Both studies were strictly desktop studies, and besides producing ideas and suggestions for SUDS retrofits, the aim was to learn from the planning processes that evolved. They are included here as examples of runoff pollutant profile predictions in real life case areas. Here follows a short description of the progress and results with main focus on the water quality aspects.

Case study: Odense

This case study was carried out in the initial phase of the Ph.D. project and ended in March 2008. The case study cropped up from the fact that the city of Odense was experiencing problems with sewer surcharge from combined sewer systems. In fact, 30 out of 59 sub-catchments served by the combined sewer system had capacity problems. Two sub-catchments in need of improved urban drainage were selected for the drafting of solutions. The approach was based on the hypothesis that the hydrological aspects defined which areas should and could be targeted, while water quality and socio-cultural aspects had to be considered subsequently. Nevertheless, three groups were

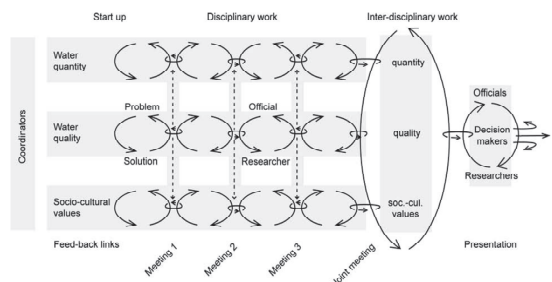


Fig. 6. Structure of the progress in the Odense case study. Three disciplinary groups worked more or less independently and met at a joint meeting to agree on the final approach to SUDS retrofits in the sub-catchments. From **paper II**

formed according to the three disciplines and the task for each group was to make a disciplinary report including recommendations for the sub-catchments in question (Fig. 6). Each group consisted of local officials as well as researchers. Finally, at a joint meeting the three groups agreed on an approach to SUDS retrofits in the two sub-catchments. A surprising overall outcome of the process was the identification of a possible demand driven view on SUDS retrofits rather than supply driven as was initially hypothesised. Thus, it was recommended that screening for strategic development goals that could be actively supported by the implementation of SUDS should be done at an early stage in the planning process. A detailed description and analysis of the planning process and the

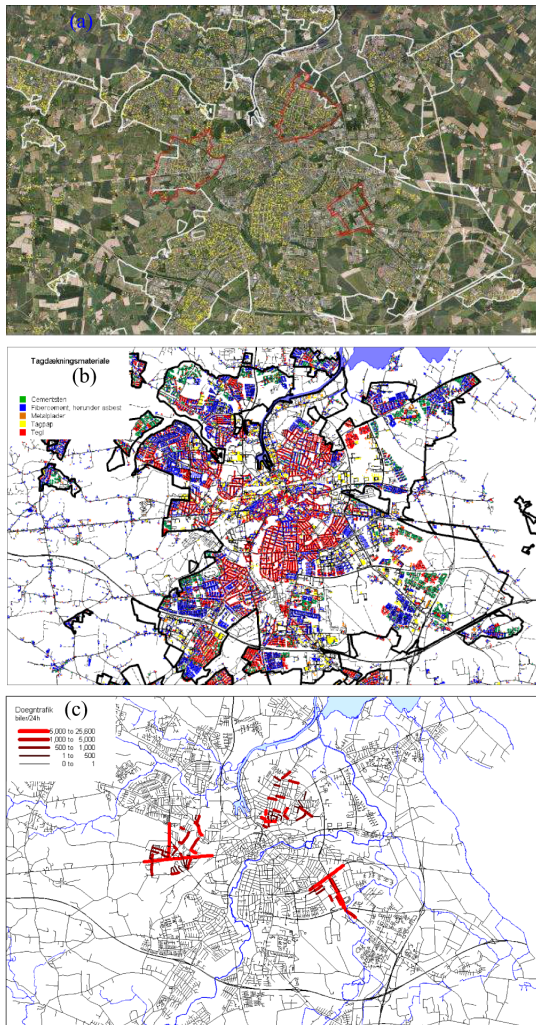


Fig. 7. Examples of GIS maps utilised in the discussion of expected runoff quality from different surfaces and urban typologies in the sub-catchments. (a) red lines outline the sub-catchments while the yellow dots signify buildings containing wood stoves, (b) five different roof materials, and (c) average daily traffic on selected roads.

outcomes as well as the results of an impact survey performed by one of the participating Ph.D. students is found in **paper II**.

Regarding the water quality disciplinary group the initial hypothesis was that the catchments could be sub-divided into urban typologies representing different qualities of the stormwater runoff and thus, different treatment needs. This was approached through a review of literature and Geographical Information System (GIS) analysis. Due to a constricted time frame the literature review was limited to Danish publications and a few international studies. Nonetheless, many possible sources of contaminants in the catchments were discussed, i.e. traffic, roof materials, and wood stoves (Fig. 7). Another issue which was thoroughly addressed was the possible impact on groundwater quality resulting from massive infiltration of slightly polluted roof runoff in a residential single-house area. With the prevailing uncertainties at that time concerning Danish implementation of EU groundwater regulations there was reluctance towards recommending massive infiltration of untreated urban stormwater. Thus, after much discussion in the group a conclusion was reached: Even though significant differences could be found between the character and land use of the present urban surfaces and typologies, it was impossible to identify

categories that could be considered to produce unpolluted runoff. Hence, all runoff in the sub-catchments was to be considered as polluted to some degree, and due to the risk of groundwater pollution it was recommended that all runoff was subjected to some degree of controlled treatment. In areas with a low degree of impervious cover, i.e. below 33% such as proposed by Duncan (1999) (Fig. 5), it was recommended that all runoff was infiltrated through an active top soil layer, preferably a filter soil such as those described in **paper III** and **IV**. However, it should be noted that at the time of this case study, filter soil was yet an undefined treatment option, but basic knowledge and previous studies within soil science implied that such soil layers often have excellent treatment abilities as well as provide easy access to the active treatment unit in comparison with underground soakaways or infiltration trenches. Furthermore, it was recommended that in areas with a high degree of impervious cover, infiltration through filter soil should be in combination with retention on or below ground. In cases of runoff from highly trafficked areas, a suitable and well documented end-of-pipe solution should be employed.

The case study was finalised in March 2008 and the process and outcomes were described in two summary reports and subsequently presented to a large group of decision makers and high-ranking officials at a concluding meeting. The findings and the experience that was gained was planned to be utilised in a later case study on a different catchment situated in the Copenhagen area as described below. The discussions and conclusions from the disciplinary work on water quality aspects reflect very well the previous conclusions concerning the variations as well as the inconsistent significance of land use and urban typologies.

Case study: Harrestrup Å, Copenhagen

The second case study was carried out from summer to December 2009. The selected case area is a relatively large catchment presently drained via a combined sewer system. The area is situated in the western part of Copenhagen, covering approximately 17% of the total area under Copenhagen Municipality (Fig. 8). A minor stream, Harrestrup Å, drains a large part of the watershed, but as most runoff is led into the sewer system, the flow is often low, namely during summer. When intense rain events occur, several overflow structures situated along the stream discharge a mixture of untreated waste- and stormwater, significantly impairing the biological and chemical quality of the stream. The stream flows into an inner salt water body, Kalvebod Brygge, which is planned to be a new recreational beach area in the southern part of the Copenhagen Area. However, this plan cannot be realised unless the number of combined sewer overflows to Harrestrup Å is significantly reduced. Solving this problem in the conventional way with large retention basins and extensions of the sewer network will be a very costly affair, and thus, both the municipality and Copenhagen Energy (the water company responsible for the sewers of Copenhagen) are highly interested in exploring the options for less costly and more sustainable solutions. It was estimated through computer simulations that around 60% of the impervious area should be disconnected from the sewer system in order to obtain the desired water quality. Therefore, the aim of this case study was to suggest a theoretical drainage strategy for the Harrestrup Å catchment based on SUDS while taking into consideration other possible synergies with municipality plans in order to optimise the

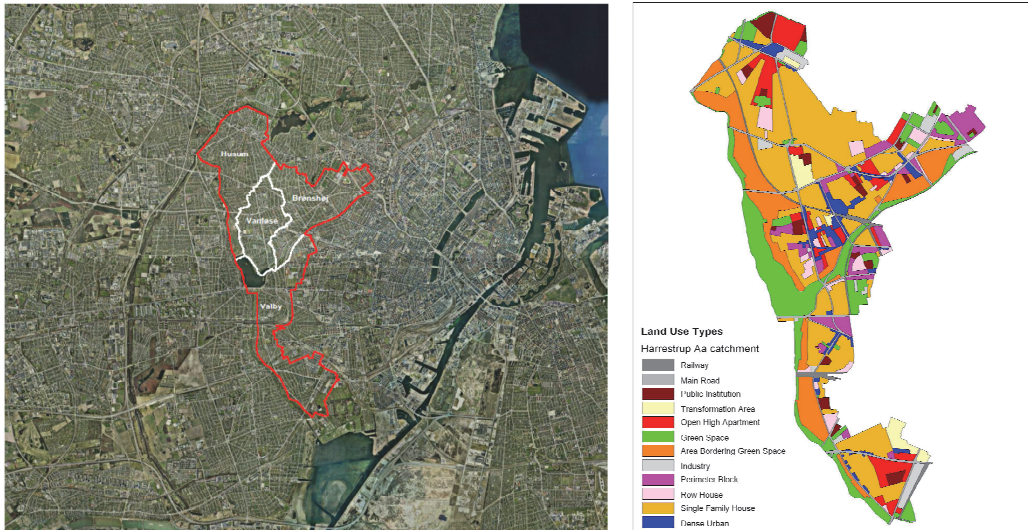


Fig. 8. Left: Red line outlines the Harrestrup Å case area, while the white lines are borders between city parts. Right: The case area divided into 12 urban land use types.

societal value. The proposed strategy was meant to serve as help and inspiration for the Copenhagen Municipality and Copenhagen Energy in their efforts to achieve their goals.

Thus, based on analyses of geography, land use types, synergy possibilities, hydrological conditions, and pollution aspects, three geographically distinguished strategies were proposed. Due to hydrological restraints only limited increased infiltration could be recommended; hence, solutions based on evaporation, retention, and transport to receiving surface water bodies were preferred. Three dominating areal types were suggested as target for stormwater-sewer disconnections, each representing an individual strategy: (i) Promotion of controlled infiltration, (ii) Promotion of blue/green park roads, and (iii) Promotion of stormwater transport to receiving water. Obviously these areal types had floating boundaries and at many locations a combination of the strategies are probably necessary. All the analyses and recommendations of the case study were described in a booklet which was handed out to all interested parties. Upon finalisation, the case study served as the basis for a national theme day where all aspects of the process were presented by the participating Ph.D. students.

Regarding the water quality aspects of the case study, it was the aim to estimate the needs for treatment of the runoff in the suggested areal types as well as suggest suitable treatment options. The catchment primarily consisted of residential areas, but with a number of larger approach roads and railways cutting through. 22% of the impervious area was dense urban, while 20% were roads and railways. The remaining fraction consisted primarily of single-family houses, apartment buildings, commerce, row houses, and public institutions. In the lack of concrete stormwater quality measurements from the case area we again had to rely on estimates based on national and international literature. In addition, a database compiled in the EU funded project Source Control Options for Reducing Emissions of Priority Pollutants (ScorePP) was used to identify the relevant sources and emissions of EU priority pollutants to road runoff (ScorePP,

2009). However, since emissions in this database are given as pollutant mass km^{-1} (Lützhøft et al., 2009), estimates of road length, annual average traffic intensity as well as annual precipitation in the catchment were needed in order to produce an average runoff concentration. Thus, the estimates based on the ScorePP database were considered to be of equal uncertainty as those based on literature reviews. Although the catchment was sub-divided into different land use types (Fig. 8), the experience from the Odense case study indicated that a distinction between runoff types from the different urban typologies was not possible. The estimates of runoff concentrations for a wide range of pollutants based on national as well as international measurements indicated that in the worst cases the required treatment needs could be difficult to meet for some pollutants in runoff from trafficked areas, namely Cu and the high molecular weight PAH compounds such as benzo(ghi)perylene. This is mainly associated with extremely stringent water quality criteria for surface waters. Metal roofs and roof surfaces treated with biocides could also be problematic in terms of meeting surface water quality criteria, but in such regards it is possible for the municipality to encourage private households and public institutions not to use these materials, or choose not to disconnect that particular property from the sewer. The overall conclusion regarding water quality control in the catchment was that many pollutants could potentially occur in the urban runoff, but at concentration levels that could be highly variable on the spatial as well as the temporal scale. The needs for treatment were highest in relation to runoff from trafficked areas, metal roofs, and biocide impregnated roof materials. In the worst cases it was not considered possible to treat the runoff sufficiently unless more technical end-of-pipe solutions were employed. Thus, the conclusions reached in this case study did not change the existing knowledge about the highly variable pollutant profile of urban runoff and the fact that most urban runoff needs some level of treatment before it can be discharged to the aquatic environment.

