Integrated Modelling of Stormwater Management Strategies and CSO Impacts on Urban Rivers

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an der Fakultät VI – Planen Bauen Umwelt
der Technischen Universität Berlin
zur Erlangung des akademischen Grades

Doktor der Ingenieurwissenschaften
– Dr.-Ing. –

genehmigte Dissertation

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Tag der wissenschaftlichen Aussprache: 17. Februar 2022

Berlin 2022

Preface

First, I would like to express my gratitude to my supervisor Reinhard Hinkelmann for giving me the possibility to conduct my doctoral thesis at the Chair of Water Resources Management and Modeling of Hydrosystems at Technical University Berlin.

Second, I would like to express my deep gratitude to Pascale Rouault for giving me the chance to work on so many interesting projects dealing with urban water systems, which partly resulted in this thesis. Also thank you, Pascale, for being such an excellent and dedicated boss since more than 10 years and for your trust and support in constantly developing and broadening my skills.

I would also like to thank my other colleagues at Kompetenzzentrum Wasser Berlin (KWB) for making this such a nice place to work at. I am especially grateful to Andreas Matzinger, who not only introduced me to the world of environmental modelling but who has always been a great discussion partner in any imaginable topic. Further, I would like to thank Nicolas Caradot for his open mind and helpfulness and for opening me the door to the field of sewer asset management, another fascinating research topic. I am also thankful to Hauke Sonnenberg for having introduced me to the world of programming and for having raised a smile so many times, with his humour and his handstands. And finally thanks to the two Michaels of our team, Michael Rustler and Michael Stapf, for always having an open ear for any kind of question.

Many thanks also go to the colleagues of Berliner Wasserbetriebe (BWB) with whom I worked together over the past years. Here, I would like to point out the special support I received from Erika Pawlowsky-Reusing, who generously shared her in-depth knowledge of the Berlin drainage system and its specifics. I am also very thankful to the colleagues of the Senate Departments of Berlin and the Waterways and Shipping Administration (WSA) who likewise supported this thesis and the related projects with a lot of data and useful feedback.

Gratitude is also expressed to the colleagues of the German Federal Institute of Hydrology (BfG), in particular to Volker Kirchesch, Andreas Schöl, Annette Becker, and Helmut Fischer, who spent great effort on the development and constant improvement of the river water quality model QSim. The same goes to Frank Schumacher of Ingenieurbüro für Wasser und Umwelt (IWU) and to Christian Wolter of the Leibniz-Institute of Freshwater Ecology and Inland Fisheries (IGB), who have been much valued discussion partners for questions on river hydraulics and fish ecology.

I am deeply grateful for the support I got from my family and my friends in so many ways.

And finally, a warm and heartfelt "Thank you!" to you, Annika, and to our sons Theo and Piet for your support and motivation and even more for the joy and distraction you gave whenever needed. You are the best!

Publications of cumulative doctoral thesis

Journal publications (in chronological order):

1. **Riechel, M.**, Matzinger, A., Pawlowsky-Reusing, E., Sonnenberg, H., Uldack, M., Heinzmann, B., Caradot, N., v. Seggern, D., Rouault, P. (2016). Impacts of combined sewer overflows on a large urban river – understanding the effect of different management strategies. *Water Research* 105, 264-273. https://doi.org/10.1016/j.watres.2016.08.017
2. **Riechel, M.**, Matzinger, A., Pallasch, M., Joswig, K., Pawlowsky-Reusing, E., Hinkelmann, R., Rouault, P. (2020). Sustainable urban drainage systems in established city developments: Modelling the potential for CSO reduction and river impact mitigation. *Journal of Environmental Management* 274, 111207. https://doi.org/10.1016/j.jenvman.2020.111207

Book chapters (in chronological order):

1. **Riechel, M.**, Uldack, M., Heinzmann, B., Pawlowsky-Reusing, E., Matzinger, A. (2015). Demonstration of a planning instrument for integrated and impact based CSO control under climate change conditions. Book chapter in: *Climate Change, Water Supply and Sanitation: Risk assessment, management, mitigation and reduction*, 65-69. IWA Publishing. https://doi.org/10.2166/9781780405001
2. **Riechel, M.**, Seis, W., Matzinger, A., Pawlowsky-Reusing, E., Rouault, P. (2019). Relevance of Different CSO Outlets for Bathing Water Quality in a River System. Book chapter in: *New trends in Urban Drainage Modelling: UDM 2018*, 859-863. Editor: Giorgio Mannina, Springer. https://doi.org/10.1007/978-3-319-99867-1

Conference papers:

1. **Riechel, M.**, Matzinger, A., Sonnenberg, H., Caradot, N., Meier, I., Heinzmann, B., Rouault, P. (2012). Validation and sensitivity of a coupled model tool for CSO impact assessment in Berlin, pp. 8. *Proceedings of the 6th International Congress on Environmental Modelling and Software (iEMSs)*. Leipzig, Germany, 2012.

Abstract

Diffuse pollution from combined sewer overflows (CSO) is one of the most prevailing challenges for the ecological quality of urban rivers. Of particular concern for flow-regulated rivers and their aquatic organisms are depressions in dissolved oxygen (DO), as regularly observed in the Berlin River Spree and its side channels. In some cases, ammonia toxicity is also an issue. Moreover, pathogen emissions via CSO outlets upstream of bathing waters pose a risk for human health. Ongoing climate change in the form of more intense rainfalls and higher temperatures is expected to aggravate these negative impacts.

Two different approaches are considered for CSO impact mitigation: sewer-based CSO control measures that provide underground storage or treatment and sustainable urban drainage systems (SUDS), such as green roofs or infiltration swales. However, it is not straightforward to decide which mitigation measures would be ecologically best for the receiving water body. This becomes evident from the scarcity of full-scale demonstrations of integrated sewer-river-models, especially for complex systems with a multitude of CSO outlets. Further, there appear to be no modelling studies that quantify the potential of SUDS to mitigate adverse impacts on aquatic organisms. As a consequence, little is known about measure effects, in particular those of SUDS, and governing processes in the receiving water body.

The aim of this thesis is to quantify CSO impacts in the form of DO depressions, the importance of associated river processes, and the effect of different mitigation measures with a strong focus on SUDS. For this purpose, an integrated modelling approach consisting of a sewer model implemented in InfoWorks, a river water quality model implemented in Hydrax-QSim, and an impact assessment approach for fish-critical DO conditions in the river was developed for the city of Berlin. The modelling approach was used to explore the river processes that lead to DO depressions after CSO and quantify the effects of different mitigation strategies, also under consideration of climate change. In a second step, the model tool was complemented with a detailed rainfall-runoff model that comprises model components for a multitude of SUDS. This enhanced integrated model was used to quantify the potential of realistic SUDS strategies to reduce CSO emissions and mitigate critical oxygen deficits in the river. Finally, a virtual tracer approach to determine the microbiological contamination potential of different CSO outlets was developed and tested. The approach can be used to locate effective measures for improving bathing water quality.

As a first outcome, integrated sewer-river-models were shown to be capable of representing CSO impacts at the city-scale under complex urban conditions in good agreement with measurements. Model sensitivity to changes on the catchment's surface, in the sewer system, and in the river was demonstrated as an important plausibility check and prerequisite for scenario analysis. The research further identified three river processes that dominate the DO

budget after CSO and that vary in importance with flow time: i) the degradation of organic matter by heterotrophic bacteria, ii) the inhibition of the phytoplankton activity due to CSO-induced turbidity, and iii) mixing of river water with CSO spill water poor in DO.

The identified processes are influenced by different mitigation measures which, in turn, can determine where along the river positive effects will unfold. Important improvements were found for an increase in storage capacity in the sewer system and for a reduction in surface runoff via SUDS, which both influence all three identified river processes. As an example, acute oxygen deficits can be mitigated completely with the investigated SUDS strategies, that comprise about one third of the catchment's impervious area. However, climate change would partly counterbalance these achievements and further increase background oxygen stress during dry weather. Regarding required model complexity, the detailed SUDS simulation, which considers attenuation and delay of runoff, outperforms global runoff reduction scenarios, which hence are only recommended for preliminary planning purposes.

Finally, the developed tracer approach for pathogen emissions via CSO revealed that hotspots for bathing water contamination do not necessarily correspond with CSO volume hotspots. This finding is especially important for locating measures to improve bathing water quality, which may have their maximum effect at other locations than measures that tackle oxygen deficits.

The results of this thesis enhance knowledge on CSO impacts, measure effects, and integrated modelling techniques. This knowledge can, in turn, be used by decision makers for impact-based strategic planning of CSO mitigation measures in urban areas with oxygen deficits being the main stressor for aquatic organisms.

Zusammenfassung

Die diffuse Verschmutzung durch Mischwasserüberläufe ist eine der größten Herausforderungen für urbane Gewässer. Besonders problematisch für abflussregulierte Fließgewässer und die darin lebenden Organismen ist das akute Absinken der Sauerstoffkonzentration infolge des Abbaus von eingeleitetem organischen Material, wie es in der Berliner Stadtspree und ihren Nebenkanälen regelmäßig beobachtet wird. In einigen Gewässern führen auch erhöhte Konzentrationen von Ammonium zu ökologischen Beeinträchtigungen. Darüber hinaus resultieren aus den über Mischwasserüberläufe eingetragenen pathogenen Keimen auch Risiken für die menschliche Gesundheit in stromabwärts gelegenen Badegewässern. Unter fortschreitendem Klimawandel werden sich diese negativen Auswirkungen weiter verschärfen.

Um die Auswirkungen von Mischwasserüberläufen abzumildern, werden Maßnahmen der Kanalnetz- und der Regenwasserbewirtschaftung eingesetzt, z. B. unterirdische Speicherbecken, Reinigungsanlagen, Gründächer oder Versickerungsmulden. Die Auswahl geeigneter Maßnahmen erfolgt jedoch in den meisten Fällen emissionsbasiert, d. h. ohne Nachweis des ökologischen Nutzens für das Gewässer. Ein zentrales Hindernis ist die geringe Verbreitung integrierter Kanalnetz- und Gewässergütemodelle, insbesondere für komplexe Systeme mit einer Vielzahl unterschiedlicher Mischwasserauslässe. Über die konkrete Wirkung von Maßnahmen und die im Gewässer ablaufenden Prozesse ist daher nur wenig bekannt. Auch die positiven Effekte der dezentralen Regenwasserbewirtschaftung auf die Wasserqualität und die aquatischen Organismen wurden bisher kaum quantifiziert.

Ziel der vorliegenden Dissertation ist es, die Auswirkungen von Mischwasserüberläufen auf den Sauerstoffhaushalt urbaner Fließgewässer, die damit verbundenen Prozesse sowie die positiven Effekte der Kanalnetz- und insbesondere der Regenwasserbewirtschaftung aufs Gewässer zu quantifizieren. Zu diesem Zweck wurde ein integriertes Modellwerkzeug basierend auf dem Kanalnetzmodell InfoWorks, dem Gewässergütemodell Hydrax-QSim und einem immissionsbasierten Bewertungsansatz für fischkritische Sauerstoffdefizite am Beispiel von Berlin aufgebaut. Das Modellwerkzeug wurde verwendet, um die nach Mischwasserüberläufen im Gewässer vorherrschenden Prozesse des Sauerstoffhaushaltes und die Auswirkungen verschiedener Bewirtschaftungsstrategien, auch unter Berücksichtigung des Klimawandels, zu quantifizieren. In einem zweiten Schritt wurde das Modellwerkzeug um ein detailliertes Niederschlag-Abfluss-Modell erweitert, das über Modellkomponenten für verschiedenste Maßnahmen der Regenwasserbewirtschaftung verfügt. Mit dem erweiterten Modellwerkzeug wurden unterschiedliche, konkret in einem Stadtquartier verortete Maßnahmen hinsichtlich ihres Potenzials zur Reduzierung von Mischwasserüberläufen und zur Vermeidung kritischer Sauerstoffdefizite im Gewässer untersucht. Darüber hinaus wurde ein virtueller Tracer-Ansatz

zur Ermittlung des mikrobiologischen Verschmutzungspotenzials verschiedener Mischwasserauslässe entwickelt und getestet. Der Ansatz ermöglicht es, Maßnahmen zur Verbesserung der Badegewässerqualität optimal zu verorten.