Significance of sampling method and frequency

Besides the factors already mentioned to influence the pollutant profile of urban runoff (Table 5) it is evident that thorough consideration should be given to the design of monitoring programmes if the dynamic variations are to be represented in the final monitoring results. In fact, sampling procedures, -location, -equipment, and -frequency may influence the validity and comparability of analysis results to such a degree that the imposed error overrules those imposed by other factors (Maestre and Pitt, 2005; Skarzynska et al., 2007). Thus, suitable guidelines and planning of sampling procedures are a must if realistic representations of pollutant profiles are to be obtained.

Depending on the circumstances, runoff samples can be collected manually or by means of an automated sampling system. Manual sampling is preferred if the needed amount of samples is small, if limited resources are available, or if the pollutants being sampled for require special handling. However, in most cases automatic sampling is preferred, not only for convenience reasons, but also because it tends to enhance the reproducibility and ability to produce composite samples. In case of large amounts of samples an investment in automatic sampling may prove to be cheaper than personnel. Furthermore, automatic sampling enables frequent capture of the initial runoff, which may be missed in manual sampling protocols due to delayed response. Based on

statistical analysis of data in the National Stormwater Quality Database (NSQD), Maestre and Pitt (2005) found that the total concentrations of SS, COD, Cu, Pb, Zn and P were statistically lower when manual sampling was performed compared to automatic sampling. This was ascribed to the fact that manual sampling more frequently misses the potential first flush effect due to delays in arriving at the site to initiate sampling. While there was a difference in total concentrations, this was much less evident for dissolved concentrations suggesting that the difference is primarily linked to the capture of SS. However, as the occurrence of a first flush should exert the same effect on dissolved as particulate pollutant concentrations, this is not in line with the assumption that missing the first flush is the explanation for the lower total concentrations. The overall occurrence of first flush has also been analysed using the data in NSQD. In this context a first flush is determined by comparing a grab sample collected within the first 30 minutes of a runoff event with the composited EMC. It was found that a first flush effect was not present for all land use categories nor was it present for all quality parameters. Even in the areas with the highest degree of impervious cover the data indicated first flush in less than 50% of the sampled events. Thus, it is likely that missing the first flush is not the sole explanation for the observed differences between manual and automatically collected samples.

Runoff samples are usually collected as grab samples during a rain event. These samples can then be combined to a flow- or time weighted composite sample representing the event mean concentration (EMC) for a given pollutant. Automatic samplers are usually programmable to perform the desired compositing of samples. In general, flow-weighted composite samples provides a better representation of the EMC than time-weighted (Leecaster et al., 2002; Maestre and Pitt, 2005; Ma et al., 2009; Ackermann et al., 2011). However, due to the high variations in flow and pollutant content a substantial number of samples are needed to capture this variation. In an attempt to evaluate different sampling designs Leecaster et al. (2002) collected stormwater discharge samples for an entire water year (1997-1998) in Santa Ana, California. Samples were collected every 15 minutes resulting in over 1700 samples which were used to calculate the 'true load'. They found that 12 in-storm flow weighted samples yielded significantly better results than 4 or 8 in-storm samples. Furthermore, it was found that sampling seven storms per year provided 95% confidence intervals that were approximately half as wide as sampling three storms with respect to yearly mass emission of total SS. Based on a large suite of grab samples of highway runoff from 35 rain events and statistical simulation of various sampling strategies, Ma et al. (2009) came to the conclusion that 30 volume weighted samples were required to estimate the EMC of COD within 20% average error.

In order to establish a representative annual average EMC it is important to consider the number of storms to sample such as illustrated by Leecaster et al. (2002). A good example of this was provided by Maestre and Pitt (2005) who performed a statistical analysis on the monitoring results from the most well sampled site in the NSQD. At this site 28 rain events were sampled during two years (1998-1999). In order to visualise the variation of possible annual average EMCs when sampling only three events per year a statistical test was performed in which 6 storm events were selected (three for each year). This corresponded to 5600 possible combinations of sampled

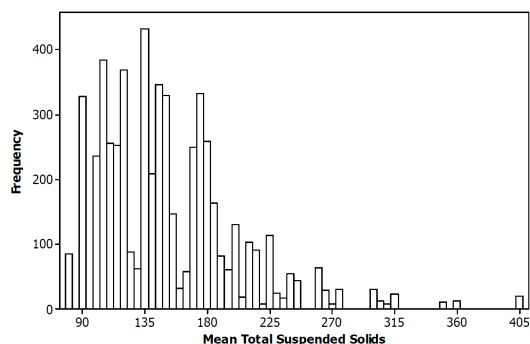


Fig. 9. Histogram showing the distribution of possible average EMCs of total SS based on randomly selecting six out of 28 sampled storm events (Maestre and Pitt, 2005).

rain events. The result of this analysis displayed as the distribution of possible annual average EMC of total SS can be seen in the histogram in Fig. 9. The ‘true’ median of the 28 sampled events was 170 mg L^{-1} with a 95% confidence interval ranging from 119 to 232 mg L^{-1} . As indicated in Fig. 9, a substantial number (~40%) of the possible combinations were outside the 95% confidence interval. Thus, with only three events sampled per year, the accuracy of the calculated average EMC is questionable until many years have passed.

According to the study by Leecaster et al. (2002) five years of sampling 3 events per year reduced the error to approximately 20%.

To sum up, it can be concluded that sampling strategies should be designed according to the purpose of the monitoring programme and desired accuracy of the results. The number of in-storm samples producing an EMC as well as the number of sampled storm events has a significant influence on the confidence interval of average EMCs. Less accurate EMCs may be sufficient information in some situations whereas others require higher accuracy in order to be useful, i.e. field testing treatment efficiency of SUDS or modelling urban stormwater quality.

Limited monitoring of urban stormwater runoff has been carried out in Denmark and if more national data is to be gathered it is crucial that more attention is paid to the sampling procedures. These aspects are not transparent for many practitioners and decision makers, and several small scale stormwater projects have been carried out based on a few grab samples from a few rain events. In the lack of time and understanding of potential errors induced by inaccurate sampling there is a tendency that uncertain numbers (i.e. concentrations) unconsciously become widely accepted and used in further research and decision making. This can be exemplified by probably the most cited Danish study of runoff from two urban typologies: A highway and a residential area (Kjølholt et al., 1997). In this study six storm events were sampled from each site using automatic collection of flow weighted composite samples ($n = 30$). However, in their description of the method it is stated that in several of the samples only half of the desired sample amount was observed in the sample container. It was speculated that this was caused by lack of water in the system as the pump was signalled to draw the acquired amount. Thus, it is further stated that due to these uncertainties considerable reservations should be made in terms of regarding the results as representative of event mean concentrations. However, this reservation is practically never mentioned in later publications referring to these results, and this report is very likely not the only case of highly uncertain results unconsciously transforming into generally accepted results. Representative sampling of urban runoff is a difficult task and many errors can be made, both by the experienced and inexperienced. Thus, in a Danish context it seems highly relevant to develop a set of guidelines for appropriate sampling of urban runoff. In order to enable decision makers and other

professionals to include potential uncertainties in their evaluations and planning of SUDS, there is a need for better transparency of existing as well as future monitoring results. One way of addressing this could be to develop a reliability classification system, i.e. based on sampling method as well as number and nature of events sampled.

Methods for predicting pollutant profiles

Different approaches have been taken in order to develop tools for predicting pollutant profiles of urban runoff. A short account of two approaches, namely development of surrogate parameters and stormwater quality modelling, is given here.

Surrogate parameters

Based on the hypothesis that the presence or concentration of some specific water quality parameters can serve as indicators of other parameters, a number of studies have investigated potentially predictive relationships between several parameters to appoint so-called surrogate parameters. The overall purpose is to minimise the parameters and costs of monitoring programmes.

Thomson et al. (1997) studied data compiled in a comprehensive stormwater quality database for Minnesota highway runoff containing more than 400 sampled storm events distributed among four sites. Based on analysis of inter-constituent correlations and subsequent multiple linear regression for EMCs from the most sampled site ($n = 211$), a number of regression based surrogate parameter relationships were developed. These equations were based on the four common water quality parameters, total SS, total dissolved solids, total organic carbon, and total volatile solids, which proved to be suitable for predicting the concentrations of a range of metals ($r^2 = 0.634 - 0.910$), ionic species ($r^2 = 0.501 - 0.996$), and nutrients ($r^2 = 0.237 - 0.836$). Subsequently, the near-site and far-site portability of these predictive models was investigated by applying the relationships to the other three sites of the Minnesota database as well as four selected inter-state highway sites. In general the near-site portability was good for ionic species, whereas the metal and nutrient relationships only proved valid at similar urban sites. The far-site portability was generally good for chloride, while it was only partially acceptable for metals.

Similar approaches have been taken in later studies using a Californian state wide highway runoff database (Kayhanian et al., 2007) as well as on site rainfall simulation (Miguntanna et al., 2010). Although rather good correlation relationships (Pearsson's correlation coeff. = 0.80 – 0.97) were observed among a range of surrogate pairs in the Californian study the portability was not further investigated. The relationships established by Miguntanna et al. (2010) showed acceptable portability for some and poor for others. A prerequisite for adequate portability of such relationships is that the relative importance of pollutant sources present in one catchment is more or less equal to that of other catchments. According to Duncan (1999) who provided a statistical overview of urban stormwater quality by analysis of a great number of international studies, correlations between water quality parameters over many measurement sites are often low. While in few cases, one quality parameter may provide a good estimate of another parameter, no single parameter provides a good estimate of a range of other parameters (Duncan 1999).

Stormwater quality modelling

Mathematical modelling of stormwater quality is an attractive approach as the costs of obtaining results in this manner are substantially lower than by the more conventional monitoring programmes. Although there are many different approaches and classifications of models for urban runoff quality, it seems reasonable to distinguish between two general approaches: (i) Regression models which rely on monitoring data to establish mathematical relationships (i.e. linear, multiple linear, exponential, etc.) between dependent variables such as water quality parameters and explanatory variables such as rain depth, degree of impervious cover, antecedent dry period, etc., and (ii) process-based models which attempt to simulate the pollutant build-up and wash-off processes based on conceptual ideas of how these processes work, typically involving linear, exponential, or power functions based on explanatory variables, i.e. antecedent dry period and runoff volume and -intensity.

Regression models are based on the same concepts as those described for the surrogate parameter relationships and face similar problems with portability to other catchments. However, once established and well calibrated for a certain catchment or region, they may be useful for decision makers in relation to future local planning.