Zunächst konnte gezeigt werden, dass integrierte Modelle für das Kanalnetz und das Gewässer in der Lage sind, die Auswirkungen von Mischwasserüberläufen auch unter komplexen Bedingungen in guter Übereinstimmung mit Messungen abzubilden. Das aufgebaute integrierte Modell reagiert sensitiv auf Veränderungen im Stadtgebiet, im Kanalnetz sowie im Gewässer, was eine wichtige Plausibilitätsprüfung und Voraussetzung für die Szenarienanalyse ist. Drei Gewässerprozesse wurden identifiziert, die den Sauerstoffhaushalt nach Mischwasserüberläufen in Abhängigkeit der Fließzeit dominieren: i) der Abbau von organischem Material durch heterotrophe Bakterien, ii) die reduzierte Photosyntheseaktivität aufgrund erhöhter Trübung und iii) die Einmischung von sauerstoffarmem Mischwasser in die Gewässerströmung.

Die identifizierten Prozesse werden durch verschiedene Bewirtschaftungsmaßnahmen in unterschiedlicher Weise beeinflusst. Bedeutende positive Effekte wurden für eine Erhöhung der Speicherkapazität im Kanal und für eine Reduzierung des Oberflächenabflusses durch Maßnahmen der Regenwasserbewirtschaftung ermittelt, die alle drei identifizierten Gewässerprozesse beeinflussen. Mit beiden Maßnahmentypen können fischkritische Sauerstoffdefizite in Häufigkeit und Dauer deutlich reduziert und im Falle der Regenwasserbewirtschaftung sogar vollständig verhindert werden. Der Klimawandel würde diese positiven Effekte jedoch zum Teil kompensieren und zudem die Hintergrundbelastung des Sauerstoffhaushaltes während Trockenwetter erhöhten. Die durch einige Maßnahmen der Regenwasserbewirtschaftung erzielte Abflussdämpfung und -verzögerung kann je nach Regenereignis einen wichtigen Anteil an der Reduzierung von Mischwasserüberläufen haben. Daher sollten diese Effekte für die detaillierte Maßnahmenplanung möglichst gut im Modell abgebildet werden.

Zuletzt zeigte der entwickelte Tracer-Ansatz für Emissionen pathogener Keime, dass Hotspots mikrobieller Verschmutzung nicht immer dort liegen, wo auch die größten Volumina eingeleitet werden. Daraus resultieren wichtige Schlussfolgerungen für die Verortung von Maßnahmen zur Verbesserung der Badegewässerqualität.

Die Ergebnisse dieser Arbeit bringen wertvolle Erkenntnisse zu Auswirkungen von Mischwasserüberläufen und den Effekten unterschiedlicher Maßnahmen aufs Gewässer. Die verwendeten und erweiterten Ansätze der integrierten Modellierung können für die immissionsbasierte Maßnahmenplanung, insbesondere in urbanen Systemen mit regelmäßig auftretenden akuten Sauerstoffdefiziten, eingesetzt werden.

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List of Abbreviations

BMP	Best management practice
BOD	Biological oxygen demand
Chl-a	Chlorophyll-a
COD	Chemical oxygen demand
CSO	Combined sewer overflow
DO	Dissolved oxygen
LC_{50}	Lethal concentration for 50% of the organisms
LID	Low impact development
MPN	Most probable number
NH_4-N	Ammonium nitrogen
NSE	Nash-Sutcliffe efficiency
PID	Proportional-integral-derivative (controller)
PO_4	Ortho-phosphate
Q	Discharge
r^2	coefficient of determination
RMSE	Root mean square error
RTC	Real-time control
SUDS	Sustainable urban drainage system
TN	Total nitrogen
TP	Total phosphorus
TSS	Total suspended solids
UV	Ultraviolet
v	Flow velocity
WSUD	Water sensitive urban design
WWTP	Wastewater treatment plant

Chapter 1: Introduction

1.1 Combined sewer overflows and river impacts

Over the last 150 years, urban drainage systems have helped significantly in solving two major problems associated with urbanisation. In the first place, they have led to improvements in public health by effectively removing sanitary sewage from human settlements. Further, they have reduced the risk of urban flooding by collecting and conveying stormwater from impervious areas. Despite these achievements, urban drainage systems have also produced a shift of hygienic and aesthetic problems from the cities to the surface waters by transporting wastewater to treatment plants and then – after treatment – to rivers, lakes, or the sea. In the case of storm drainage, they have increased the risk of fluvial flooding due to the rapid runoff and discharge of stormwater, which produces higher and more sudden peaks in river flow (Fletcher et al., 2013). Lastly, urban drainage systems discharge pollutants washed off surfaces and, for combined sewer systems, also originating from sanitary sewage into the receiving water bodies.

Combined sewer systems were the technical standard in the early days of urban drainage and are still encountered today in many European cities (Lund et al., 2019). In Germany, 53% of the population is connected to combined sewers with a total length of 247,000 km (Dettmar and Brombach, 2019). In contrast to separate sewer systems, where stormwater and sanitary sewage are collected separately, combined sewer systems drain both water types in a single network. This mix of water is usually treated in a wastewater treatment plant before being discharged to a receiving water body. However, in case of intense rainfalls, when treatment plants, pressure mains, and pumping stations reach their capacity limits, part of that water is conveyed directly to the water body in an event called combined sewer overflow (CSO).

In a CSO event, considerable volumes and pollutant loads are emitted over a relatively short duration, typically ranging from minutes to hours (Andrés-Doménech et al., 2010; Sandoval et al., 2013). These short-term discharges can easily exceed natural river discharges (Zhu et al., 2017) and lead to serious water quality deficits (Borchardt et al., 2003; Krejci et al., 2004a). The most relevant pollutants found in CSO are organic matter (Seidl et al., 1998; Even et al., 2007), suspended solids (Brzezinska et al., 2016), nutrients (Barone et al., 2019), heavy metals (Birch et al., 2011; Xu et al., 2018), and pathogens (Passerat et al., 2011; Madoux-Humery et al., 2013; Al Aukidy and Verlicchi, 2017). Over the past decade, research has further emphasized the presence of priority pollutants, such as pesticides, pharmaceuticals, or illicit drugs (Gasperi et al., 2012; Launay et al., 2016; Munro et al., 2019), and microplastics (Dris et al., 2018).

Impacts of CSO on surface waters and aquatic organisms are diverse and depend strongly on the type of affected water body. Table 1.1 gives an overview on short- and long-term CSO impacts and their main affected water body types.

Table 1.1: Possible CSO impacts and affected water body types, after Borchardt et al. (2001), Rossi (2004) and Matzinger et al. (2012a).

Impact	Affected water body type *
Short-term	
Hydraulic stress	Small streams
Oxygen depressions	Flow-regulated rivers
Ammonia toxicity	Small streams and flow-regulated rivers
Microbiological contamination	Small streams, medium to large streams, flow-regulated rivers, lakes, and sea (if used for bathing)
Long-term	
Sediment accumulation and contamination	Flow-regulated rivers, lakes, and sea
Eutrophication	Flow-regulated rivers, lakes, and sea
Xenobiotics	Small streams and flow-regulated rivers

* River types are distinguished as follows:

- Small streams: average discharge $Q < 0.1$ m^3 s^{-1}, width < 1 m
- Medium to large streams: $Q > 0.1$ m^3 s^{-1}, width > 1 m and flow speed $v > 0.5$ m s^{-1}
- Flow-regulated rivers: $Q > 0.1$ m^3 s^{-1}, width > 5 m and $v < 0.5$ m s^{-1}

Flow-regulated rivers are potentially impacted by all quality-related CSO impacts listed in Table 1.1, mainly due their comparably low discharges and dilution capacities. Of particular concern – and the main focus of this thesis – are depressions in dissolved oxygen (DO), as observed in the Berlin River Spree and its side channels every year during summer months (Riechel, 2009). Oxygen deficits after CSO were also described for the River Seine in France (Even et al., 2007), the Dommel River in the Netherlands (Benedetti et al., 2013; Moreno-Rodenas et al., 2017), and the Chicago waterway system in the United States (Melching et al., 2013), among others.

Oxygen deficits occur primarily as a consequence of i) the immediate degradation of organic compounds or ammonium in the water column or ii) the delayed degradation after settling of particulate compounds to the river (Harremoës, 1982; Hvitved-Jacobsen, 1982). Additional processes leading to a decrease in oxygen concentrations after CSO have been investigated in the framework of this thesis (Riechel et al., 2016).

The specific effects of low oxygen concentrations on the aquatic fauna, including fish and invertebrates, are manifold and range from behavioural impairment to death. In general, the following reactions can be observed according to Borchardt (1992) and Lammersen (1997):

- changes in general behaviour, e.g. increased respiration or reduced swimming activity;
- escape behaviour and drift;
- reduced growth and changes in other physiological variables;
- mortality.

Critical DO concentrations of aquatic organisms are primarily a function of exposure time but also depend on life stage and external factors, e.g. water temperature or concurring ammonia toxicity. Lammersen (1997) proposes lethality thresholds of 2.5 mg DO L^{-1} for salmonid fish and 1.5 mg DO L^{-1} for cyprinids at exposure times of 10 minutes, based on the work of Downing and Merkens (1957) and Milne et al. (1992), among others. These thresholds increase with exposure time and should protect all life stages of a population, even larvae and juvenile fish which have higher oxygen demands than adult individuals (Doudoroff and Shumway, 1970; US EPA, 1986). Sublethal effects, such as reduced swimming activity or escape behaviour of fish, have been observed already at concentrations of around 5 mg DO L^{-1} (Doudoroff and Shumway, 1970). For salmonid fish, reduced embryonic development and larval growth were already reported at concentrations above 5 mg DO L^{-1} for durations of around 24 hours (Doudoroff and Shumway, 1970). For invertebrates, which are typically more robust than fish, drift and migration have been observed at DO concentrations of 3.5 to 4.5 mg L^{-1} over an exposure time of 1 to 6 hours (Gammeter and Frutiger, 1990). High temperatures and elevated ammonia concentrations generally intensify oxygen stress (Downing and Merkens, 1957; Milne et al., 1992).

A second detrimental effect of CSO dealt with in this thesis is the microbiological contamination of surface waters, relevant if used for bathing. It is the result of peak loads of hygienically relevant bacteria (e.g. *Salmonella* or *Campylobacter*), viruses (e.g. *Adenovirus* or *Rotavirus*), or parasites (e.g. *Entamoeba* or *Giardia*), that primarily originate from wastewater discharges (Kreikenbaum, 2004). They can have negative effects on human health by causing gastrointestinal and respiratory diseases, pneumonia, and bronchitis and thus restrict the recreational use of the water bodies (US EPA, 2004). In the longer term, bacteria and viruses may be of importance due to the conservation in sediments (Borchardt et al., 2003). Microbial safety of recreational waters is regulated by the European Bathing Water Directive (EU, 2006) via percentile thresholds for *Escherichia coli* and intestinal enterococci, two fecal indicator bacteria. CSO discharges can also be a threat for drinking water sources if receiving waters are used for water supply (Marsalek and Rochfort, 2004; US EPA, 2004).

The observed negative impacts from CSO, in particular oxygen stress, are expected to aggravate under ongoing climate change (Astaraie-Imani et al., 2012). First, more intense rainfalls, as observed and predicted for many localities (e.g. Grieser and Beck, 2002; Reusswig et al., 2016; Myhre et al., 2019), lead to higher hydraulic loadings to urban drainage systems and hence more frequent and intense CSO events (Bendel et al., 2013; Abdellatif et al., 2015). Second, the already observed and furtherly expected increase in temperature (e.g. Lotze-Campen et al., 2009; IPCC, 2018) aggravates the vulnerability of the ecosystem towards oxygen stress. Reasons for that are:

- decreased oxygen solubility at warm temperatures (Weiss, 1970);
- accelerated degradation processes and hence oxygen consumption by heterotrophic bacteria (Lønborg et al., 2018);
- an increased oxygen demand of fish and invertebrates (Downing and Merkens, 1957).

The European Water Framework Directive (EU, 2000) acknowledges CSO as a major thread for urban water bodies and requests environmental authorities to reduce negative impacts and improve ambient water quality in general. Several countries have also established non-binding technical guidelines that recommend water-quality goals for short-term CSO exposure in the form of duration-concentration thresholds (e.g. BWK, 2007; FWR, 2018). These guidelines follow an impact-based approach and take the ecologic requirements of the affected aquatic organisms into account. In practice, however, CSO control measures are still typically designed on basis of emission criteria, such as CSO frequency or volume (Botturi et al., 2020), without directly addressing the specific impacts in the river.