Numerous different process-based models exist developed by academic institutions, regulatory authorities, government departments, and engineering consultancy companies. Build-up and wash-off on impervious surfaces are typically described by empirical exponential relationships based on simple first-order kinetics, but the processes are generally poorly understood and the relationships rarely reliable (Zoppou, 2001). This is backed up by Charbeneau and Barret (1998) who stated that antecedent dry period is not an appropriate parameter to base build-up models on; too many other parameters affect this process, i.e. those mentioned in Table 5. Furthermore, it was concluded that total SS did not follow simple wash-off models and that the use of average EMCs is an appropriate approach for generating information about long-term pollutant loadings. Egodawatta et al. (2007) used simulated rain fall to investigate the wash-off processes for total SS on urban road surfaces. They found that a certain runoff intensity was associated with a certain capacity to mobilise solids, but this relationship did not follow a mathematically predictive relationship. Consequently three categories of rainfall intensities were defined for which individual capacity coefficients could be used. In a later study the same authors (Egodawatta et al., 2009) investigated the build-up and wash-off of total SS from two different roof surface types. They found that maximum build-up capacity was smaller than for roads. This is in accordance with observations by Duncan (1999) who argued that due to the higher location of roofs they were subject to more wind-induced turbulence compared to roads which are often situated at the lowest altitude. They were able to describe both the build-up and wash-off processes using a power equation and an exponential equation, respectively. However, while the use of simulated rainfall is expedient for studying and understanding wash-off processes the mathematical descriptions may not be applicable for natural conditions where variables such as in-storm intensity dynamics and raindrop arrival rate may significantly influence the wash-off behaviour (Dunkerley, 2008). Recently, Dotto

et al. (2011) evaluated the performance and parameter sensitivity of complex process-based (MUSIC) and simple regression (KAREN) stormwater models based on a large dataset collected at five catchments of different land uses in Melbourne, Australia. Their results indicated that in spite of thorough calibration and the use of a parameter sensitivity approach, the water quality models had a high level of uncertainty and poorly represented the complex reality pollution build-up and wash-off. This is verified by the conclusions of Bertrand-Krajewski (2007) who stated that all models produce a partial, simplified, incomplete, and subjective representation of reality and that the ability, even of well calibrated and verified models, to predict pollutant profiles remains highly questionable. It is further argued that the main interest in stormwater quality models lies in their ability to answer operational questions such as design, conception, and management, and in order to improve the performance of such models there is a great need of more and reliable data to capture the natural variability observed within urban drainage.

Summary and recommendations

In order to address the challenges concerning prediction of pollutant profiles, available literature and data sets on urban stormwater quality from around the world was thoroughly investigated. It should be kept in mind that the term ‘pollutant profile’ refers not only to the presence of pollutants in terms of runoff concentrations, but also to their speciation and partitioning. The number of possible pollutants and factors that influence their concentration and behaviour in urban runoff is substantial making accurate prediction of individual pollutant profiles virtually impossible unless high resolution catchment specific monitoring programmes are carried out in advance. Thus, catchment specific stormwater quality models and parameter relationships may be useful when calibrated and applied correctly, but sensitivity and far-site portability is often poor.

In terms of monitoring, there is evidence that sampling methods and frequency are critical parameters in order to obtain representative pollutant concentrations and the imposed uncertainties are not transparent to decision makers and other professionals in this area of expertise. Future improvements could involve national guidelines for sampling of urban runoff as well as a classification system for existing and future monitoring results to indicate the degree of reliability.

Recognising the highly variable nature of pollutant profiles of urban runoff and the prevailing difficulties in predicting these variations, it seems reasonable to assume that, regardless of the amount of future research, strictly theoretical approaches will continuously yield a high degree of uncertainty. In order to determine whether theoretical or monitoring approaches should dominate the future planning of SUDS in Denmark a rational way forward could be for decision makers and researchers to agree on an acceptable level of uncertainty with regard to our knowledge about runoff pollutant profiles. The balance between this acceptable uncertainty and the costs associated with extended monitoring programmes will determine the overall preferences.

Challenge #2: Setting emission limit values

It is the purpose of this section to give an account of the laws and regulations that may influence the requirements to the quality of urban stormwater discharges, predominantly in relation to Danish and European legal framework. This subject was not part of the research objectives identified for the PhD project, but being a major driver for decision makers it was considered to be of high relevance to obtain a thorough understanding of the legal framework. There is generally little to be found in the national and international laws specifically addressing urban stormwater discharges. However, in order to determine whether or not a SUDS provides sufficient treatment, it is crucial to have well defined emission limit values. But on what background and assumptions can or should these limit values be defined? Thus, more specifically it is the aim of this section to identify the potential challenges connected with setting emission limit values for SUDS.

Water Framework Directive (WFD)

The WFD (2000/60/EC) issued by the European Union (EU) provides the legal framework within which member states should arrange their national legislation. The overall framing objective of the WFD is for all surface water bodies within the EU to achieve *good ecological and chemical status* and groundwater bodies to achieve *good chemical and quantitative status* by 2015 as well as avoid deterioration of water bodies that already fulfil these criteria. The good ecological status is defined as only a slight deviation from natural conditions with respect to hydromorphological, biological, and physicochemical properties, while the good chemical status is defined by the prevailing concentrations of priority substances described in the EU daughter directive on environmental quality standards in the field of water policy (2008/105/EC) or other substances of site specific relevance (Fig. 10). An important feature of the WFD, is the formulation of the *combined approach* (article 10) for point and diffuse sources which incorporates both the advances of source control options by application of best available technology (BAT) and water quality objectives for the receiving water body. Thus, the approach yielding the lowest emission limit value should be chosen, meaning that if the BAT is not sufficient to achieve the water quality objective then additional or improved ones are required. Furthermore, in cases of diffuse pollution, member states are obliged to implement the best available practice (structural or non-structural). Implementation of national legislation as well as development of region specific *programmes of measures* (PM) has been accomplished in most member states. In Denmark the first generation of these PMs for the 23 designated regions have been in hearing and are currently being finalised by regional authorities.

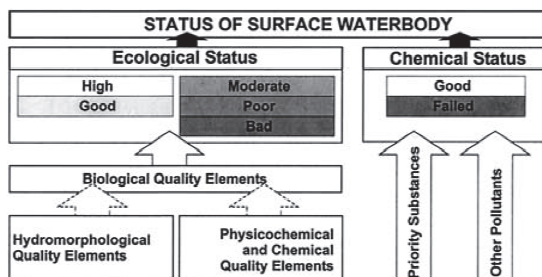


Fig. 10. Classification system for surface water bodies. From Achleitner et al. (2005).

The next milestone is for the municipalities to implement the regulations of the PMs in municipal action plans. Since water quality criteria as well as hydrological conditions for surface waters differ greatly from those of groundwaters the considerations regarding potential urban stormwater discharges are approached separately in the following sections.

Discharge to surface water

In Denmark it has been established that an extra effort is needed in 50% of the streams, 75% of the freshwater lakes, and 90% of the coastal waters in order to reach the objectives of the WFD. However, it seems that most surface water bodies have primarily been assessed with respect to eutrophication and potential oxygen depletion following discharges of wastewater while, still, little information is available about their chemical status in relation to EU priority substances (Danish Nature Agency, 2011a). In terms of pollution point discharges from urban areas the Danish PMs mainly distinguish between those from wastewater treatment plants and so-called *stormwater caused* discharges. However, stormwater caused discharges mainly refer to combined sewer overflows and separate sewer systems, while little consideration is given to potential increased implementation of SUDS. Thus, it is left to the local authorities, the municipalities, to implement prospective measures concerning emissions from SUDS in the municipal action plans. As a result of the combined approach and the fact that cost-effective BATs within stormwater treatment have not yet been properly benchmarked, emission limit values for SUDS will initially have to rely on assessments of individual receiving water bodies.

The EU daughter directive on environmental quality standards in the field of water policy (2008/105/EC) sets rigorous water quality standards for 33 priority substances, many of which are mentioned in Table 3 and 4. In Denmark this directive has been implemented in a departmental order (BEK. 1022, 25-08-2010) which also appoints water quality standards for a few additional substances, i.e. Cu and Zn. Like the EU daughter directive it includes water quality standards on an annual average basis (AA-EQS) as well as maximum allowable concentration (MAC-EQS) where applicable. Compliance with AA-EQS requires that for each representative monitoring point within a given water body, the arithmetic mean of the concentration measured at different times during the year is below the standard. Compliance with MAC-EQS means that the measured concentration at any time during the year must not exceed the standard. These standards will probably constitute the backbone of future decision making regarding emission limits for SUDS. However, considering some of the concentrations of pollutants observed, i.e. such as the examples described under ‘Challenge #1’ (Duncan 1999; Göbel et al., 2007; Birch et al., 2011, Zgheib et al., 2011, and the case study on Harrestrup Å), and the Danish and EU quality standards, there may be a great number of situations where sufficient treatment is virtually impossible. Thus, if national and international water quality standards are enforced to the letter they could seriously limit the implementation of SUDS in Danish cities.

In setting appropriate emission limit values according to the water quality of the receiving water body there are a number of potential uncertainties to consider which require thorough assessments, namely (i) the dilution factor, (ii) the extent of mixing zones, (iii) the current chemical water and sediment quality, and (iv) the presence of other similar sources of pollution. The two first issues essentially do not pose problems in that the knowledge and hydrodynamic models needed to predict both phenomena are at hand. Here the challenge may be restricted to implementing the right tools (models, maps, etc.) into the decision process. However, the two latter issues require substantial monitoring which may not be at hand or easily achieved. Currently, there

is lack of knowledge concerning the presence and behaviour of a range of the EU priority pollutants in urban runoff, and in combination with the fact that little is known about the current status of priority pollutants in most Danish surface waters and their sediments, this constitutes a major uncertainty in relation to setting emission limit values. In addition to these unknowns there is a lack of knowledge regarding the potential accumulation of pollutants in sediments as well as the interactive effects with the water column (Ellis and Mitchell, 2006). Although it is stated in the EU daughter directive (2008/105/EC) that monitoring of sediment and biota should be carried out in order to provide sufficient data for a reliable long-term trend analysis, it is difficult to estimate the time frame and the potential impact a negative trend could have on the setting of emission limit values. Essentially, this issue is partly covered by the overall objective about good ecological status since toxic sediments would have a negative impact on the biology of a given surface water body, but it seems that in the water policy field this issue has been neglected. Potential trends should be identified in a revised national monitoring programme (Danish Nature Agency 2011b).

The challenge concerning EU priority pollutants is further toughened by substantial analytical difficulties in relation to compliance monitoring. For some of the critical pollutants the sensitivity in standard analysis packages does not yet allow for valid assessment of compliance with water quality standards. This is namely the case for tributyltin (TBT) and some of the high molecular weight PAH compounds (Lepom et al., 2009), but could also be a problem in relation to the national water quality standards for some of the heavy metals, i.e. Cu and Pb with AA-EQS and MAC-EQS in the area of $<0.34 - 12 \mu\text{g L}^{-1}$. In a recent interview brought in the Magazine of the International Water Association (IWA), 'Water 21', the commercial director of the U.K. National Laboratory Service, Ian Rippin, states that "I think it is a really significant challenge to the environmental monitoring industry, which includes the Environment Agency itself" and "We know for a fact that across Europe, because we network through various forums, laboratories are struggling to respond to these very low levels of detection" (Hayward, 2011). Thus, it seems that the WFD is driving laboratories beyond the routine testing carried out to date. This may temporarily limit our knowledge expansion concerning the occurrence and fate of these substances until sufficient analytical capacity has been achieved.

Finally, it should be recognised that in many cases urban stormwater discharges cannot be regulated according to the polluter-pays principle, simply because the pollution originates from a number more or less well defined sources and no unambiguous responsibility can be placed. Thus, being the authority granting the discharge permissions in projects involving SUDS, municipalities may be left with a great responsibility in case of failure, possibly resulting in overly precautionary approaches or an increased tendency to continue with business-as-usual.

Discharge to groundwater

Groundwater bodies are also regulated through the objectives of good chemical and quantitative status under the WFD. Being one of the most widely used mechanisms within sustainable urban stormwater management infiltration may have significant influence on the quantity and quality of groundwater reservoirs below or in the vicinity of urban areas. Previously, groundwater was

regulated primarily according to drinking water interests, but with the implementation of the groundwater directive (2006/118/EC) under the WFD member states are forced to include potential deterioration (any significant and sustained upward pollution trend should be identified and reversed) as well as interactions with soil and surface water in their considerations regarding threshold limits (Quevauviller, 2008). A rather long time frame may be required in order to identify upward pollution trends, namely with respect to some of the EU priority pollutants which have not been part of the national monitoring programme before (Danish Nature Agency, 2011b). Only threshold limits for nitrates and pesticides are given by the European Commission along with a minimum list of water quality parameters that member states have to at least consider establishing threshold limits for. Recognising that for many pollutants there are significant differences between the human tolerance and that of small aquatic organisms, it is obvious that water quality standards are lower for surface waters than for drinking water, namely for some of the heavy metals where the difference may comprise several orders of magnitude.

Thus, two major challenges regarding the management of groundwater quality are overcoming the uncertainties related to (i) the identification of possible upward pollution trends with respect to EU priority pollutants (Jørgensen and Stockmarr, 2008; Visser et al., 2009) and (ii) the understanding of possible chemical interactions with soil, sediment and surface waters (Griseck et al., 2002; Hayashi and Rosenberry, 2002; Sophocleus, 2002). Although being addressed in the Danish Groundwater Monitoring Programme (Jørgensen and Stockmarr, 2008; Thorling et al., 2010; Danish Nature Agency, 2011b) it is likely that these issues will be a challenge for local authorities in their considerations concerning emission limit values for SUDS utilising infiltration and percolation to groundwater reservoirs. Additional uncertainties that may need consideration are the extent of dilution and the fate of pollutants in the vadose zone above the groundwater table. The latter may be highly variable depending on a number of factors, i.e. the depth to the groundwater table, redox conditions, soil texture and structure, and inherent physicochemical properties of the pollutants. A recent Danish report (Danish Nature Agency, 2010) provided a preliminary tool to assess the hazard potential of chemicals with respect to groundwater pollution. However, although discussing the potential importance of other transport mechanisms, this tool was solely based on the physicochemical properties (partition coefficient and biodegradability) of substances as well as basic assumptions about the adsorption capacity of sandy and clayey soils. Such simplified approaches are not representative of the potential contaminant mobility in many soils where preferential flow paths, complexation and colloid facilitated transport may be the dominating mechanisms (McCarthy and McKay, 2004; Degryse et al., 2009). Thus, in terms of urban stormwater infiltration it is reasonable to say that although considerable extra treatment is likely to take place in the vadose zone below infiltration facilities, it is difficult to quantify these processes and therefore the extent to which they should affect the emission limit value in the infiltration permission.

Summary and recommendations

It seems that the overall challenge faced by local authorities in terms of setting appropriate emission limit values is taking all the prevailing uncertainties into account without limiting the possibilities for future implementation of SUDS. If we are to improve our knowledge basis and experience with SUDS it seems essential that authorities dare granting realistic discharge permissions in spite of certain uncertainties. However, adequate monitoring and reporting of emissions should be required in order to identify and reverse potential non-compliance with emission limit values. Regardless of the approach and level of emission limit values, the overall compliance with EU goals will be difficult to evaluate during the first coming years until the revised national monitoring programme has been running for a number of years (Danish Nature Agency, 2011b).

Concerning emission limit values, it is suggested that for a yet undefined period of time (i.e. 5 to 10 years) authorities need to define case specific emission limit values for each SUDS employed. However, in this time period we should aim at gaining sufficient experience and documentation regarding the treatment efficiency of a range of SUDS to enable implementation of design criteria (best available technology) which ensures emission limit values in compliance with the given water quality standards. This aspect of design criteria is further discussed under ‘Challenge #3 – Selecting treatment options’.

Challenge #3: Selecting treatment options

The third major challenge deals with providing decision makers with the necessary knowledge and experience to facilitate the selection of proper treatment options within SUDS. Whether designed for it or not, the majority of SUDS inherently employ one or more treatment processes, i.e. sedimentation, filtration, adsorption, degradation, etc (Table 7). The challenge is to quantify and document the effects of these processes and make treatment performance comparable among the wide selection of SUDS. Thus, it is the purpose of this section to review and discuss the status and the challenges of selecting among SUDS and benchmarking their treatment performance.

Challenges of benchmarking

In this context *benchmarking* can be perceived as a procedure to compare SUDS using ‘treatment performance’ as the specific indicator. There are many indicators on which SUDS could be benchmarked, i.e. sustainability, hydraulic performance, or costs, but a great concern for decision makers is often whether the system can provide sufficient treatment to comply with the water quality criteria. However, benchmarking of treatment performance is not straight-forward as there are many factors that hamper the comparability. The most significant factors are mentioned here.

SUDS design and definitions

The wide spectrum of possible SUDS and inherent treatment processes (Table 7) is tantamount to regional design variations and different/overlapping definitions of SUDS. Many of the infiltration facilities mentioned in Table 7 are essentially different versions of the same facility, i.e. infiltration basins and swales, bioretention and rain gardens. However, different regional guidelines and

Table 7. Overview of SUDS and possible inherent treatment processes.

SUDS type	Primary treatment processes	Examples of facilities
Infiltration	Sedimentation Filtration Adsorption Degradation Plant uptake	Soakaway / trench Bioretention Infiltration basin Infiltration swale Rain garden Permeable paving/asphalt
Retention	Sedimentation Degradation Volatilisation Photolysis	Tank Forebay Dry pond Wet pond Wetland Technical basin
Filters/ Separators	Filtration Separation <ul style="list-style-type: none"> • Sedimentation • Flotation Adsorption Degradation	Oil separator Hydrodynamic separator Manufactured filter media Dual porosity filtration
Conveyance	Filtration Sedimentation	Grassed filter strip Grassed ditch Bioswale

traditions with respect to planting and soil media cause the systems and their potential treatment performance to be essentially different. Thus, in order to achieve a reliable and comparable measure of the treatment performance of a specific type of SUDS, the design guidelines and definitions should be scaled to national or international level, or more realistically, benchmarking should be specific to regions. As discussed below there are other reasons why this may be the most realistic scenario.

Water quality parameters

An inherent problem with the term ‘treatment performance’ is the subjectivity of the concept. What might be a good performance in one setting may be a poor performance in another, i.e. good removal of some parameters, but poor removal of others. This has to do with the inherent physicochemical properties of various pollutants (Scholes et al., 2008). Thus, it all depends on the target water quality parameters which ultimately are a result of catchment specific assessments of receiving water bodies. It could be argued that a benchmark for treatment performance should be based on a set of pre-defined parameters which ensure broad-spectrum testing of the treatment facility by covering a wide range of physicochemical properties (**paper I**).

Operation conditions

As emphasised under ‘Challenge #1’ the inflow pollutant profiles from different urban surfaces may be highly variable under field conditions and as the outflow concentrations from SUDS are related to inflow concentrations this may have a significant influence on the monitoring results. Examples of incorporating this into the comparison of facilities are provided later. Other site specific factors that may influence the performance of seemingly identical SUDS are climatic parameters such as temperature and wind, i.e. in facilities where microbial degradation or sedimentation are primary treatment processes, suggesting that sound comparison can only be made within regional or local areas of similar climatic conditions.

Sampling methods

As discussed for the sampling of urban runoff under ‘Challenge #1’ the choice of sampling equipment, -procedure and -frequency may significantly influence the results of the final analysis. Overall the same aspects apply for sampling of effluent from SUDS as for urban runoff, but there may be several situations where neither the inlet nor the outlet is well-defined. This is especially the case in infiltration facilities where special constructions or installations may be needed to enable collection of representative samples.

Data analysis

It is important for the outcome of a benchmarking process how the treatment performance is quantified, presented, and used. There has been a tendency to normalise output parameters in the form of removal percentage. However, as argued by Jones et al. (2008) there are a number of reasons why percent removal should be avoided as benchmark. Most importantly, removal percentage is primarily a function of influent quality in the sense that higher influent pollutant concentrations almost always result in higher relative pollutant removals. Thus, removal percentage may be more reflective of how “dirty” the influent water is and therefore says little about the actual effluent quality. Furthermore, methods for calculating percent removal are inconsistent (i.e. event by event, mean of event percent removals, inflow median to outflow median, inflow load to outflow load, etc.) and accordingly very different percent removals could be reported from the same data set.

SUDS selection tools

There are different approaches to facilitating the SUDS selection process. In most cases, the final choice depends on specific preferences or needs concerning hydraulic performance, environmental impact, sociological aspects, and economy. A few examples of approaches to selection tools are described below.

Multicriteria decision support

Martin et al. (2007) suggested a multicriteria decision aid approach allowing for ranking of solutions while taking into account different preferences and strategies of local decision makers. Eight of the SUDS mentioned in Table 7 were included in their study and evaluated with respect to eight different criteria: Pollution retention, probability of system failure, operation and maintenance, impact on groundwater, amenity level, contribution to sustainable development policies, capital costs, and maintenance costs. Based on knowledge obtained from a survey among users of SUDS as well as existing literature and previous experience, the evaluations resulted in eight scores for each SUDS evaluated, i.e. pollution retention was assigned a score on a scale from 1 to 5 based on expected pollutant removal percentage. The eight criteria could then be weighted according to local preferences or needs, i.e. need for minimising costs, resulting in different rankings of available SUDS. However, although this approach may be used or serve as inspiration for decision makers to prioritise among SUDS, it doesn't seem to emphasise situations where a certain level of treatment is needed. With reference to the previous section ("Challenges of benchmarking"), the assessment of pollution retention is simply too rigid.

A similar approach was taken by Young et al. (2009) who advocated for using the so-called Analytical Hierarchical Process (AHP) which is based on constructing SUDS specific comparison matrices for different site specific selection criteria. The comparison matrices are constant, but the number of relevant selection criteria may vary from application to application, and the weighting of each criterion may be adjusted to site specific preferences or needs. This approach is essentially not different from that of Martin et al. (2007). However, in their pollution retention assessment it was recognised that the performance of a given SUDS is greatly influenced by its design configuration, size, and influent pollutant profile. Thus, they computed the median effluent concentrations for total SS, total P, and total N from each type of SUDS using the information in three major databases on SUDS treatment performance. These values were used to subdivide SUDS into categories: "high", "moderately high", "moderate", and "low". It was stated that the pollutant removal performance should only be interpreted as a generalised relative comparison across types of SUDS.

In general, the power of such multicriteria decision tools lies in the normalisation of influential criteria which may differ with respect to units and quantification method. However, in terms of ensuring compliance with certain goals, i.e. emission limit values, they are not sufficient. In an attempt to facilitate a more detailed comparative assessment of treatment performance of SUDS, Scholes et al. (2008) developed a methodology for theoretical assessments. Their approach was to identify the governing fundamental *unit operating processes* responsible for the removal of

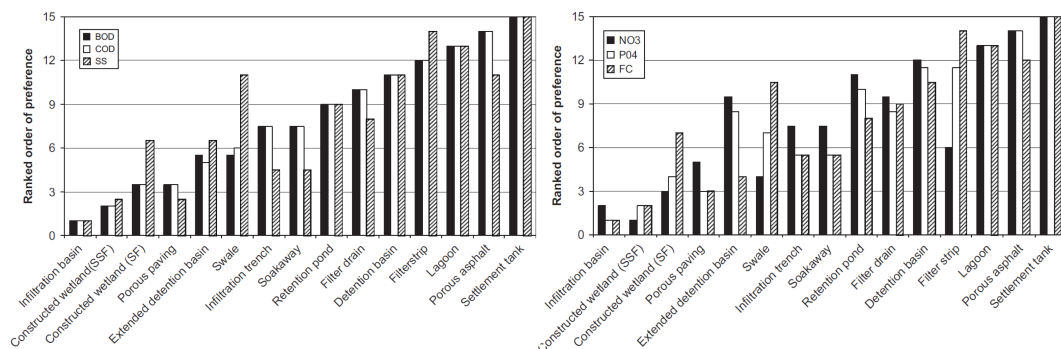


Fig. 11. Predicted order of preference for 15 types of SUDS based on their ability to remove BOD, COD, SS, nitrate, phosphate, and fecal coliforms (Scholes et al., 2008).

individual pollutants within SUDS. This was done by first ranking the relative importance (high, medium, low) of seven unit operating processes (adsorption, settling, degradation, filtration, plant uptake, volatilisation, and photolysis) to each type of SUDS. Next, the relative importance of these processes for the removal of individual pollutants was assessed and ranked. Combining these results yielded a predicted order of preference for the use of SUDS to remove individual pollutants (Fig. 11). Comparing the predicted order of preference from the theoretical approach for removal of SS with existing field observations showed fairly good agreement with only a few exceptions. However, the relative differences seen in Fig. 11 are not comparable to the results from field studies underlining the fact that the tool can be used for comparative assessments only. The authors imply that this tool may find its primary use in the interim period until more robust field data becomes available as well as feed into discussions and considerations regarding the selection of SUDS for urban stormwater treatment. Although potentially rather suitable, the approach by Scholes et al. (2008) has not been found implemented in any multicriteria decision tools such as those suggested by Martin et al. (2007) and Young et al. (2009).