The river perspective, however, is important as the severity of detrimental effects on aquatic organisms cannot be solely explained with emission criteria (Lau et al., 2002; Freni et al., 2010). Instead, they vastly depend on the river discharge, the level of background pollution, and, first and foremost, the species that populate the respective water body. In this context, an impact-based modelling and assessment approach is employed in this thesis, to quantify the effect of mitigation measures and analyse associated river processes. In addition, a novel tracer approach to estimate the hygienic relevance of different CSO outlets of a large combined sewer network is demonstrated, helping to allocate effective measures for the improvement of bathing water quality.

1.2 Mitigation measures

Measures for the reduction of negative impacts from CSO can be divided into two main categories: sewer-based CSO control measures and sustainable urban drainage systems. Sewer-based CSO control measures are part of the drainage system and aim at preventing combined sewage and associated pollutants from being discharged into receiving waters. They rely on temporary retention or treatment, usually underground, and principally aim at reducing CSO emissions. Sustainable urban drainage systems (SUDS), on the other hand, are located on the catchment's surface and aim at preventing stormwater runoff and associated pollutant from entering the drainage system. Besides the protection of surface waters, SUDS often have multiple additional functions or effects, e.g. the reduction of urban heat islands (Norton et al., 2015), the increase in biodiversity (Pille and Saeumel, 2017), or the restoration of the natural water balance (Khadka et al., 2019). Comprehensive overviews on SUDS - also referred to as low impact development (LID), best management practice (BMP), or water sensitive urban design (WSUD) - are given by Dietz (2007), Ahiablame et al. (2012), Zhou (2014), and Eckart et al. (2017). Fletcher et al. (2015) elaborate on the diverse use of terminology related to SUDS.

In the following, the principal sewer-based CSO control measures and SUDS as well as a few alternative options for CSO impact mitigation are discussed. Figure 1.1 visualises the implementation of mitigation measures in an exemplary city quarter.

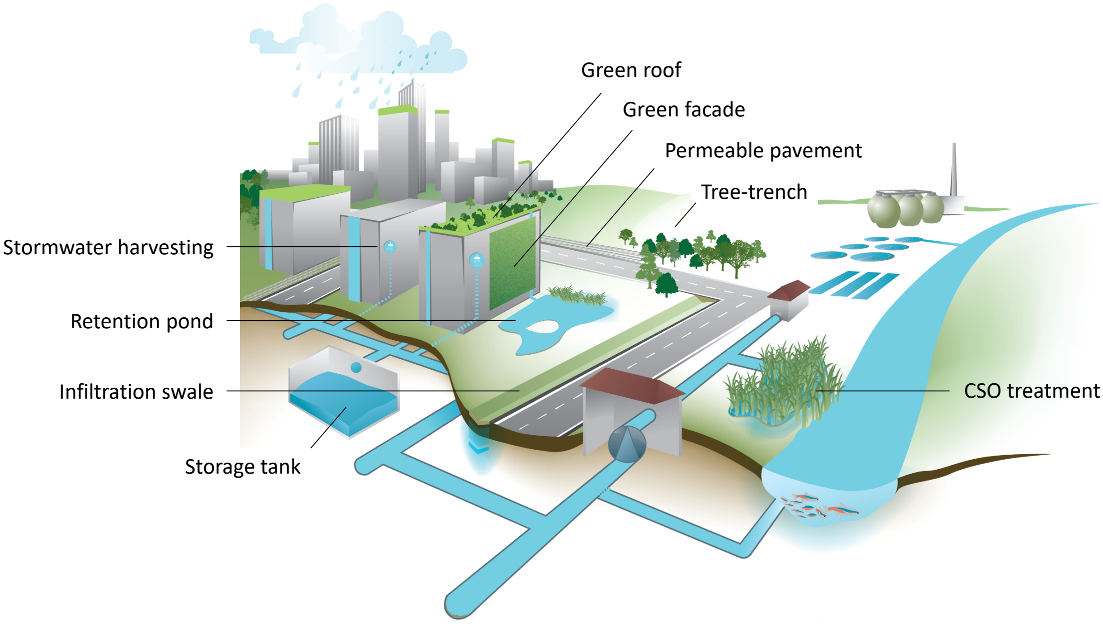

Figure 1.1: Sustainable urban drainage systems and sewer-based CSO control measures, specifically storage tanks and CSO treatment, adopted from Matzinger et al. (2017a). Real-time control and filter drains, described in sections 1.2.1 and 1.2.2, are not shown.

1.2.1 Sewer-based CSO control measures

Storage tanks

The main function of storage tanks, as illustrated in Figure 1.1, or storage tunnels is to retain combined sewerage during high-flow periods and convey it to the wastewater treatment plant in the subsequent low-flow periods. By that, water can be prevented from overflowing into the receiving water body and hydraulic loadings to the wastewater treatment plant can be reduced for smaller rain events.

Storage can be provided online and offline. In an online arrangement, water flows through the tank both in wet and dry weather conditions and is flowing over when a critical water level is exceeded. In an offline arrangement, the tank is only utilised when the upstream water level rises above the level of the inflow crest (Butler and Davies, 2010). In some cases, storage can also be located offshore in the receiving river (Gantner et al., 2008).

Besides volume retention, storage facilities can also provide sedimentation for particulate pollutants and attached substances, e.g. heavy metals and pathogens. Llopart-Mascaró et al. (2015) report removal efficiencies for suspended solids between 45 and 60%.

Real-time control

Real-time control (RTC) is defined as the use of continuously monitored process data to operate a flow-regulating device or, less frequent, a water treatment device in a sewer system (Schütze et al., 2001). A RTC system consists of i) a sensor, e.g. a water level sensor or rain gauge, that monitors the process evolution, ii) a controller, e.g. a proportional-integral-derivative (PID) control mechanism, that converts the sensor signal into a desired action, and iii) an actuator, e.g. a pump or sluice gate, that executes the desired action (Schütze et al., 2004). A typical RTC application is the activation of in-sewer storage during storm events by means of variable weirs or sluice gates, controlled with online water level measurements (e.g. Dirckx et al., 2011; Seggelke et al., 2013; Philippon et al., 2015). A sub-type of RTC is model predictive control where the actuator is optimised on basis of online model predictions and rainfall forecasts, as demonstrated by Lund et al. (2020).

RTC is generally considered as an ecologically and economically efficient way of upgrading existing urban drainage systems (Kroll, 2019). Nonetheless, RTC implementation is not straight-forward and usually requires an elaborate hydrodynamic model as well as laborious trial-and-error or mathematical optimisation techniques to find a suitable strategy (Schütze et al., 2004). In practice, RTC further requires a data transmission system and regular maintenance of sensors and actuators.

Even without designated storage tanks or real-time control solutions the sewer system itself provides a certain retention volume due to a temporary water level increase during storm events. This effect is particularly pronounced in flat sewer systems with large pipe diameters, as is the case in Berlin, and can make up more than one third of the total storage capacity of a combined sewer system (Pawlowsky-Reusing, 2010).

CSO treatment

CSO treatment aims at reducing pollutant loads in combined sewer overflows by means of mechanical or biological techniques. Disinfection can further be applied to reduce pathogen concentrations in CSO discharges.

Screens and sieves are the most widely used mechanical treatment techniques (Butler and Davies, 2010). They remove suspended solids as well as absorbed substances from CSO discharges and can be installed either at the CSO structure or at the river outlet. Removal efficiency for suspended solids varies largely between 25 and 90% (Hanley et al., 2019) and is basically a function of screen spacing or sieve mesh size. One of the main disadvantages is the high maintenance effort related to cleaning the screen or sieve.

Constructed wetlands facilitate biological and mechanical treatment of combined sewage, stormwater, or treated wastewater. They are constructed as soil filters with a detention basin on top of a filter layer which is vegetated with reed plants. Vertical flow systems are also referred to as retention soil filters (Ruppelt et al., 2018). Constructed wetlands remove both particulate and dissolved pollutants and further attenuate hydraulic peaks of CSO discharges. Filtration rate and detention time are usually controlled by an outlet throttle. According to Uhl and Dittmer (2005), removal rates for chemical oxygen demand (COD), ammonium nitrogen (NH_4-N), and total suspended solids (TSS) range between 85 and 99%. Removal rates for pathogens amount to approximately 1 to 2 orders of magnitude, as reported by Waldhoff (2008) for *Escherichia coli* and intestinal enterococci and lately confirmed by Ruppelt et al. (2018) and Tondera et al. (2019). A recent review of different constructed wetland applications and their performance is given by Rizzo et al. (2020). Constructed wetlands are usually operated downstream of a larger drainage system but – as they are constructed aboveground – also comprise some of the benefits typically attributed to SUDS, e.g. cooling effects (Riechel et al., 2017). An illustration of a constructed wetland for CSO treatment is given in Figure 1.1.

Disinfection via ultraviolet (UV) radiation, ozone, or chlorine dioxide is further used to reduce pathogen concentrations in CSO discharges, especially relevant if receiving waters are used for bathing. Removal efficiency of the different disinfection methods depend on the applied dose and exposition time and typically ranges between 2 and 4 orders of magnitude (US EPA, 1999). Main disadvantages, however, are the formation of toxic by-products (in the case of chlorine dioxide), high initial capital costs (in the case of ozone), and the reduced efficiency in combination with high TSS concentrations (in the case of UV radiation).

1.2.2 Sustainable urban drainage systems

Green roofs

Green roofs, as illustrated in Figure 1.1, are vegetative covers of conventional flat or steep roofs that retain and evaporate stormwater runoff and attenuate runoff peaks. They have depths between 8 and 100 cm and consist of a vegetation layer, a substrate or filter layer, and a drainage layer (Riechel et al., 2017). A distinction is made between extensive and intensive green roofs. Extensive green roofs have a thin substrate layer (< 15 cm) and are also suitable for retrofitting due to their low weight. Typical species of the vegetation layer are sedum species and mosses. Intensive green roofs, in contrast, have a thick substrate layer (> 15 cm) and can be used to create versatile roof garden landscapes with trees, paths, ponds, and wetlands. Main positive effects of green roofs are the reduction in hydraulic loadings to sewer systems (Hathaway et al., 2008; Liu and Chui, 2019), an increase in evapotranspiration (Cascone et al., 2019), and the enhancement of urban biodiversity (Francis and Lorimer, 2011; Pille and Saeumel, 2017). Several modelling approaches that simulate the hydrological effects of green roofs have been developed and tested successfully in the past years (e.g. Andrés-Doménech et al., 2018; Hamouz and Muthanna, 2019).

Green facades

Green facades, as illustrated in Figure 1.1, consist of plants that grow up and along walls of buildings to form a green covering. They can be used for stormwater management when watered with roof runoff or similar. A general distinction is made between i) earth-bound vegetation in the form of climbing plants such as wild vine, ivy, or climbing hydrangea, typically supported with a wire framework, and ii) system-bound vegetation where plants grow in suspended planter boxes in modular construction (Köhler, 2008). While earth-bound facade greening is usually irrigated directly from the connected roof via the root zone of the plants, system-bound facade greening requires appropriate and automatic drip irrigation and fertilisation systems (Manso and Castro-Gomes, 2015). The main positive effects of green facades are their value for the urban landscape and their inhabitants (Elsadek et al., 2019), the cooling potential in and outside of the respective building (Koch et al., 2020), and the capability to increase urban biodiversity (Madre et al., 2015). Effects on surface runoff, however, have been scarcely investigated (Riechel et al., 2017).

Stormwater harvesting

Stormwater harvesting, as illustrated in Figure 1.1, is the collection, treatment, and storage of stormwater for non-potable reuse as toilet flushing or irrigation. The stormwater is typically collected from roofs or other less contaminated surfaces, mechanically filtered with sieves, and stored in cisterns, usually underground, where sedimentation is facilitated. The collected and partially treated stormwater is pumped and provided to the individual consumption points via a second distribution network. In case of particularly large or prolonged storm events, excessive volumes are discharged to the sewer network via an internal overflow structure (Palla et al., 2017). Depending on the connected area, the storage capacity of the cistern, and the extraction rate, a portion of the stormwater runoff can be retained (Riechel et al., 2017) and drinking water resources can be conserved, in turn. However, as the cisterns are usually placed underground, no additional benefits for the local climate or biodiversity are observed.