Treatment performance database

The largest collection of results from monitoring programmes on SUDS treatment performance is found in the International Stormwater Best Management Practices (BMP) Database which is available online at www.bmpdatabase.org. Note that SUDS can be regarded as structural stormwater BMPs. This project began in 1996 and at present the database contains more than 400 individual BMP studies, but new results are continuously being added. The purpose of the database is to provide scientifically sound information to improve the design and performance of BMPs as well as facilitate the selection process for decision makers. On the web page there are guidelines for how to monitor and report results. Thus, the BMP database is probably the most powerful tool available for comparison of treatment performance among a variety of SUDS.

As mentioned previously the use of percent removal as a measure for treatment performance is a poor approach to inter-comparison of SUDS. In a comparison study of four types of SUDS (retention ponds, extended detention ponds, vegetated swales, and sand filters), Barret

(2008) utilised the at that time available data in the BMP database to present the event mean effluent concentrations in relation to the corresponding event mean influent concentrations (Fig. 12, lower row). Thus, on a pollutant-by-pollutant basis the treatment performance of the four types of SUDS could be clearly visualised. Although the amount of facilities included in the database was high, a number of facilities were excluded due to lack of information regarding design or operation conditions. Thus, the number of extended detention ponds and sand filters included in the assessment was rather low. Therefore an ellipse was drawn on the plots to indicate the likely performance of well-designed systems (fig. 12). Accordingly the slope of the ellipse indicates the dependence of the effluent concentrations on the influent quality, i.e. effluent TSS concentrations in swales seem to be very sensitive to influent concentrations while sand filters and retention ponds are less sensitive and consistently provides efficient treatment ($< \sim 40 \text{ mg L}^{-1}$) regardless of high influent concentrations.

In a more recent study Fassman (2011) used the data of the International BMP database to compare the expected effluent water quality from more conventional end-of-pipe SUDS (retention ponds, detention ponds, constructed wetlands, and media filters) with those of more recent types of SUDS such as grassed swales, bioretention, and permeable pavements. Only flow weighted composite event mean effluent concentrations were extracted from the database to create so-called effluent probability plots (Fig. 12, upper row). This plot is straightforward and directly provides a clear picture of the effluent water quality. According to the monitoring guidelines on the database web page (Geosyntec Consultants and Wright Water Engineers, 2009), curves of this type are the single most instructive piece of information that can result from a BMP evaluation study, and it is strongly recommended that the stormwater industry accept this approach as a standard “rating curve” for BMP evaluation studies. The examples shown in Fig.12 include all event mean

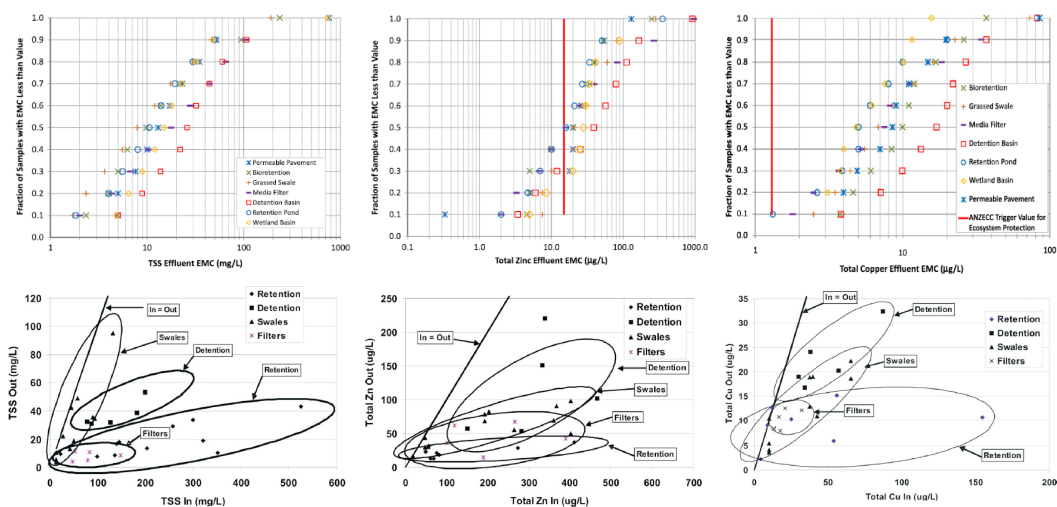


Fig. 12. Observations of SUDS treatment performance in the International Stormwater BMP Database presented in two different ways. Upper row: effluent probability plots for total SS, total zinc, and total copper (Fassmann, 2011). Lower row: plots of event mean effluent concentrations as a function of event mean influent concentrations for total SS, total zinc, and total copper (Barret, 2008).

effluent concentrations equally weighted across all sites. Another possible approach is to consider all sites equally by representing each site by one event mean effluent concentration. The choice of approach should be based on the amount of available data and the desired level of detail, i.e. of inter-site variability. Based on the information provided in probability plots, it is not transparent to what extent the performance of a system depends on the influent quality. This means that if a major fraction of the included events had low mean influent concentrations, the plots may indicate better treatment than what would be the case for more polluted water. Overall, the probability plots seem to provide very useful information for decision makers as they allow for assessing the frequency with which the discharge from a given type of SUDS may exceed guidelines. Thus, on this basis SUDS can also be selected according to a desired protection level, i.e. ninety percent of events produce discharge concentrations lower than a certain emission limit value.

Overall, the International Stormwater BMP Database provides much useful information for decision makers and designers of SUDS and will continue to do so in the coming decades as new data is entered. However, there are some limitations to its use. According to Barret (2008) a popular misconception has been that the database contains well-designed BMPs, when instead, the systems included in the database mostly were those whose monitoring programmes were well documented. Thus, there is substantial scatter in the available data which might be the result of less than optimum designs. However, it is expected that new data being added to the database will, to a greater extent, consist of well designed and -monitored systems which may improve the potential of the database for evaluating and improving design criteria of the included SUDS. The final outcome could be development of appropriate design criteria for a range of SUDS for which the treatment efficiency is so well documented that compliance with these criteria is sufficient documentation for attaining a certain emission value. In other words, the database could in the future lead to the appointment of well documented best available technologies (BAT) as mentioned under “Challenge #2”. In an international context, it is a prerequisite that SUDS are designed according to the same criteria as those systems included in the BMP Database in order for the information to be transferrable. Furthermore, the database is more or less restricted to rather few parameters, namely total SS, nutrients, metals, and pathogens. Thus, the database doesn’t provide new knowledge on the treatment efficiency towards xenobiotic organic compounds such as PAHs and pesticides. Finally, although this is not a limitation to the BMP database alone, it is not possible to tell the “best” SUDS from others as no single type of SUDS produce the lowest event mean effluent concentration across all of the parameters considered. This is due to the fact that high removal efficiency toward some pollutants may often compromise the removal efficiency towards others. A well defined minimised data set such as the one proposed in **paper I** could be used for comprehensive broad-spectrum testing and comparison of SUDS treatment efficiency. However, in order to construct a system for ranking SUDS according to this data set, a method would have to be developed which considered the relative importance of the included parameters.

The minimum data set

Among the initial ideas of the present Ph.D. project was the development of a method to benchmark the treatment efficiency of various SUDS. However, after reviewing monitoring programmes and data sets available in national and international literature, it became clear that overall lack of uniformity hampered the potential for performing a solid benchmarking. In addition to erratic approaches to SUDS design and sampling procedures among monitoring studies, inconsistent use of a wide variety of water quality parameters also limited the potential for comparison. Thus, it was considered to be highly relevant to improve the basis for accumulating more comparable data by advocating for uniform, yet small, monitoring data sets. Based on a thorough review of available data sets and on urban stormwater quality, including pollutant occurrence, partitioning, and environmental impact, a so-called *minimum data set* consisting of eight pollutant parameters was suggested to provide broad-spectrum testing and produce comparable data sets. The process leading to the minimum data set was iterative, and due to the nature of the available data the analysis and interpretation relied partly on induction and partly on deduction. A full description of the proposed minimum data set and the process can be read in **paper I**.

Summary and recommendations

Benchmarking the treatment performance of SUDS is not straight-forward due to the multiplicity of factors that has to be considered, i.e. design criteria, operation conditions, sampling procedures, and data analysis. Only few tools have been developed for decision support, but they all have drawbacks or limitations curtailing their utility in decision making. The International Stormwater BMP Database constitutes the largest available collection of SUDS treatment performance assessments which may provide useful information for decision makers. The database is available to the public, but it requires time and effort to analyse the data in relation to specific needs. Some studies have successfully utilised the database in order to make sound comparisons of SUDS, i.e. by making effluent probability plots. However, in order for the information to be transferrable to local or regional conditions it is a prerequisite that SUDS are designed according to the same criteria as those evaluated in the BMP database.

In order to move ahead in a national context, it seems relevant to first of all establish which SUDS require thorough documentation of treatment efficiency in the future, i.e. which SUDS could potentially become well documented national best available technologies (BAT) for treatment of polluted urban runoff? The next step would be to establish well considered design guidelines for these SUDS, and finally establish pilot facilities and monitoring programmes with standardised data sets as well as sampling and analytical procedures. It is recommended that the minimum data set proposed in **paper I** constitutes the backbone in considerations regarding standardised data sets. Obviously, this is an idealised procedure for establishing well documented national BATs which could easily require decades to be achieved. As a very first step, future work could involve a close look into the International Stormwater BMP Database to extract those SUDS and studies that are applicable in a Danish context. A potential future BAT is discussed in the following section, namely engineered filter soil used for infiltration and treatment of urban stormwater runoff.

Engineered filter soil: A promising treatment option?

In Denmark several municipalities and private road owners have shown interest in the potential for disconnecting areas such as roads and parking lots from the sewers. This is because such surfaces usually make up a significant proportion of the impervious urban areas, often between 30 and 50%, and thus, are responsible for much of the stormwater entering the sewer system. In order to protect groundwater and drinking water interests, Danish legislation as it is (MIM, 2007) does not encourage infiltration of runoff from trafficked areas serving more than 20 vehicles per day. However, municipalities may grant exemptions from this provision if infiltration is considered not to hinder the objectives of regional programmes of measures or municipal water action plans as described under “Challenge #2”. Thus, provided that polluted runoff can be adequately treated before percolation into the subsoil or groundwater, it should technically be possible to disconnect trafficked areas from the sewer system and manage the water locally. There are a number of technical and typically rather costly installations available for treatment of road runoff, i.e. a range of different separators, technical basins (Vollertsen et al., 2009), and dual porosity filtration (Jensen et al., 2011), whereas the number of low technology SUDS with similar abilities is scarce. However, in Germany there are several examples of stormwater infiltration facilities for road runoff constructed during the last 10 to 15 years, namely the so-called Mulden-Rigolen Systemen which typically consist of a vegetated infiltration swale or basin and an underlying trench (Fig. 13). In between the swale/basin and the trench a soil layer is placed which has to comply with certain standards. This soil layer will henceforth be referred to as *engineered filter soil*. The German standards for design and construction of the systems and the engineered filter soil (Table 8) are described by the German Association for Water, Wastewater, and Waste (Deutsche Vereinigung für

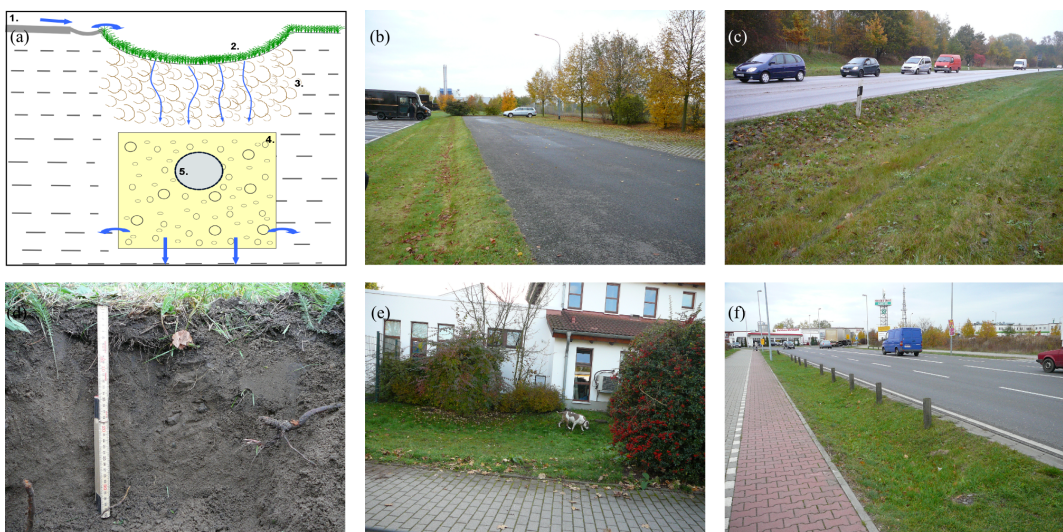


Fig. 13. (a) Sketch profile of typical German roadside infiltration swales, 1) road surface, 2) grassed swale, 3) engineered filter soil, 4) trench, 5) drain pipe. (b) Infiltration swale at a United Postal Service parking lot in Hoppegarten, Berlin. (c) Approach road (Glinderstrasse) on the outer skirts of Hamburg. (d) Profile of engineered filter soil. (e) Infiltration basin at KiTA daycare center in Dortmund. (f) Infiltration swale along an approach road (Gewerbestrasse) to an industrial area in Hoppegarten, Berlin.