Permeable pavements

Permeable pavements, as illustrated in Figure 1.1, are partially pervious surfaces used in road and footpath construction. A large variety of pavement types exists, e.g. pervious concrete, porous asphalt, paving stones, and grass pavers, which infiltrate part of the stormwater via their joints or gaps. The effect of permeable pavements on runoff dynamics depends on the pavement type, the soil composition, and particularly on the slope of the terrain (Hou et al., 2019). As a consequence, runoff reduction and infiltration potential differ largely between applications (Riechel et al., 2017). Other positive side effects, e.g. on the local climate or biodiversity, are marginal.

Infiltration swales

An infiltration swale, as illustrated in Figure 1.1, is a grassed or sometimes vegetated swale that retains, infiltrates, and evaporates stormwater runoff from streets or other impervious areas. Stormwater runoff is typically conveyed to the swale via drainage pipes or runs off directly from an adjacent area. The water is temporarily stored in the above-ground retention space before it infiltrates into the groundwater via the living soil zone. The top soil passage provides a mechanical and biological treatment. Evaporation capacity depends on retention time and climate conditions, such as temperature, solar radiation, wind, and humidity. In case of extreme rainfall events, part of the water is conveyed to the drainage system via an overflow structure. Infiltration swales retain a large portion of the runoff, enhance biodiversity, and mitigate the urban heat island effect, not only as a result of evapotranspiration but also of the lower heat capacity of natural soils compared to asphalt (Riechel et al., 2017). The implementation of infiltration swales is typically limited by space availability (typically 20% of the connected area) and subsoil permeability (> 7 mm/h). In case of lower space availability or subsoil permeability, infiltration swales can be combined with an additional sub-surface storage reservoir filled with rocks. These so-called trough-trench infiltration systems also convey a relevant part of the stormwater to the sewer system, controlled by an outflow throttle.

Tree-trenches

Tree-trenches or tree-box filters, as illustrated in Figure 1.1, are derivatives of a trough-trench infiltration system with a tree planted inside the trench. The sub-surface storage reservoir facilitates retention of stormwater and increases the availability of water for the tree. Evapotranspiration is significantly increased compared to other infiltration techniques. The soil substrate for tree-trenches should have a high porosity to ensure drainage and aeration of the soil. Planted trees should be able to tolerate periodic surcharge of the root zone. Although tree-trenches have been added to the SUDS portfolio relatively recently, similar techniques, e.g. bioretention cells or rain gardens, have been in use for around two decades now (e.g. Davis et al., 2001; Dietz and Clausen, 2006). Main positive effects of tree-trenches are their important reduction in stormwater runoff and the cooling effect due to shading and enhanced evapotranspiration (Riechel et al., 2017).

Retention ponds

Retention ponds or basins, as illustrated in Figure 1.1, are artificial water bodies created to retain stormwater and to make it visible and perceptible for the public. The stormwater can be collected from roofs or pavements and conveyed to the pond via drainage pipes or open ditches. The inflow is usually treated in neighbouring wetlands or with technical systems, such as micro-sieves or UV disinfection (Riechel et al., 2017), to avoid phytoplankton growth, reduce pathogen concentrations, and ensure amenity. Retention ponds retain and evaporate stormwater and can contribute significantly to landscape quality. They further enhance urban biodiversity and reduce heat stress, especially during the day (Riechel et al., 2017).

Filter drains

Filter drains are road gullies equipped with inlets for mechanical treatment of stormwater runoff. A large variety of technical designs exists, ranging from specially adapted shaft systems to gutter systems that also provide retention of stormwater (Sommer et al., 2016). Furthermore, there are various inserts and filter cartridges for retrofitting existing road drains. Filter drains primarily remove particulate matter and the substances adsorbed on it. Some systems also provide adsorption of dissolved phosphorus or heavy metals and moreover attenuate runoff peaks (Sommer et al., 2016). As filter drains are placed underground, they entail no further environmental, climatic, or aesthetic benefits (Riechel et al., 2017).

1.2.3 Alternative options

Besides sewer-based CSO control measures and SUDS, a number of alternative options are discussed in literature, although with limited practical relevance.

First, the complete separation of stormwater from sanitary sewage by means of a second sewer network is inherently an effective measure to minimise CSO emissions and resulting river impacts. However, it must be considered that contamination via stormwater discharges can partly counterbalance the benefits from CSO reduction if stormwater is not properly treated (Thorndahl et al., 2015). Further, the construction of a completely new infrastructure for stormwater drainage would imply huge costs and might bring along operational problems in the sanitary – previously combined – sewer system, e.g. blockages due to the absence of high-flow events.

Second, relocation of CSO outlets from sensitive to less sensitive discharge locations can also be a viable measure to mitigate river impacts but is costly and may lead to unexpected water quality deficits downstream of the new outlet.

Lastly, aeration of CSO spill water or in-river aeration with atmospheric air or manufactured oxygen gas can help to avoid low levels of dissolved oxygen, even though positive effects are probably spatially and temporally limited. A couple of studies on this topic have been published (e.g. Thibodeaux et al., 1994; Alp and Melching, 2011; Benedetti et al., 2013), although full scale implementations are generally sparse.

1.3 Modelling CSO impacts and mitigation measures

For the simulation of CSO impacts and measure effects from source to target, integrated models of the urban drainage system, the receiving water body, and, in some cases, of the wastewater treatment plant are required. In the following, the underlying simulation approaches, challenges of model integration, and methods for assessing model performance are outlined.

1.3.1 Simulation approaches

Although varying in terms of complexity, simulated variables, and spatial scale, the general simulation concepts of integrated urban water models are similar. In the following, the prevalent approaches for simulating surface runoff, pollutant build-up and wash-off at the catchment's surface as well as flow, pollutant transport, and pollutant conversion in sewers and rivers are described. Figure 1.2 visualises the different model compartments and simulated processes.

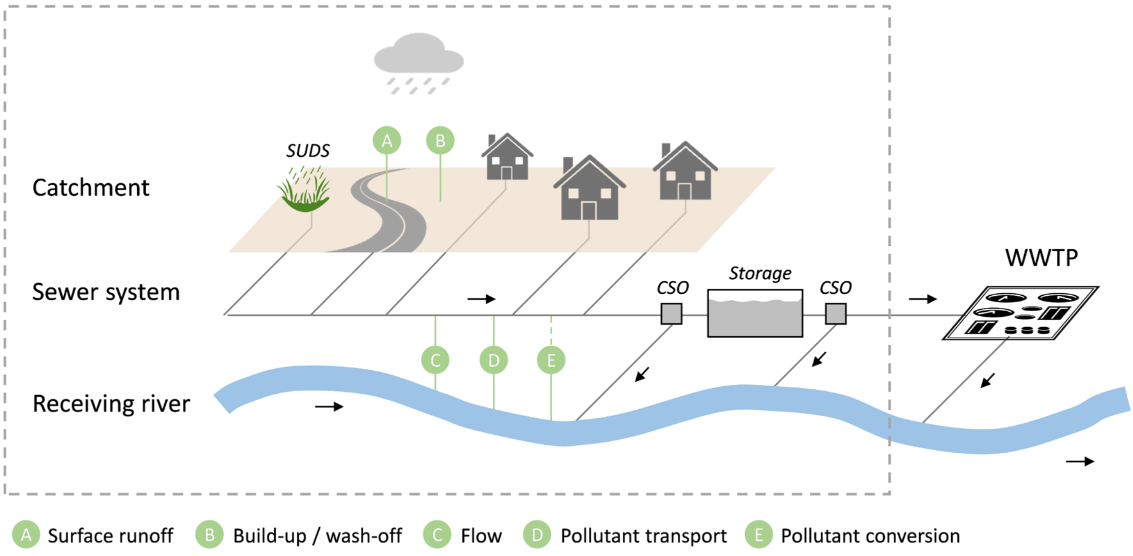

Figure 1.2: Compartments of integrated urban water models and simulated processes (green circled letters). The wastewater treatment plant (WWTP) is located outside the system boundaries (dashed grey line) and therefore not considered in this thesis. Arrows indicate flow direction. [1]

Surface runoff

Surface runoff is the portion of rainfall that runs off from urban surfaces after filling of storages and under consideration of infiltration and evapotranspiration. It is mathematically represented by the continuity equation:

$$\frac{\partial S}{\partial t} = r - q - i - e \qquad \text{(Equation 1.1)}$$

where S [mm] is the storage, t [s] is time, r [mm s^{-1}] is the rainfall rate, q [mm s^{-1}] is the runoff rate, i [mm s^{-1}] is the infiltration rate, and e [mm s^{-1}] is the evapotranspiration rate.

Rainfall, which is the main input of any urban drainage model, is typically derived from rain gauge measurements and distributed over the catchment via, e.g. Thiessen-polygons (Thiessen, 1911). Also radar rainfall data with a higher spatial resolution is frequently used. Storage is usually represented by an initial loss value which accounts for depression and interception storages filled at the beginning of a rain event. In more complex hydrological models, storage in the soil, which depends on soil type and antecedent rainfall conditions, is explicitly simulated.

For the computation of runoff volume two conceptual approaches are widely used. In the runoff coefficient method, as used e.g. in Infoworks (Innovyze, 2017), surface runoff is computed by multiplication of the rainfall volume remaining after filling of storages with a fixed or, in some cases, variable runoff coefficient. In the loss rate method, on the other hand, rainfall up to a

[1] Individual icons represented in the figure were designed by Kimmi Studio ("Rain cloud"), Carlos Dias ("SUDS"), Ataur Rahman ("Street"), Bernd Lakenbrink ("House"), and Daan ("Wastewater treatment plant"), all obtained from Noun Project (https://thenounproject.com).

certain rate infiltrates into the soil or fills up storages, while the exceeding portion of rainfall is regarded as surface runoff.

Runoff routing which determines the form of the runoff curve is often simulated with a linear reservoir model. The linear reservoir model assumes that the flow at the catchment's outlet q is proportional to the storage volume S of the reservoir. The reservoir equation, which complements the continuity equation (Equation 1.1), is formulated as follows:

$$S = k \cdot q \qquad \textit{(Equation 1.2)}$$

with k [s] being the reservoir constant determined during calibration.

As alternatives, also non-linear reservoir models or kinematic or diffusive wave approaches, that solve a simplified momentum equation and explicitly account for the catchment's width, slope, and roughness, are used for runoff routing.

Evapotranspiration, the last component of equation 1.1, can be regarded either as a constant or simulated explicitly after, e.g. Penman-Monteith (Penman, 1948; Monteith, 1965) or Haude (1954). Also infiltration can be simulated explicitly, e.g. according to Horton (1940) or Green and Ampt (1911), which take into account the hydraulic conductivity and initial water content of the soil, among other factors.

In the past decade, modelling approaches for SUDS, which provide detention storage and enhance infiltration and evapotranspiration, have been developed and added to many rainfall-runoff models (Zhou, 2014; Eckart et al., 2017). Data-driven methods, especially different types of neural networks, have also been used increasingly for stormwater runoff prediction and other hydrological questions (see review by Zounemat-Kermani et al., 2020).

Pollutant build-up and wash-off

The most popular approach for simulating pollutant concentrations in stormwater runoff is the build-up/wash-off model. During dry weather, pollutants accumulate on the catchment's surface with a linear, power, exponential, or other function of time, usually limited to a maximum possible build-up. During rainfall, pollutants are washed off at a rate proportional to the pollutant's availability on the surface and linearly or exponentially dependent on the runoff rate. Equations 1.3 and 1.4 show the exponential build-up and wash-off functions based on the original suggestions by Sartor and Boyd (1972):

$$B = B_{max} \cdot (1 - e^{-K_B \cdot t}) \qquad \textit{(Equation 1.3)}$$

$$W = K_W \cdot q^{N_W} \cdot B \qquad \textit{(Equation 1.4)}$$

where B [kg ha^{-1}] is the built-up mass per area, B_{max} [kg ha^{-1}] is the maximum possible build-up per area, K_B [d^{-1}] is the build-up rate constant, t [d] is the time, W [kg ha^{-1} h^{-1}] is the washed-off mass per area and time, K_W [mm^{-1}] is the wash-off coefficient, q [mm h^{-1}] is the runoff rate, and N_W [-] is the wash-off exponent.