Table 8. German standards for the design and construction of infiltration swales and basins (DWA, 2005).

<ul style="list-style-type: none"> • The depth of the soil layer should be 10 – 30 cm depending on the anticipated need for treatment and the treatment capacity of the soil.
<ul style="list-style-type: none"> • The mass of clay and silt should constitute no more than 10% of the total soil mass. In very sandy soils clay can be added to achieve better sorption capacity. Clay type should be secondary minerals, i.e. bentonite.
<ul style="list-style-type: none"> • Organic material (humus or compost) may be added to the soil to maintain adequate aggregate structure and enhance sorption capacity. Should constitute no more than 1-3% of the total soil mass.
<ul style="list-style-type: none"> • The soil pH should be between 6 and 8. If adjusted, it should be done with slowly soluble lime.
<ul style="list-style-type: none"> • The infiltration rate should not be lower than 10^{-5} m/s).
<ul style="list-style-type: none"> • The soil layer should be covered with suitable vegetation.

Wasserwirtschaft, Abwasser und Abfall) (DWA, 2005). While there are rather well defined standards for the design, there are no guidelines regarding the lifetime of filter soils or how to assess when they need replacement, except for recommendations that the upper soil layer should be regularly checked with respect to pollutant accumulation, approximately every 10 years (DWA, 2005). Furthermore, there are no official guidelines on how to monitor and document the treatment performance of the systems. Thus, solid documentation of their treatment performance is generally lacking, namely in the form of long term performance assessments. If Danish authorities are to implement infiltration facilities

with engineered filter soil to a wider extent in the future, it is crucial that better documentation of their treatment performance is provided. Therefore, after thorough consideration, a significant part of this Ph.D. project was devoted to investigating the status and treatment performance of existing German roadside infiltration swales (**paper III and IV**) with the aim of providing better documentation and possibly identifying potentials for improvement of the design of engineered filter soil. In order to facilitate these investigations contact was established to the German engineering company *Ingenieurgesellschaft Prof. Dr. Heiko Sieker mbH* which have managed the construction of several infiltration swales with engineered filter soil utilised for runoff from trafficked areas. Valuable help was provided by the employees, namely doctor of engineering Harald Sommer, in establishing contact to the relevant authorities and obtaining permissions for collecting samples in the relevant roadside infiltration systems. The selection of infiltration swales was based on the following criteria: i) construction expected to be in accordance with German guidelines, ii) operation age > 5 years, and iii) pollutant loadings expected to exceed background levels. Eight swales located in the Berlin area, Dortmund and Hamburg were selected, constructed

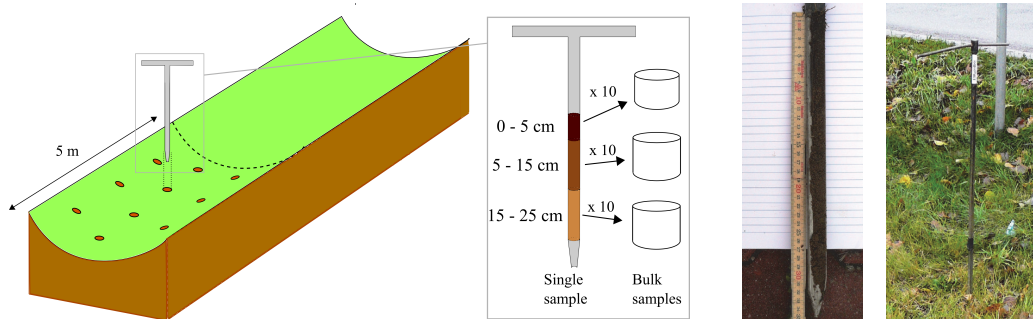


Fig. 14. Left: Sketch of the systematic sample collection in a 5 m areal segment of a swale. Ten soil cores were sampled from three 5 m areal segments in each swale (30 core samples all in all). Each core sample was sub-divided into three depth sections which were composited with the other nine samples from the same segment. Middle: Picture of core sample before being sub-divided. Right: Picture of the core sampler.

from 1994 to 2004. The investigations consisted of visual inspection of the swales, on-site measurements of the infiltration rate, collection of soil samples (Fig. 14) for basic soil characterisation and heavy metal + phosphorus content and distribution, as well as collection of intact soil columns (Fig. 15) for laboratory assessments of flow patterns and treatment efficiency towards a range of dissolved heavy metals and fine suspended solids. The investigations are described in **paper III** and **IV**, but additional information and discussion regarding the methodology of the studies is included here.



Fig. 15. Left: the stainless steel ring is placed on the soil monolith. Right: the soil column is gradually pushed downwards while making sure that no pebbles or roots get caught under the edge of the stainless steel ring.

Limitations

Although the studies on infiltration swales and their filter soil performed within this Ph.D. project comprised enough material for two papers, there are still plenty of questions to be answered regarding their lifetime and treatment performance. First of all, there is limited knowledge about the record of the systems with respect to initial soil composition and pollutant concentrations as well as the quality of the runoff that percolate the soils. Thus, it is difficult to utilise the data for explanatory statistical analysis, i.e. the significance of factors such as age and traffic intensity, as well as for calculations regarding the lifetime expectancy as shown in **paper III**. This is backed up by Achleitner et al. (2007) who conducted a similar study on infiltration swales along parking lots in Austria and concluded that the measured heavy metal concentrations in the soils were strongly influenced by initial background concentrations.

As mentioned earlier, there is a general lack of knowledge concerning some of the EU priority substances, i.e. xenobiotic organic compounds like PAHs, and their behaviour and treatment potential in SUDS. However, within the economical frames of this Ph.D. project the data set had to be limited to heavy metals, phosphorus and fluorescent microspheres. In future performance assessments it would be crucial to include other substances such as the PAHs proposed in **paper I**. Furthermore, the lack of an automated fraction collector for the soil column experiments drove the design of the column studies to fit to regular working hours with intermittent overnight drainage periods. Although subjected to similar leaching protocols, more continuous leaching experiments could improve the conditions for sound comparison of the performance of the soil columns.

Finally, it should be noted that the soil column studies may only poorly represent the variability that typically prevail under field conditions, i.e. as a function of macropores and uneven spatial distribution of runoff water in the swale. A better result for decision makers would be achieved if assessments were based on field studies in which the effluent under a larger areal

segment of the swale were collected and analysed following a number of naturally occurring rain events.

Significance of in-situ mobilisation

While several studies have shown that engineered filter soils, i.e. in bioretention facilities, may efficiently remove pollutants such as SS, heavy metals, and PAHs (**paper IV**), there are also indications that this efficiency is not consistent across all infiltration facilities and leaching sometimes may occur (Davis et al., 2006; Hsieh et al., 2007; Roy-Poirier et al., 2010; Trowsdale and Simcock, 2011). Although not documented in these studies it is likely that a major part of this leaching can be ascribed to in-situ mobilisation of organic as well as inorganic particles and colloids which act as carriers for a wide variety of pollutants and nutrients (**paper IV**). Thus, a question that often remains when assessing the treatment performance of such infiltration systems is how large a fraction of the eluted pollutants originate from the influent, and how much has been internally mobilised in the soil. Only long-term monitoring of the systems can provide valid mass balances that represent the overall treatment efficiency. Understanding the processes that control in-situ mobilisation of particles and colloids may help to interpret such assessments as well as to improve the overall treatment efficiency of stormwater infiltration soils. In the soil column experiments described in **paper IV** strong correlations were observed between several of the effluent heavy metal concentrations and dissolved organic carbon (DOC). The international scientific literature contains a wide range of studies on colloid and DOC facilitated pollutant transport in soils, primarily in agricultural soils, at contaminated sites, and land fill sites. However, the definitions of colloids and DOC as well as the interactions and conceptual overlaps between these parameters are not clear, i.e. a significant part of colloids may be comprised of DOC and vice versa.

The nature of colloids

Although particles in the size range above 10 μm may also be mobilised in natural soils, most research has been carried out for the colloidal size range, probably because colloids comprise fractions of clay, iron and aluminium oxides, and humic macromolecules with large specific surface areas and high affinity sorption sites. An account of the different perceptions of the colloidal fraction of soils was given by Nielsen (2010). There seems to be discrepancies concerning the definition of the upper size limit, some suggesting a particle diameter of 10 μm , others suggesting different diameters below 1 μm , while some base the upper limit on the settling abilities, i.e. DeNovio et al. (2004). The latter suggests that the same particle may be classified as a colloid under some conditions, but a soil particle under other conditions. This also implies that different methods are used to separate the colloidal fraction, i.e. filtration and centrifugation.

Soil colloids comprise a heterogeneous group and may consist of inorganic as well as organic constituents. Molecules of natural organic matter coat the surfaces of many colloids in natural systems, and thus play an important role for the mobility of organic as well as inorganic colloids (Baalousha et al., 2011). There are many similarities in the behaviour of inorganic and organic colloids, but there are also fundamental differences. Although potentially consisting of

stable humic macromolecules, organic colloids are to a much greater extent influenced by bacterial activity, namely under temperature and moisture conditions which favour decomposition of soil organic matter (Kalbitz et al., 2000; Neff and Asner, 2001). Furthermore, organic molecules and colloids of iron and aluminium oxides have variable charge hydroxyl surface or functional groups, while the charge of clay minerals arise from a combination of isomorphous substitution of cations into the structure of the minerals and variable charge surface groups (Borggaard og Elberling, 2004). The stability of macromolecules and organic colloids may also be influenced by steric arrangements and the affinity of organic functional groups to water (Stumm, 1992). Depending on the definition employed, organic colloids can largely be represented by the commonly measured parameter DOC, although it includes a wide and complex range of constituents from small dissolved molecules to macromolecules such as humic and fulvic acids to humus particles/colloids. Inorganic colloids are often quantified as the turbidity of a liquid sample.

Mechanisms of in-situ colloid and DOC mobilisation

Several physical and chemical factors may influence colloid mobilisation, i.e. soil texture, flow intensity, pore water composition, and soil moisture content. Increased initial effluent colloid concentrations from soil column experiments have been observed as a result of drying (Kjaergaard et al., 2004), long term sample storage (Fest et al., 2008) or even just a few days flow-interruption (Totsche et al., 2006). Figure 16 illustrates four mechanisms by which colloids could be mobilised. These mechanisms often act simultaneously in soils but their relative importance is not well established and may vary substantially between soils depending on the abovementioned factors (DeNovio et al., 2004; Laegdsmand et al., 2005). The dominating mechanisms in water filled pores seem to be dispersion (Fig. 17) as well as mobilisation resulting from shear stress (Fig. 16), while during soil wetting mobilisation can be a result of film expansion and air-water interface scouring in the pores. Another phenomenon which could also be regarded as a mobilisation mechanism is the diffusion from stagnant water in the immobile region of the soil matrix to the mobile regions, i.e. larger pores (Schelde et al., 2002).

Stormwater infiltration systems often utilise an engineered soil mix which seek to optimise matrix flow rather than macropore flow. Although macropore flow is difficult to avoid, especially given the observed biological activity (**paper III**), the dye tracer experiments (**paper IV**), as well as the fact that a significant part of the lower soil column will be saturated during high flow, a major mobilisation mechanism in most of these soils is likely to be dispersion of colloids resulting from perturbation of low ionic strength urban runoff. It should also be noted that such infiltration systems are subject to accelerated water inputs as they often infiltrate the runoff

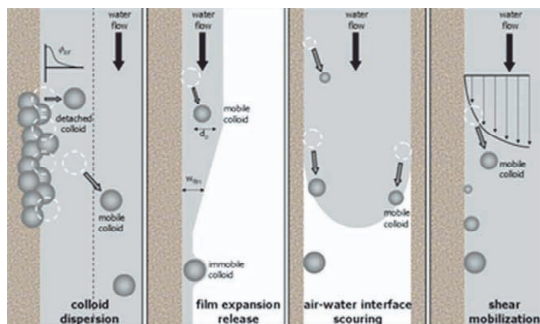


Fig. 16. Four mobilisation mechanisms potentially occurring in or along soil pores. From DeNovio et al. (2004)

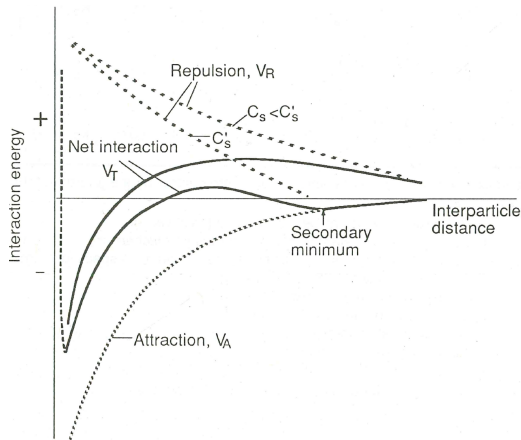


Figure 17. Conceptual model for the net surface potential of colloids (V_T). Repulsive energy (V_R) and attractive energy (V_A) decrease with increasing interparticle distance. V_R diminishes at longer interparticle distance as electrolyte concentration in the liquid (C_s) increases. From Stumm (1992).

attractive forces cause flocculation. The local physical and chemical conditions as well as the nature of the colloids determine which one of the forces predominate. In physical terms, the overall charge of the particle surface can be described as the difference between the repulsion forces induced by electrostatic interaction and attraction due to van der Waals interactions (Stumm, 1992). While both the repulsion and attraction energies depend on the interparticle distance, the repulsion energy is also affected by the electrolyte concentration (ionic strength) of the liquid. Thus, at very small interparticle distances the attraction energies predominate, whereas the repulsive forces often dominate at intermediate and long interparticle distances. However, high electrolyte concentrations compress the diffuse part of the double layer surrounding the particle potentially making the attractive energies superior at longer interparticle distances (Fig. 17).

Fine suspended solids and fluorescent microspheres

Of particular importance for the removal of many pollutants found in urban stormwater runoff are suspended solids, namely the fine fraction which regularly carry the largest relative pollutant load and is less susceptible to processes such as filtering and settling (**paper I**). There are examples of field studies on bioretention facilities where the effluent concentrations of total SS exceed those of the influent concentrations suggesting that in-situ mobilisation of particles or colloids take place in the soil, i.e. Davis (2007). Although significantly reduced compared to inflow concentrations, the effluent concentrations of total SS may vary from just a few to hundreds of mg L^{-1} (Li and Davis, 2008), but are often in the range of 10 to 20 mg L^{-1} (Davis et al., 2009). In order to identify potentials for improvements of infiltration facilities it is important to establish the origin of the effluent particles, i.e. fine suspended solids from the runoff passing through the soil versus particles internally mobilised from the soil matrix. The former may require a different approach than the

from impervious areas that are 5 – 50 times the size of the infiltration area depending on the design of the systems. Thus, shear mobilisation and air-water interface scouring may also play important roles in these systems as the flow may increase suddenly and rapidly as well as reach high intensities. However, in terms of modifying the nature of engineered soils utilised in stormwater infiltration systems to better suppress the leaching of colloids, it is clear that dispersion/flocculation holds the greatest potential for adjustments.

The basic theory concerning flocculation and dispersion of any particle is linked to repulsive and attractive forces surrounding its surface. When repulsive forces dominate the colloids are dispersed while

latter to improve the treatment efficiency. In **paper IV** a novel method is described employing fluorescent microspheres (MS) (diameter = 5 μm) to mimic fine suspended solids in urban runoff which enabled us to distinguish the added particles from those potentially mobilised in the soil matrix. MS have previously been used for a wide range of applications within medical and engineering sciences, while in the of field soil science they have been used to study the transport and spatial distribution of colloids and microorganisms (Close et al. 2006; Mishurov et al. 2008; Cey et al. 2009; Passmore et al. 2010; Nielsen et al. 2011). However, no studies have been found in which MS have been used for testing of SUDS treatment efficiency.

The method employed in **paper IV** was inspired by the methods of Burkhardt et al. (2008) and Nielsen et al. (2011) which involved filtering of sub-samples onto polycarbonate membrane filters and quantifying the number of MS using fluorescence microscopy. Depending on the number of MS on the filters (more or less than 20 MS), quantification of MS was performed either by manual counting or digital determination by taking a statistically sound number of digital

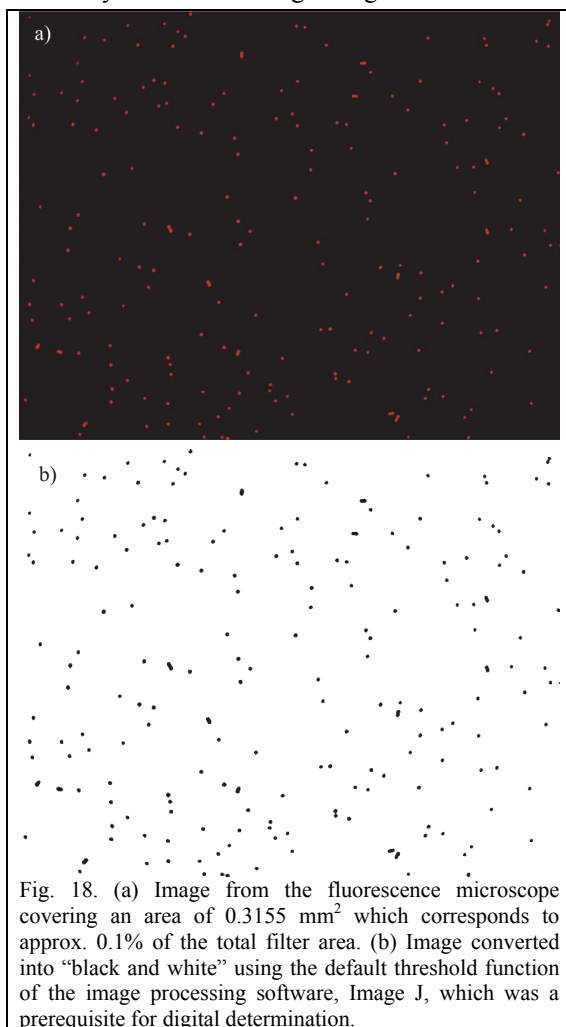


Fig. 18. (a) Image from the fluorescence microscope covering an area of 0.3155 mm^2 which corresponds to approx. 0.1% of the total filter area. (b) Image converted into "black and white" using the default threshold function of the image processing software, Image J, which was a prerequisite for digital determination.

photographs ($n = 20$) of each filter and using a java image processing programme for counting (Image J) (Fig. 18). The MS employed were custom-made Rhodamine B labelled polystyrene particles with a diameter of $4.90 \pm 0.24 \mu\text{m}$ (Microparticles GmbH, Berlin, Germany). Based on previous investigations using a heterogeneous but cheaper batch of fluorescent MS, it was found that in order to avoid flocculation and sticking to materials the MS had to be manufactured with a negative surface charge. This yielded fairly disperse solutions and uniform distribution on the filters as seen from Fig. 18. However, in case of mixing these MS with other substances, i.e. such as the synthetic road runoff fabricated for the performance evaluation of the filter soil columns (**paper IV**), their interaction with dissolved species should be thoroughly investigated. Although the method employed in this study proved to work well, the use of fluorescence microscopy for quantification is a rather time consuming process. The digital counting of particles could be further optimised by setting up the software for managing many images simultaneously. However, individual inspection of each image

seems to be necessary as tiny differences in the distance between the microscope objective and the MS, i.e. caused by the presence of larger soil particles or simply by irregularities of the filter or cover tape (**paper IV**), may cause the digital pixel area corresponding to one MS to vary significantly among images. Thus, the uncertainty of the method will inevitably increase as a result of more fully automated counting.

Other methods for quantification of fluorescent MS comprise flow cytometry equipped with a laser excitation lamp (Niehren and Kinzelbach, 1998; Bele et al., 2002) and spectrofluorometry (Goepfert and Hoetzl, 2009). Both methods have proven suitable and also possess a much greater potential for continuous in-situ measurements. However, in the case of flow cytometry it may be necessary to add a dispersion agent and shake thoroughly in order to separate flocculated particles. Furthermore, even though the density of the MS (1.05 g/cm^3) was close to that of water they do settle to the bottom of samples and therefore need shaking immediately before quantification. In the case of spectrofluorometry, the sensitivity seems to be limited by a rather high detection limit, i.e. 10^6 MS L^{-1} for $1 \mu\text{m}$ MS as shown by Goepfert and Hoetzl (2009).

As an alternative to using fluorescent MS as surrogates for fine suspended solids, Spencer et al. (2010) and (2011) used holmium labelled montmorillonite (a secondary clay mineral) as a sediment tracer to determine fine sediment transport dynamics in the aquatic environment as well as within a stormwater detention pond. Holmium can easily be incorporated into the lattice of the minerals in batch sorption experiments and recovered by ICP analytical techniques. The labelled sediment was ground to $<63 \mu\text{m}$ which is in accordance with the definition of fine suspended solids proposed in **paper I**. The study revealed that the sediment tracer clearly flocculated, but interactions with natural suspended sediments caused formation of flocs with significantly different properties than natural mud. In the stormwater pond system, the flocculation behaviour seemed to be similar to that of the natural pond sediment. However, in order to fully understand the dynamics and treatment potential of the pond towards the tracer there is a need to develop a suitable sampling protocol of the tracer 'cloud' (Spencer et al., 2011). All in all, it seems that this tracer with its near-natural flocculation behaviour could provide a more realistic picture of the removal potential of SUDS towards fine suspended solids compared with the use of fluorescent MS. On the other hand, the MS used were spherical, low density (1.05 g cm^{-3}), and dispersed particles which probably increased their tendency to stay in suspension compared with natural fine solids. Thus, the use of MS as surrogates for fine suspended solids probably represents a worst-case scenario in terms of treatment potential. The Holmium labelled tracer sediment could prove to be less costly than fluorescent MS, since the production of the sediment is cheaper than manufactured fluorescent MS, but this will also depend on the number of samples that need to be digested and analysed by ICP techniques. Finally, it should be noted that unless certified to represent a rather well documented and replicable particle size distribution the comparability among assessments of SUDS using the Holmium labelled tracer sediment is diminished compared with using continuously uniform and inert fluorescent MS.

Summary and recommendations

While no official guidelines exist for acceptable soil quality in infiltration facilities the results of the assessments of engineered filter soils in German roadside infiltration swales indicate that overall basic characteristics, i.e. soil composition and infiltration rate, are still largely in accordance with German guidelines for construction of such systems after 6 to 16 years of field operation. The observed heavy metal soil concentrations varied significantly among sites, and a distribution pattern with highest concentrations in the upper soil layer was observed for Cu and Zn, but not for Cd, Cr, and Pb. In most of the swales the concentration levels of one or more elements exceeded the limits for unpolluted soil. However, most of the soils would be acceptable for unrestricted usage in open construction works and do not pose a health risk to humans and wildlife. The soil P concentrations were generally higher than expected and for most soils corresponded to concentrations observed in fertilised agricultural soils which could indicate that some of the soils were in fact constructed using formerly agricultural soil.