The model parameters B_{max}, K_B, K_W, and N_W are usually determined via calibration, which can be a difficult task given the numerous surface types often included. As an alternative to the explicit computation of build-up and wash-off, event mean concentrations derived from regression models that account for rainfall and flow characteristics are sometimes used (e.g. Mourad et al., 2005; Sun and Bertrand-Krajewski, 2012; Dotto et al., 2014). More recently, data-driven methods such as artificial neural networks have also been tested to predict pollutant concentrations in stormwater runoff (e.g. Jeung et al., 2019; Moeini et al., 2021) with relatively good results, given the small number of data sets and input variables.

Flow in sewers and rivers

Flow in combined sewer systems is composed of sanitary sewage from households and industries, stormwater runoff from the catchment's surface, and often a constant base flow originating from groundwater infiltration. Daily variations in sanitary sewage quantity are usually accounted for via hydrographs based on hourly and weekday-specific scaling factors. To represent water flow in gravity sewer pipes and rivers, the Saint-Venant equations are typically applied, which can be derived from a mass and momentum balance for one-dimensional, non-uniform, and unsteady flow in open channels. In many cases, one-dimensional flow simulation can be considered as sufficiently accurate, as vertical or horizontal flow gradients along the pipe or river cross section are often negligible. The Saint-Venant equations are a simplification of the Navier-Stokes equations (Fischer et al., 1979) and consist of a continuity (Equation 1.5) and a momentum equation (Equation 1.6):

$$\frac{\partial A}{\partial t} + \frac{\partial Q}{\partial x} = 0 \qquad \textit{(Equation 1.5)}$$

$$\underbrace{\frac{\partial Q}{\partial t} + \frac{\partial}{\partial x}\left(\frac{Q^2}{A}\right)}_{\text{inertia term}} + \underbrace{g \cdot A \cdot \frac{\partial h}{\partial x}}_{\text{pressure term}} - \underbrace{g \cdot A \cdot \left(S_0\right.}_{\text{gravity term}} \underbrace{\left.- S_f\right)}_{\text{friction term}} = 0 \qquad \textit{(Equation 1.6)}$$

where Q [m³ s^{-1}] is the discharge, A [m^2] is the wetted cross section of the sewer or river, h [m] is the water level, g [m s^{-2}] is the acceleration due to gravity, t [s] is time, x [m] is the distance in flow direction, S_0 [-] is the invert slope and S_f [-] is the friction slope.

The full Saint-Venant equations, also called dynamic wave equations, are implemented in many state-of-the-art sewer and river modelling programs. However, since the computation of the complete Saint-Venant equations is costly and only necessary for abrupt changes in water level or flow, some models make use of further simplifications. A common simplification is the diffusive wave approximation which neglects the inertia term in Equation 1.6 and is applicable for a wide range of cross-sectional geometries. A stronger simplification is the kinematic wave approximation which does not only neglect the inertia term but also the pressure term in Equation 1.6, assuming uniform flow condition. The kinematic wave approximation is valid when changes in water level or flow velocity are negligible compared to the gravitational forces of the invert slope. For these so-called normal flow conditions, the momentum equation is reduced to:

$$S_0 = S_f \qquad \textit{(Equation 1.7)}$$

which assumes that the gravitational forces induced by the invert slope equal the friction losses. The equation can be solved, e.g., after Manning-Strickler:

$$Q = \frac{1}{n} \cdot A \cdot R_h^{2/3} \cdot \sqrt{S_0} \qquad \textit{(Equation 1.8)}$$

where n [s m$^{-1/3}$] is the friction coefficient after Manning and R_h [m] is the hydraulic radius.

The above equations are only valid for free-surface flow and not directly applicable to surcharged or pressurised pipes in sewer networks. To overcome this limitation, the so-called Preissmann slot has been introduced (Cunge and Wegner, 1964), as also implemented in Infoworks (Innovyze, 2017). The Preissmann slot is a virtual slot along the pipe's soffit which allows for a smooth transition between gravity and pressurised flow, and vice versa, while solving the Saint-Venant equations. Alternatives for computing flow in surcharged or pressurised pipes are the Darcy-Weisbach or Hazen-Williams equations, as implemented in the recent version of SWMM (Rossman and Huber, 2017), although not applied in this thesis.

Pollutant transport

Pollutants in the sewer system originate from wash-off from the catchment's surface, dry-weather flow, and resuspended sediments from the sewer pipes. In rivers, pollutants mainly originate from diffuse emissions from agricultural land uses, discharges of urban drainage systems (including CSO, stormwater outlets of the separate sewer system, and wastewater treatment plants), and atmospheric deposition.

Pollutant transport in sewer pipes and rivers is typically simulated with the one-dimensional advection-dispersion equation for a conservative tracer (Equation 1.9), accounting for the two dominant transport processes, advection and dispersion:

$$\frac{\partial C}{\partial t} = -\frac{\partial (v \cdot C)}{\partial x} + \frac{\partial}{\partial x}\left(D_L \cdot \frac{\partial C}{\partial x}\right) \qquad \textit{(Equation 1.9)}$$

where C [mg m^{-3}] is the concentration of the simulated pollutant, t [s] is time, x [m] is the distance in flow direction, v [m s^{-1}] is the flow velocity, and D_L [m^2 s^{-1}] is the longitudinal dispersion coefficient.

The transport-dependent temporal change of a pollutant's concentration in a pipe or river segment (term on the left side of Equation 1.9) is the result of advection and dispersion. The advection term (first term on the right side of Equation 1.9) describes the amount of the substance flowing through the pipe or river segment. The flow velocity v or, respectively, the discharge Q and the wetted cross section area A are outputs from the hydraulic model after solving the flow equations. The dispersion term (second term on the right side of Equation 1.9) describes longitudinal substance dispersion which depends on velocity and concentration gradients between water parcels (Rossman and Huber, 2016). Dispersion usually plays a minor role and is therefore often neglected (Rauch et al., 1998b), although considered in this thesis.

Pollutant conversion

In many urban drainage systems and especially in river ecosystems, pollutants are not only transported with the flow but are also subject to conversion processes. To account for biogeochemical reactions between state variables, reaction terms are added to the right-hand side of Equation 1.9. As a result, transported pollutants are no longer conservative but can undergo physical, chemical, and biological transformations (Rauch et al., 1998a).

Typically, a substance such as dissolved oxygen (DO) interacts with a variety of other state variables, e.g. biological oxygen demand (BOD), heterotrophic bacteria, phytoplankton, and nutrients. The form of the interaction depends on model parameters, which are either determined in field experiments or fitted during model calibration.

One of the first water quality modelling approaches was described by Streeter and Phelps (1925). The approach considers two state variables (DO and BOD) and two processes (biodegradation and atmospheric exchange):

$$\frac{\partial DO}{\partial t} = k_r \cdot (DO_{sat} - DO) - k_d \cdot BOD \cdot e^{-k_d \cdot t} \qquad \textit{(Equation 1.10)}$$

where DO [mg L^{-1}] is the concentration in dissolved oxygen, t [d] is time, BOD [mg L^{-1}] is the biological oxygen demand, k_d [d^{-1}] is the degradation rate, k_r [d^{-1}] is the reaeration rate, and DO_{sat} [mg L^{-1}] is the oxygen saturation (here considered as a constant, neglecting temperature dependency).

Pollutant conversion processes are numerous and can make water quality models very complex, especially when interactions with sediments and different aquatic species are considered. The river water quality model QSim (Kirchesch and Schöl, 1999), as an example, counts more than 40 process equations directly associated with DO which can be grouped into i) photosynthetic production, ii) species respiration, iii) degradation of organic matter, iv) nitrification, and

v) atmospheric exchange, as schematised in Figure 1.3. Model complexity in QSim is particularly high, since three different phytoplankton species (green algae, cyanobacteria, diatoms) and six different fractions of organic carbon (dissolved and particulate fractions, distinguished into easily and fast degradable compounds, as well as monomeric and inert fractions) are simulated.

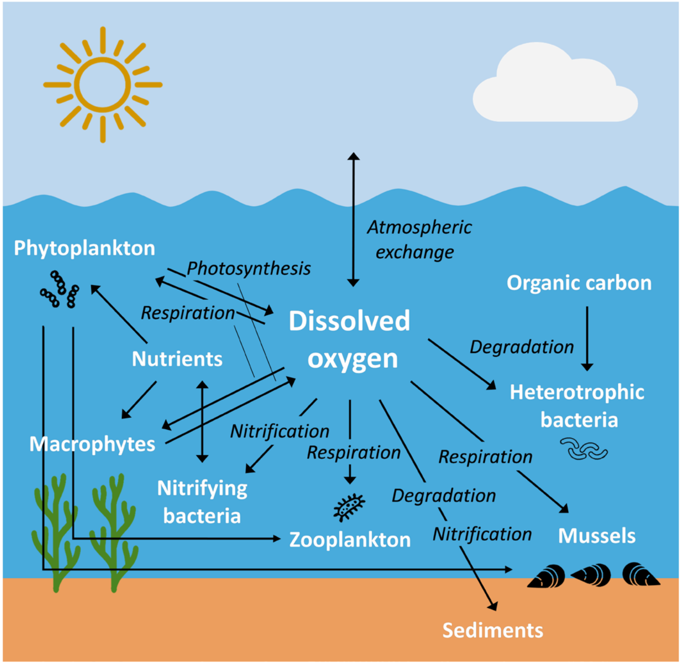

Figure 1.3: Main aquatic processes of the oxygen cycle as implemented in QSim. Note that interactions between phytoplankton, macrophytes, zooplankton, mussels, and organic carbon as well as sedimentation of organic compounds are not shown in detail. [2]

1.3.2 Model integration

Typical applications of integrated models that incorporate the above described simulation concepts are i) the analysis of toxic effects from ammonia (e.g. Holzer and Krebs, 1998; Andrés-Doménech et al., 2010), ii) the simulation of fecal coliforms (e.g. De Marchis et al., 2013), and iii) the assessment of oxygen depletions in the receiving river (e.g. Even et al., 2007; Fu et al., 2009; Moreno-Rodenas et al., 2019). Rauch and Harremoës (1996b), Schütze et al. (1996), and Vanrolleghem et al. (1996) were the first to apply deterministic models to the total system, including the sewer system, the wastewater treatment plant, and the receiving river.

Integrated models are either composed of individual coupled sub-models for the different systems or consist in one integrated simulation environment that combines all. While the first usually represents one-directional flow of information from the upstream to the downstream

[2] Individual icons represented in the figure were designed by Kylie Whittaker ("Phytoplankton", "Zooplankton"), Sarah Mautsch ("Macrophytes"), Pixelz Studio ("Mussels"), Boris Belov ("Heterotrophic bacteria"), Faith Henke ("Sun"), and Nico Tzogalis ("Cloud"), all obtained from Noun Project (https://thenounproject.com).

model, the latter accounts for feedback between the systems, which can be important for the simulation of integrated real-time control strategies.

In the case of individual coupled sub-models, an urban drainage and a river water quality model are required for simulating CSO impacts. State-of-the-art urban drainage models are SWMM (Rossman, 2015), Infoworks ICM (Innovyze, 2017) or MIKE URBAN (DHI, 2019), to name a few examples. These models simulate the above described processes (with the exception of pollutant conversion in the case of Infoworks ICM) and the effect of CSO control measures as storage tanks and real-time control. In addition, model components for SUDS are included in recent software releases. Widely used models for river water quality are Qual2K (Chapra et al., 2012), MIKE ECO Lab (DHI, 2017), RWQM1 (Reichert et al., 2001; Shanahan et al., 2001; Vanrolleghem et al., 2001), and Hydrax-QSim (Kirchesch and Schöl, 1999). All these models simulate biochemical interactions between organic carbon, dissolved oxygen, nutrients, and plankton, but differ in terms of process complexity and their hydraulic simulation approach. A detailed overview is given in Matzinger (2009).

Two of the most popular integrated simulation environments, that combine different water domains and enable to simulate bidirectional interactions, are SIMBA# (Alex et al., 2015) and WEST (Vanhooren et al., 2003). They provide a library of blocks for the sewer system, the wastewater treatment plant, and the river and allow to build integrated models graphically by dragging, dropping, and linking selected blocks. An overview is given in Bach et al. (2014).

Integrated modelling is facing many important challenges that have limited its practical use within the past two decades. One well-known problem is the use of different state variables in the different sub-models (Rauch et al., 2002a). This problem is particularly frequent for carbon and nitrogen compounds and was investigated by Fronteau et al. (1997). The conversion of state variables between models, e.g. from COD to BOD, requires assumptions which contribute to overall uncertainties (Fronteau et al., 1997; Rauch et al., 2002a; Bach et al., 2014). This problem even aggravates when state variables which are essential for the downstream model are completely missing in the upstream model.

Another frequent obstacle in integrated modelling is the excessive model complexity. Often, one of the sub-models simulates processes and variables which are not essential for the investigated impact. This over-complexity results in several drawbacks: First, models become computationally costly, which is particularly relevant if simulations for the different sub-systems have to be conducted simultaneously to account for interactions between systems. Moreover, complex integrated models are hard to calibrate due the high number of model parameters, which in many cases are non-identifiable (Freni et al., 2011). To avoid immoderate complexity of models, only variables and processes with a direct influence on the impact variable should be simulated (Rauch et al., 1998a).

Beyond that, the different components of integrated models may simulate at different spatial or temporal scales, which would require interpolation, aggregation, or upscaling of information between models. Besides the resulting need for additional data processing routines, the different spatio-temporal scales can contribute to overall model uncertainty (Moreno-Rodenas et al., 2019). An attempt to harmonise data exchange between model components over different space-time scales is the certified standard OpenMI (Gregersen et al., 2007), although it did not prevail in past years.

Lastly, integrated models require measurement data at a high temporal resolution for calibration-validation and as model input. Comprehensive monitoring campaigns across several compartments of the entire system are expensive and hence not always conducted adequately. In turn, missing data further contribute to model uncertainties (Moreno-Rodenas et al., 2019).

1.3.3 Assessment of model performance

An important task in integrated modelling is the thorough calibration and validation of the different sub-models (from upstream to downstream), as uncertainties may add up along the model chain (Moreno-Rodenas et al., 2019; Tscheikner-Gratl et al., 2019). For the assessment of model performance throughout the calibration and validation process, two types of approaches are typically used: i) qualitative assessment based on graphical methods and ii) quantitative criteria based on goodness-of-fit or error index statistics.

Qualitative assessment of model performance relies on graphical techniques such as time series plots, residual plots, or x-y-scatter plots. These plots give a good, although subjective impression of the agreement between simulated and measured data and indicate potential bias or systematic variance in model results (Moriasi et al., 2015). Moreover, they reveal strengths and weaknesses regarding the representation of extreme values, e.g. peak flow or minimum DO concentrations, and help to detect temporal anomalies, e.g. daily or seasonal variations in model accuracy.

Quantitative criteria, on the other hand, provide an objective measure of accuracy that allows to compare different models or calibrations. Compared to qualitative visualisation techniques, they reduce the risk of misjudgement and may serve as an objective function in automatic calibration or model selection processes (Bennett et al., 2013). Two of the most popular goodness-of-fit indicators in hydrological and water quality modelling are the Nash-Sutcliffe efficiency (NSE; Nash and Sutcliffe, 1970) and the root mean square error (RMSE) (see chapter 3.2.5 for further details and formulas).

Regarding its timescale, both the qualitative and quantitative assessment of model performance should focus on the events of interest, e.g. the periods impacted by CSO. Choosing the wrong calibration or validation period may lead to too optimistic or pessimistic results with regards to the simulation goal (Motavita et al., 2019). Nonetheless, even in the case of event-based calibration and validation procedures, quantitative criteria and graphical methods may lead to misinterpretations. For instance, a systematic delay of simulated effects will lead to low goodness-of-fit indicators and large residuals in scatter plots, even though the duration and extent of the effects are correctly simulated and the model may be perfectly valid for decision making. To overcome this gap, impact-based assessment criteria applied to the model output and measurement data are introduced in this thesis as a supplement to the above mentioned qualitative and quantitative methods.

1.4 Identified gaps and scope of this thesis

1.4.1 Identified gaps

A number of deficits and research gaps have been identified which are addressed in this thesis and specified in the following.

Impact-based modelling for strategic planning

Several integrated modelling studies have been conducted in the past that allow to simulate interactions between two or more urban water compartments and the specific impacts of CSO (e.g. Holzer and Krebs, 1998; Fu et al., 2009; Motta et al., 2010). However, many of these studies have shown some important shortcomings when it comes to strategic planning and impact mitigation. First, full-scale demonstrations for complex systems with a multitude of CSO outlets and a network of receiving channels and rivers, as is the case of Berlin, are sparse. Second, modelling studies are often based on measurement data of poor spatial and temporal resolution which attributes to high uncertainties, in particular for the downstream model components. Lastly, assessment of model performance does usually not account for the frequency or duration of adverse conditions in the river. Consequently, the practical use of integrated models for impact-based decision making and strategic planning in large urban areas is not yet fully demonstrated.

Process understanding

The degradation of organic matter, either in the water column or delayed in the sediments, as the main driver for oxygen deficits in the river after CSO is well documented (e.g. Harremoës, 1982; Hvitved-Jacobsen, 1982). However, other physical or biogeochemical processes that additionally affect the oxygen budget after CSO are scarcely addressed in literature. A better understanding of processes and their dynamics would not only allow to allocate selective mitigation measures and so maximise benefits for the river. The enhanced knowledge would also help in the selection of processes and variables for other integrated modelling studies that aim to simulate CSO impacts on dissolved oxygen.

Effects of mitigation measures

Different authors have highlighted the effects of SUDS on runoff dynamics and also on pollutant loads in stormwater runoff (e.g. Hathaway et al., 2008; Hatt et al., 2009; Drake et al., 2014). Further, SUDS effects on CSO emissions have also been investigated (e.g. Joshi et al., 2021), although in limited number and often disregarding the spatially diverse feasibility and dynamic nature of SUDS. Nevertheless, modelling studies on the potential of realistic SUDS strategies to mitigate river impacts from CSO are completely lacking. A major barrier for the implementation of integrated SUDS-CSO-river models are the diverging spatial scales between SUDS (building or city quarter scale) and the receiving river (catchment or watershed scale). In this context, the required model complexity for the representation of SUDS effects remains an unanswered question.

Identification of pathogen hotspots

Different deterministic modelling approaches for microbiological contaminants in urban drainage systems have been developed (e.g.De Marchis et al., 2013; Huang et al., 2015). However, these approaches are often unnecessarily complex when the main objective is not the simulation of temporarily discrete pathogen concentrations but the general identification of pathogen emission hotspots. Simple tools for the prioritisation of CSO outlets in terms of their hygienic relevance and thus for the allocation of effective mitigation measures are missing. This gap is especially noticeable in large and complex sewer systems with a multitude of CSO outlets and high spatial variance in pathogen loads.

1.4.2 Scope of this thesis

In this thesis, different setups of an integrated model are used to improve the understanding of CSO related river processes and close research gaps related to the effects of SUDS and other mitigation measures on the ecological status of urban rivers. In addition, a novel method to determine the relevance of different CSO outlets for downstream bathing water quality is developed and tested at city scale. In this context, answers to the following research questions are given:

- How and at which accuracy can CSO emissions, resulting river impacts, and measure effects be simulated for a complex urban water system?
- Which are the relevant biogeochemical river processes that lead to oxygen depressions after CSO?
- What is the general potential of different mitigation strategies and which are the specific effects of SUDS on river impacts after CSO?
- To which extent will the positive effects of mitigation measures be negated by global climate change?
- How can the hygienic relevance of different CSO outlets in large combined sewer systems be quantified?

The thesis is structured into eight chapters, consisting of the introduction (current **Chapter 1**), two peer-reviewed journal articles, two reviewed book chapters, one peer-reviewed conference paper, a synthesis, and a summary of further related work.

Chapter 2 introduces an integrated modelling approach consisting of the urban drainage model Infoworks CS, the river water quality model Hydrax-QSim, and an impact assessment approach for fish-critical DO conditions in the river. The modelling approach allows to investigate river impacts from CSO and quantify the effect of different mitigation measures. Further, a model validation approach based on impact assessment criteria is presented which allows to highlight deficits in model performance that would potentially remain undiscovered by conventional methods.

Chapter 3 demonstrates the developed modelling approach for a major part of the combined sewer system of Berlin, Germany, and proves the applicability for strategic planning. After a thorough model validation with continuous measurement data from various river sections, sensitivity of the model to changing boundary conditions and model parameters is shown. The validated model is used to analyse the dynamic nature of different river processes that lead to oxygen stress in the receiving river and quantify CSO impacts for different mitigation strategies.

In **Chapter 4** the developed modelling approach is tested for selected climate change scenarios, considering projected temperature increase and changes in rainfall intensity. Climate change impacts on background levels of DO during dry weather are distinguished from highly critical impacts after CSO. It is further quantified to which extent the positive effect of planned mitigation measures will be counterbalanced by climate change. The results can be partly transferred to local change scenarios, e.g. an increase in impervious area.

Chapter 5 explores the effects of SUDS strategies, developed for an established city quarter, on surface runoff, CSO emissions, and river impacts by adding a distinct rainfall-runoff model with detailed SUDS representation to the developed model tool. The detailed simulation approach for SUDS is compared with a classical approach consisting in global runoff reduction scenarios. The barrier of diverging spatial scales between SUDS and the receiving river is tackled with a simple yet powerful upscaling method that extrapolates SUDS effects from a specific city quarter to the entire catchment.

Chapter 6 presents a simple tracer approach to quantify the microbiological contamination potential of drainage systems with a high number of CSO outlets. The tracer approach uses the quantity of wastewater in CSO discharges as a proxy for pathogen emissions and is tested for the entire city of Berlin, Germany, with 176 CSO outlets. Given the large differences in pathogen concentrations of wastewater and stormwater, the approach can determine emission hotspots responsible for hygienic deficits in downstream bathing water quality.

Chapter 7 synthesizes the main outcomes of the thesis and gives recommendations for stakeholders. Moreover, limitations of the presented methods and results are discussed and an outlook on potential future research is given.

An overview of supplementary scientific work related to this thesis, including several first-author contributions to national and international conferences, is presented in **Chapter 8**.

Chapter 2: Setup and validation of a model tool for CSO impact assessment

This study was published as:

Riechel, M., Matzinger, A., Sonnenberg, H., Caradot, N., Meier, I., Heinzmann, B., Rouault, P. (2012). Validation and sensitivity of a coupled model tool for CSO impact assessment in Berlin, pp. 8. *Proceedings of the 6th International Congress on Environmental Modelling and Software (iEMSs)*. Leipzig, Germany, 2012.

This is the postprint version of the article.

Abstract

In the city of Berlin combined sewer overflows (CSO) can lead to severe depressions in dissolved oxygen (DO) of receiving urban rivers and hence to acute stress for the local fish fauna. To quantify CSO impacts and optimize sewer management strategies, a model-based planning instrument has been developed. It couples the urban drainage model InfoWorks CS which simulates hydraulics and pollutant transport in the sewer with the river water quality model QSim which simulates hydraulics, mass transport and various biogeochemical processes in the receiving water body. To identify simulated CSO impacts, concentration-duration-frequency thresholds for DO are applied to river model results via an impact assessment tool. Two kinds of impacts are distinguished: i) suboptimal conditions and ii) critical conditions for which acute fish kills are possible. In the case of Berlin, suboptimal conditions are observed on up to 92 days per year, predominantly during periods of low discharge and high temperatures whereas critical conditions only occur after CSO. For model calibration and validation, continuous measurements in both river and sewer are used. First simulations show good accordance between simulated and measured DO concentration in the river with Nash-Sutcliffe efficiencies between 0.70 and 0.79 for an eight-month time period at three different river monitoring points. However, to assure satisfactory model performance for adverse DO conditions in particular, impact assessment results for measured and simulated data are compared. Regarding suboptimal DO conditions simulated and measured data show good agreement. Nevertheless, model representation for critical conditions is poor for some river sections and requires further improvement for CSO conditions. The results underline the importance of combining different validation approaches when dealing with complex systems.

2.1 Introduction

In the city of Berlin regular combined sewer overflows (CSO) lead to acute stress of aquatic organisms in the receiving River Spree and its side channels. Of most concern is the occurrence of depressions in dissolved oxygen (DO), which has been acknowledged as a major issue in the Berlin inventory for the EU Water Framework Directive (SenStadt, 2004).