Laboratory experiments performed on intact soil columns from two of the swales indicated that at high flow rates corresponding to catchment rain intensities around 34 mm h^{-1} , significant preferential flow may take place in the soils. However, in spite of this, the soils exhibited good treatment abilities towards dissolved heavy metals and fine suspended solids, but some internal DOC- or colloid facilitated mobilisation of these pollutants was observed resulting in potentially critical effluent concentrations of Cu, Zn, and Pb. In order to suppress DOC mobilisation in filter soils it is suggested that in the future, systematic testing is conducted of the influence of pH, base saturation capacity, and content of iron and aluminium oxides, as these parameters are found to be important for DOC mobilisation in soils. Fluorescent microspheres as surrogates for fine suspended solids were successfully utilised in the performance assessment and enabled us to distinguish incoming particles from those potentially mobilised in the soil.

In future roadside infiltration swales utilising engineered filter soil, it is suggested that special attention is paid to ensure low initial soil concentrations of Pb, Cd, and P, as well as the content and nature of organic matter in order to prolong the lifetime of the soils and limit DOC leaching. Furthermore, there is a need for more field performance assessments of such infiltration systems. Thus, it is recommended that future roadside infiltration swales are equipped with a system for long term field monitoring of the effluent from the filter soil covering a considerable areal segment of the swale.

Conclusions

The objectives of this Ph.D. project were approached on three different levels: A general level involving review and interpretation of existing literature and data sets on urban stormwater quality, case studies at the catchment level, and experimental assessments of methods and treatment efficiency on single facility level. Three essential challenges of controlling urban stormwater quality in the context of decentralised sustainable urban drainage systems (SUDS) have been identified and reviewed during the project. Additionally, existing roadside infiltration swales with engineered filter soil for infiltration and treatment of road runoff have been studied. Besides minor specific conclusions mentioned in the individual papers, the following specific and general conclusions may be drawn from the project:

Challenge #1 – Predicting the pollutant profile

- The number of possible pollutants and factors that influence their concentration and behaviour in urban stormwater runoff is substantial resulting in highly variable pollutant profiles that are difficult to predict and consistently categorise according to land- or surface use types. While trafficked surfaces on average yield higher pollutant concentrations compared to roofs, significant variations and deviations from this assumption prevail, and, in spite of seemingly similar surface types, the results from one surface may often not be transferable to another.
- Methods commonly used to more or less accurately predict the pollutant profile of urban runoff include the development of surrogate parameters as well as regression- and process based stormwater quality models. However, once established such relationships and models are predominantly site specific and thus rarely transferrable to other sites, unless thoroughly recalibrated with monitoring results.
- In terms of monitoring urban stormwater quality, results may be of limited use unless special emphasis is placed on sampling methods and frequency since the number of in-storm samples and the number of sampled storm events has a significant influence on the confidence interval of average concentrations. This issue should be made more transparent to decision makers and other professionals in this field of expertise in the future.

Challenge #2 – Setting emission limit values

- Local authorities are about to implement measures in municipal action plans in order to comply with the requirements of the EU Water Framework Directive, but the potential impacts of urban stormwater discharges via SUDS are still unclear and thus difficult to lucidly address in such plans.
- Concerning SUDS discharge to surface waters a number of circumstances make definitions of appropriate emission limit values difficult, including rigorous international water quality standards, uncertainties regarding the presence and behaviour of EU priority pollutants in

urban runoff and surface waters, uncertainties regarding potential impacts on sediment quality, as well as frequent lack of responsibility placement.

- Concerning emission limit values for SUDS that discharge to groundwaters the major uncertainties are related to identifying upward pollution trends, possible interactions between groundwater and surface waters, soils, and sediments, as well as the fate of pollutants in the vadose zone below infiltration devices.
- The overall challenge faced by local authorities in terms of setting appropriate emission limit values seems to be taking all the prevailing uncertainties and knowledge gaps into account without limiting the possibilities for future implementation of SUDS.

Challenge #3 – Selecting treatment options

- Few multicriteria decision tools have been developed for supporting the selection of SUDS, but in terms of integrating treatment performance they all have drawbacks or limitations curtailing their utility in decision making.
- The International Stormwater BMP Database constitutes the largest publicly available collection of SUDS treatment performance assessments which may provide useful information for decision makers, but non-existing design criteria for SUDS or inconsistent compliance with such criteria has limited its use. As the database further develops with well-designed SUDS it should be noted that in order for the data to be transferrable to local or regional SUDS it is a prerequisite that local conditions and design criteria match those of the BMP database.
- Solid benchmarking of the treatment performance of various well defined SUDS is a prerequisite for sound decision making regarding discharge quality and selection of SUDS, but this is complicated by the multiplicity of factors that influence the performance of SUDS, i.e. design criteria, operation conditions, sampling procedures, and data analysis.
- In terms of treatment efficiency there is no unambiguously “best” SUDS as no single system can provide the lowest effluent concentrations across all water quality parameters.
- Based on a comprehensive review of urban stormwater quality data, a *minimum data set* of water quality parameters is suggested to facilitate future broad-spectrum testing and benchmarking of SUDS treatment performance. This data set consists of the following water quality parameters: (i) fine fraction of suspended solids ($< 63 \mu\text{m}$), (ii) total concentrations of zinc (Zn) and copper (Cu), (iii) total concentrations of phenanthrene, fluoranthene and benzo(b,k)fluoranthene, and (iv) total concentration of phosphorus (P) and nitrogen (N).

- There is still very limited data available on the behaviour and fate of EU priority pollutants in SUDS.

Assessment of filter soil as treatment option

- This study constitutes the first larger assessment of several existing long term functioning (6-16 years) German roadside infiltration swales utilising engineered filter soils intended for optimised infiltration and treatment of road runoff.
- The investigations indicate that after 6 to 16 years of field operation, the overall basic characteristics such as soil composition and infiltration rate are still acceptable according to German guidelines for such systems.
- In most of the swales the concentration levels of one or more heavy metals exceeded the limits for unpolluted soil whereas most of the soils would be acceptable for unrestricted usage in open construction works and do not pose a health risk to humans or wildlife.
- Observations and calculations indicated that low initial soil concentrations of Pb, Cd, and P, should be ensured in order to prolong the lifetime of the soils.
- The soils generally showed good treatment abilities towards dissolved heavy metals and excellent removal of fine suspended solids, but some internal DOC- or colloid facilitated mobilisation was observed resulting in potentially critical effluent concentrations of Cu, Zn, Pb, and P.
- Fluorescent microspheres as surrogates for fine suspended solids were successfully employed for testing the treatment performance and enabled differentiation between incoming particles and those potentially mobilised in the soil.
- Overall, the use of filter soil as a treatment option shows great promise but needs to be further developed and documented in order to ensure that internal mobilization issues are limited

Perspectives

Due to the nature of the project and the many interactions with decision makers and other professionals during the process, the work conducted was frequently reviewed in a greater context. Here the implications, recommendations, and future perspectives of the project are considered in relation to *management practices* of urban stormwater quality and *future research* needs.

Management practices

Given the consistently dynamic pollutant profile of urban stormwater runoff and the fact that high uncertainty within predictions is likely to prevail, it is suggested that in the near future decision makers and researchers in Denmark should agree on an *acceptable level of uncertainty* on which to base decisions regarding future practices. In principle, this level should be defined individually by the municipalities, but in order to communicate all relevant knowledge and facilitate the process, national guidelines based on scientific and well-considered argumentation could be developed. In this context and if we are to improve our knowledge basis and experience with SUDS it seems essential that authorities have the courage to grant realistic discharge permissions in spite of certain uncertainties and current knowledge gaps. However, adequate monitoring and reporting of emissions should be required in order to identify and reverse potential non-compliance with emission limit values. Concerning emission limit values, it is suggested that for a yet undefined period of time (perhaps 5 to 10 years) authorities need to define site specific emission limit values for each SUDS employed. However, within this period of time we should aim at gaining sufficient experience and documentation regarding the treatment efficiency of a range of SUDS to enable implementation of design criteria rather than emission limit values. Thus, in a national context, it seems relevant to establish which SUDS require thorough documentation of treatment efficiency in the future, i.e. which SUDS could potentially become well documented national best available technologies (BAT) for treatment of polluted urban runoff? In order to frame the premises for documenting the selected SUDS it is essential to establish well considered design criteria as well as best SUDS practices, i.e. operation and maintenance. Finally, a number of pilot facilities and monitoring programmes employing standardised data sets as well as sampling and analytical procedures should be launched. In relation to the testing and documentation of existing SUDS as well as emerging technologies, it is recommended that the minimum data set proposed in **paper I** should constitute the backbone in considerations regarding standardised data sets. As a very first approach to evaluating the treatment efficiency of SUDS, future work could involve an examination of the International Stormwater BMP Database to extract those SUDS and studies that may be applicable in a Danish context.

There are many ways to design monitoring programmes of urban stormwater runoff, but often they are dictated by limited budgets and time frames yielding results of questionable reliability. In order to make qualitative assessments of monitoring results more transparent to decision makers, it is suggested that a classification system should be developed in which monitoring results can be grouped according to their reliability. It is beyond the scope of this Ph.D. project to develop such a system, but as a first shot from the hip it is suggested that (i) the method of sampling (i.e. automatic vs. manual, single grab samples vs. flow proportional composite

samples), (ii) the number of in-storm samples collected, and (iii) the number of sampled storm events should be included in the system to develop a transparent ranking of reliability, i.e. in terms of probability of accuracy. However, the development of such a system should be based on a thorough statistical analysis in line with that of Maestre and Pitt (2005).

Within a recently established national partnership for climate adaptation and innovation called *Water in Urban Areas* (www.vandibyer.dk/english/), several new innovation projects and consortia are currently being initiated or are in the application phase. An innovation consortium concerning the future management of urban stormwater quality as well as documentation of SUDS is currently in the application phase. Provided it is approved, it is likely that, to the extent it is possible, many of the findings and considerations described in this thesis will be further addressed within this consortium.

Future research

It has been established that our current knowledge about a wide range of xenobiotic organic compounds, i.e. many of the EU priority pollutants, is scarce. It is important that more measurements of these compounds are performed in the future to establish which ones occur in potentially critical concentrations in urban stormwater runoff as well as to what extent their removal in SUDS may be documented by the parameters of the minimum data set proposed in **paper I**, predominantly the fine suspended solids and PAHs.

The assessment of existing German roadside infiltration swales with engineered filter soil (**paper III**) pointed out that no guidelines are available for acceptable soil quality, i.e. in terms of maximum allowable heavy metal concentrations. Thus, currently it is not possible to unambiguously evaluate the soils based on soil analysis. Future projects and research on such systems should address this issue and explore the possibilities for linking the soil concentrations to leaching potential. The approach taken in **paper III** and **IV** did not allow for such assessments.

The leaching studies conducted on intact soil columns from existing German roadside infiltration swales (**paper IV**) showed that in-situ mobilised DOC (or colloids in general) may act as a carrier of adsorbed pollutants out of the filter soil layer. In order to suppress DOC-associated pollutant mobilisation in filter soils it is suggested that systematic testing is conducted of the influence of pH, base saturation capacity, and content of iron and aluminium oxides, as these parameters are found to be important for DOC mobilisation in soils. Furthermore, to support the results of **paper III** and **IV** as well as account for the variations not represented in laboratory soil column studies, there is a need for more field performance assessments of such infiltration systems. Thus, it is recommended that future roadside infiltration swales are equipped with a system for long term field monitoring of the effluent from the filter soil covering a considerable areal segment of the swale. A newly started innovation project under the national partnership, *Water in Urban Areas* deals with testing and development of engineered filter soil. At present the project is scheduled to run for four years, and one pilot facility has been established along a new parking area (approximately 200 cars) at the Southern University of Denmark in Odense. This infiltration swale has been constructed in a way that allows for collecting effluent samples from immediately below

the filter soil over larger areal segments. It is possible that other infiltration swales employing engineered filter soil will be included in the project. As the writer of the present thesis is responsible for this innovation project, the experiences and lessons learned from this Ph.D. project will find direct usage here.

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