To assess the impacts of CSO on the Berlin River Spree a model-based planning instrument has been developed. It will be used by decision makers to study different sewer management and climate change scenarios. The planning instrument couples

- the sewer model InfoWorks CS (WSL, 2004), which simulates volumes and substance loads of CSO,
- the river water quality model Hydrax/QSim (Kirchesch and Schöl, 1999; Oppermann, 2010) which simulates the effects of these CSO on hydraulics and water quality in the receiving water body and
- an impact assessment tool which evaluates and quantifies the simulated effects of CSO on the receiving river.

Before employing the coupled model-tool for prediction of CSO impacts in the river, calibration and validation has to be conducted. Both procedures are particularly important when different models are coupled and uncertainties on process variables, input data or model structure may add up. The assessment of the model performance is a fundamental part of calibration and validation, typically basing on two approaches: i) the subjective (or qualitative) evaluation of the model behaviour via visual inspection of simulated and observed hydrographs and ii) the objective (or quantitative) evaluation by means of mathematical error estimation methods that quantify the deviation between model results and measurements (Krause et al., 2005).

However, both methods might be insufficient in the case of the above described planning instrument for CSO control. For instance, river model performance regarding state-variable DO might be poor in visual terms, e.g. due to a delayed simulation of the resulting DO sag after CSO, but nonetheless simulations and measurements agree on the frequency of adverse DO conditions. On the other hand, model efficiency according to objective goodness-of-fit measures, e.g. the coefficient of determination r^2 or the Nash-Sutcliffe efficiency (Nash and Sutcliffe, 1970) could be identified as satisfactory over the simulated time period, even though critically low DO values after single CSO events are not simulated properly. As a consequence, the use of impact assessment parameters, such as the number of days with suboptimal or critical DO conditions, is introduced as a supplementary step for model validation throughout this paper.

First applications of the planning instrument aim at testing the sensitivity of the model output to changing boundary conditions. Different sewer management solutions such as the increase of storage volume, the reduction of the impervious area, decentralized treatment of storm water runoff or end of pipe treatment of overflow water will be combined with expected climate change effects like temperature rise or change in rainfall intensity and analysed with the coupled model tool. Once the simulated CSO impacts in the river fit well with the observations, the model tool can support decision makers in finding appropriate sewer management strategies or analyse the effect of future climate change.

2.2 Study site

The combined sewer system of the city of Berlin covers a drained area of approximately 100 km² and collects storm water and sewage of 1.5 million inhabitants via a sewer network of 2,000 km length. After intense rainfalls the 180 CSO outlets located along the River Spree and its side channels can become a major source of pollution. According to estimations of the Berlin water utilities 20 to 30 CSO events per year are counted on average. However, CSO frequency strongly depends on the spatial and temporal distribution of rainfall and on the specific properties of each sub-catchment (e.g. storage volume, size of drainage area, runoff coefficient).

The receiving River Spree is a flow-regulated lowland river of 50 to 70 m width and 2 to 3 m depth. It has an average monthly discharge between 12 and 45 $m^3 s^{-1}$ (time period: 2000 to 2010) with flow velocities between 6 and 24 cm s^{-1}. The Berlin section of the River Spree is strongly affected by various human activities leading to a highly degraded river morphology, homogeneous river substrates and reduced biodiversity (Leszinski et al., 2007). Periodic CSO events implicate additional stress to the ecology of the urban water courses.

The simulated stretch of the River Spree and its side channels has a total length of 27 km and receives storm water and sewage from 67 CSO outlets each of which is represented in both the urban drainage model and the river water quality model (Figure 2.1).

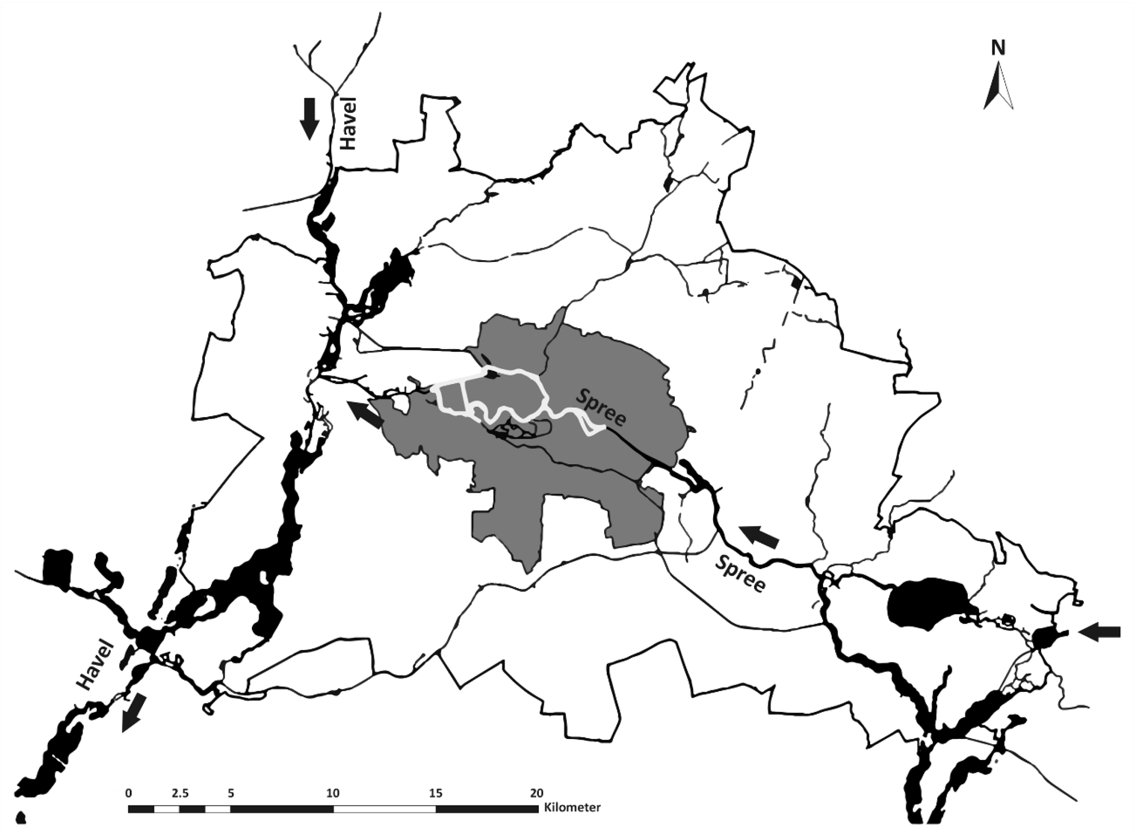

Figure 2.1: Map of the city of Berlin with its main waterways (black), the combined sewer area (dark grey area) and the simulated river stretch (light grey lines).

2.3 Material and methods

2.3.1 Dynamic models

Since the Berlin sewer system is highly complex with little slope at most reaches, flow in the sewer is simulated hydrodynamically with the software package InfoWorks CS (WSL, 2004). InfoWorks CS solves the full St. Venant equations and thus accounts for backwater effects and reverse flow, both of which occur in the Berlin sewer system. In terms of pollutants it simulates the transport of dissolved and solid fractions of biochemical oxygen demand (BOD), chemical oxygen demand (COD), ammonium (NH_4), total Kjeldahl nitrogen (TKN), total suspended solids (TSS), total phosphorus (TP) and ortho-phosphate (PO_4). Degradation processes are not considered assuming that the travel time in sewers is too short for significant decay of constituents. The Berlin model is calibrated and validated based on measurements taken with online probes in a major overflow sewer (Caradot et al., 2011).

For the simulation of DO dynamics in the river, the hydraulic model Hydrax (Oppermann, 2010) with the coupled complex water quality model QSim (Kirchesch and Schöl, 1999), developed at the German Federal Institute of Hydrology (BfG) were chosen. The hydraulic model Hydrax solves the full St. Venant equations and allows simulating various special features that affect river hydraulics, such as macrophyte cover or spur dykes. The water quality model QSim calculates onedimensional transport and reactions of all major water quality parameters, covering a great number of biological parameters, including both planktonic forms that move with the water (green algae, diatoms, cyanobacteria and rotifers) and sessile species (benthic algae, macrophytes and filter feeders) (Schöl et al., 1999; Schöl et al., 2002). For model calibration and validation long-term continuous measurements at several river points are considered.

For the link between the sewer and the water quality model, each of the 67 CSO outlets simulated with InfoWorks CS is represented by a separate inflow in Hydrax/QSim to resolve the spatial distribution of CSO impacts along the river. Although the sewer model simulates the most critical parameters for DO depressions in the river (e.g. BOD, COD, TSS, NH_4) it does not cover all state variables of QSim. Accordingly, assumptions have been made on parameters such as phytoplankton (which is probably close to zero in the sewer) or DO (which is assumed to be 0 mg L^{-1} in CSO).

2.3.2 CSO impact assessment

Regarding impact assessment, the quality standards proposed by Lammersen (1997) for lowland rivers are applied to measured and simulated water quality data of the River Spree. The Lammersen-approach defines DO thresholds for eight exposure durations ranging from 10 minutes to 24 hours (Figure 2.2). For each threshold temperature-dependent correction factors are used, since the oxygen demand of aquatic organisms depends on the water temperature (Downing and Merkens, 1957). The protocol aims at protecting fish and invertebrates from any adverse effect ranging from impairment of swimming behaviour to death. Throughout this paper, a time period is classified as an event with suboptimal conditions, when at least one of the eight concentration-duration-criteria is met.

In addition to these suboptimal conditions, the LC50-value of the asp (*aspius aspius*), the fish species of the River Spree that is most sensitive to low DO, was chosen as a second quality standard for highly critical conditions. If DO concentration remains below 2 mg L^{-1} for more than 30 minutes, lethal impacts on aquatic organisms have to be expected. This approach follows Buikema and Benfield (1980) who suggest to focus on the requirements of the least tolerant species, to protect the local biocenosis as a whole. Figure 2.2 illustrates the quality standards for both suboptimal and critical conditions.

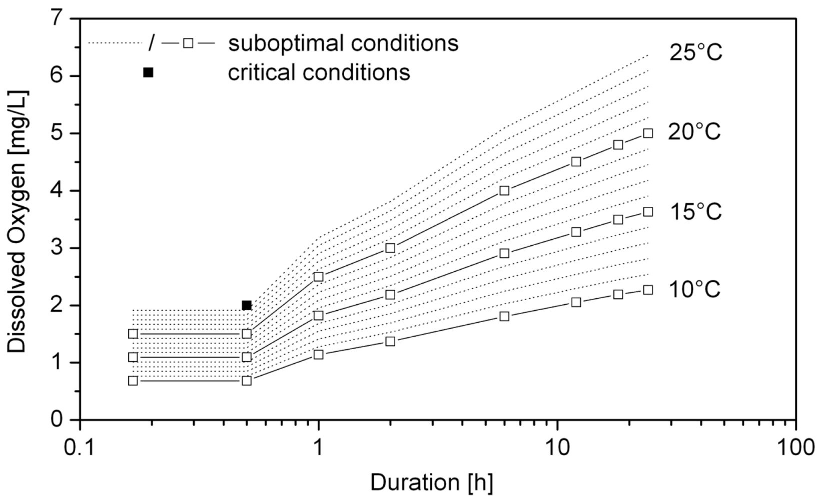

Figure 2.2: Quality standards for CSO impact assessment. Thresholds for suboptimal conditions are adapted from Lammersen (1997) and are defined as a function of duration and temperature. The threshold for critical conditions refers to the LC50-value of the asp (aspius aspius).

To provide a semiautomatic and standardised tool for the evaluation of model results a database application for CSO impact assessment has been developed. The application provides a graphical interface allowing the user to select the simulated DO time series of a certain scenario. The chosen time series is then analysed by successively comparing it to the quality standards described above. Results are provided in terms of tables and graphs, displaying the number of events and calendar days with suboptimal or critical conditions at different points in the River Spree. The impact assessment tool was established primarily for comparison of scenarios regarding their impact in the river. However, it also allows validation of model results with respect to the occurrence of suboptimal/critical DO conditions.

2.4 Results and discussion

Coupling of the sewer and the river water quality model has been successfully demonstrated for the eight-month time period April to November 2010.

2.4.1 Sewer model

In the time period April to November 2010 a total CSO volume of 2.9 million m³ was simulated to have entered the River Spree. Half of that volume was discharged during only five CSO events. For an exemplary storm event on 23 July 2010 sewer simulations with InfoWorks CS identified an overflow volume of 4,480 m³ and pollutant loadings of 51 kg, 251 kg and 811 kg for BOD, COD and TSS at a monitored CSO outlet at km 9.7 of the modelled river stretch. Figure 2.3 shows the rain intensity and the simulated and measured water flow (left panel) as well as the simulated and measured loadings of TSS (right panel) at the observed outlet.

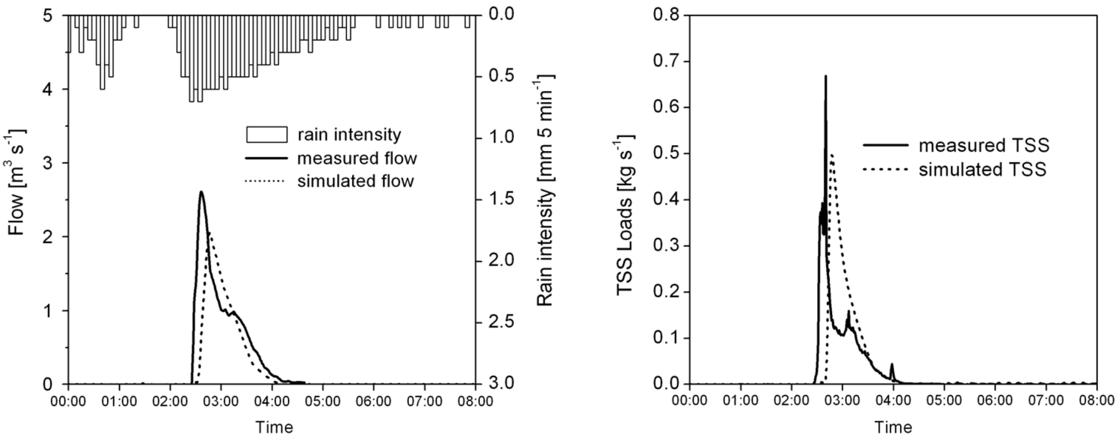

Figure 2.3: Measured and simulated water flow (left panel) and loadings of total suspended solids (TSS) (right panel) at a major overflow sewer during a storm event on 23 July 2010 with a total rain height of 22.7 mm.

2.4.2 Coupled sewer and river model

The seasonal pattern of goal state variable DO is well represented by the coupled model for the entire simulated time period (Figure 2.4, left panel). Quantitative evaluation of model performance yielded Nash-Sutcliffe-efficiencies between 0.70 and 0.79 at different monitoring stations. Looking at the specific CSO event shown in Figure 2.3, the river water quality model QSim shows a significant drop in dissolved oxygen to 3.5 mg L^{-1} at km 10.3 of the modelled river stretch 600 m downstream of the CSO structure (Figure 2.4, right panel, dotted line). However, visual inspection shows that DO depression in the River Spree is not simulated to the same extent as indicated by measurements. In particular, the lowest DO occurrence is not met by the model. Thus, satisfactory Nash-Sutcliffe-efficiencies obtained for the eight-month simulation period described above do not necessarily imply that all observed DO depressions can be properly predicted with the model tool.

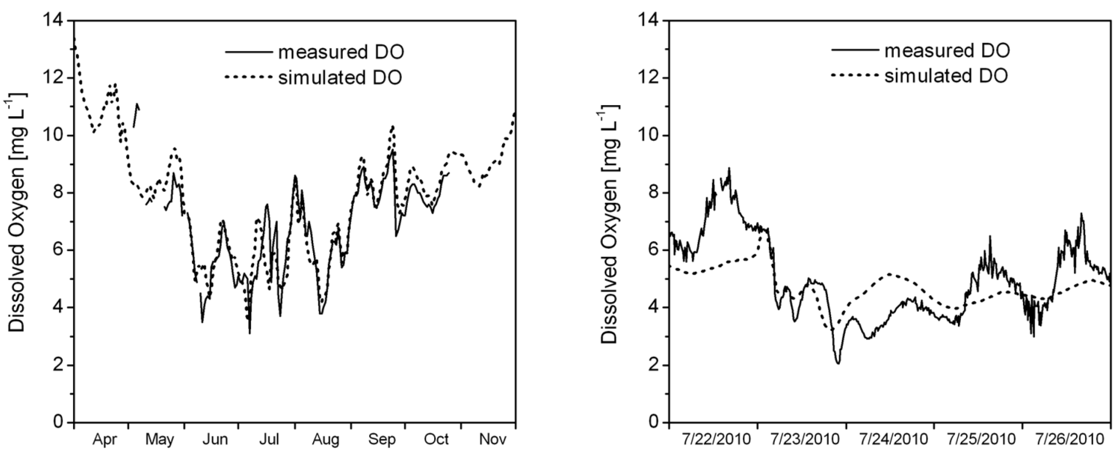

Figure 2.4: Measured and simulated concentrations of dissolved oxygen (DO) at a CSO affected river point. The left panel shows daily averages for the eight-month period April to November 2010. The right panel shows 15-min-values during a storm event on 23 July 2010.

2.4.3 CSO Impact assessment

Applying the two impact assessment approaches described above to measured and simulated data of the year 2010 shows good agreement regarding encountered suboptimal DO conditions (Figure 2.5, upper panel). According to simulations, DO concentration has been low enough to potentially harm aquatic organisms on 50 to 54 calendar days, whereas measurements indicate between 34 and 50 calendar days with suboptimal conditions depending on the river section looked at. However, the coupled model tool shows significant drawbacks when it comes to representation of critical DO conditions (DO < 2 mg L^{-1} for ≥ 30 minutes). While such conditions are measured on up to 4 days at a highly CSO-influenced river section in the city centre (km 4.8), simulations show no critical conditions at all (Figure 2.5, lower panel), indicating insufficient model representation of low DO concentrations. Since the occurrence of DO concentrations below 2 mg L^{-1} is the main reason for occasional fish kills in the Berlin River Spree, further improvement is necessary to obtain a reliable instrument for CSO control.

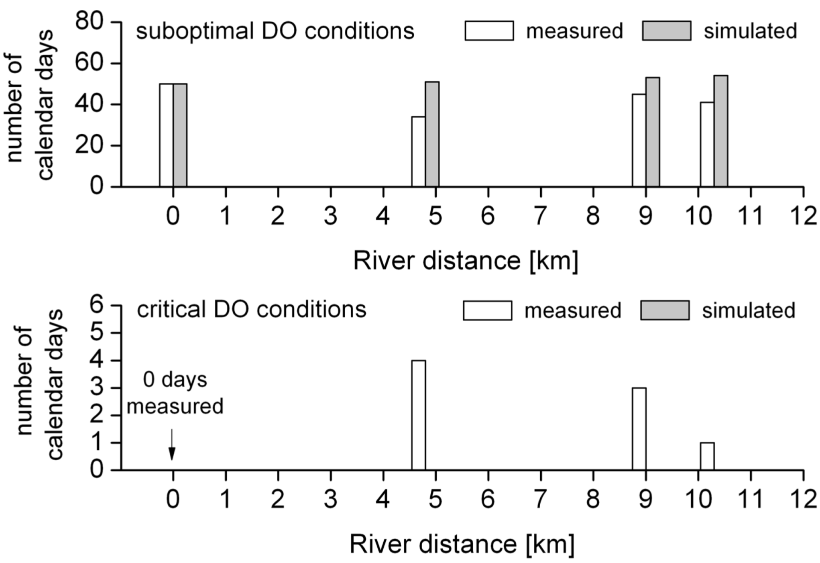

Figure 2.5: Quantification of suboptimal (upper panel) and critical DO conditions (lower panel) at four locations along the river (km 0, km 4.8, km 9.0, km 10.3) according to measured and simulated data for 2010.

2.4.4 Model sensitivity

Both the sewer and the river water quality model have been successfully calibrated to dry weather conditions. Currently, improved calibration to CSO conditions is conducted for both models. When calibration aims at adapting urban drainage parameters, it has to be assured that not only the sewer but also the river water quality model reacts sensitively regarding goal state variable DO. Sensitive behaviour to changing model input is not only fundamental when calibrating a model. It is also the basic prerequisite for using the model tool for the comparison of different sewer management and climate change scenarios. Again, the sensitivity of simulated impacts in the river to changes in boundary conditions should be assessed by looking at both DO concentration and the number of days with suboptimal or critical conditions.

First results on the sensitivity of goal variable DO to CSO pollutant concentrations show that for the tested range of input values the minimum DO concentration simulated in the river nearly depends linearly on the biochemical oxygen demand (BOD) simulated in spill water (Figure 2.6, left panel). Changes in river model input via adaptation of the sewer model can improve representation of CSO impacts in the river. For the studied storm event on 23 July 2010, a BOD increase of 100 % (e.g. due to sewer model calibration) would lead to a decrease of the lowest DO concentration by 40 % (Figure 6, right panel, dotted line). Hence, DO in the river would remain below 2 mg L^{-1} for more than 30 minutes, better agreeing with measurements on the occurrence of critical DO conditions.

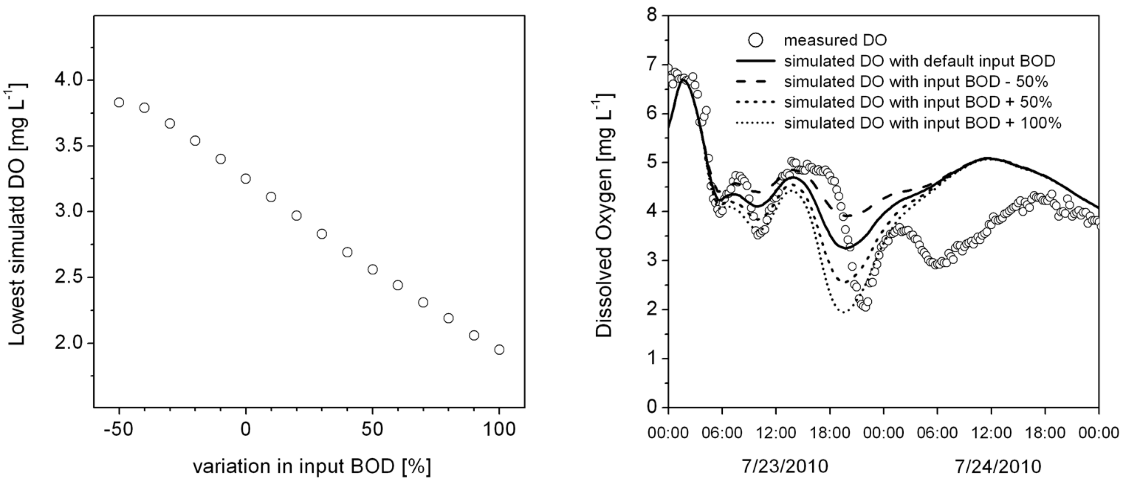

Figure 2.6: Sensitivity of dissolved oxygen (DO) in the river to changing biochemical oxygen demand (BOD) in CSO for a storm event on 23 July 2010. Left panel shows the lowest resulting DO concentration for different BOD input concentrations. Right panel shows measured and simulated DO time series for four different BOD input concentrations.

2.5 Conclusions

The results indicate that the simulation of CSO impacts in receiving rivers is possible at good quality through the coupling of a sewer model with a river water quality model. Even for complex urban water systems such as the Berlin River Spree, highly interdependent state variables such as DO can be simulated at high temporal resolution. Classical validation of model results by visual evaluation and the use of conventional goodness-of-fit measures demonstrated good agreement with DO concentrations measured in the river. In contrast, application of legislative goal functions revealed shortcomings in the performance of the model tool. The four most severe DO depressions observed in the studied 8-month time period could not be represented by the model tool. While these four depressions were rather short with a total duration of only 24.5 hours, they are expected to be most critical for the river ecosystem. Since the improvement of the ecological quality is the main objective of the future use of the coupled model tool, further development is necessary regarding the representation of such situations.

We conclude that coupled model tools can be valuable instruments for decision makers to find appropriate strategies for CSO control or to analyse the expected impact of climate change on the river. However, for validation of model results it is crucial to compare model output and measurements with regards to legislative goals in addition to conventional validation methods. By that it can be provided that the planning instrument is able to meet the specific requirements of CSO representation. Once the model tool is validated, its sensitivity to realistic changes in boundary conditions such as the increase of storage volume or the reduction of the impervious area needs be tested. The analysed change scenarios should be developed in close cooperation with local decision makers to provide that only realistic solutions or effects will be considered already at the testing stage. Only after validation with legislative goal functions and sensitivity analysis for feasible change scenarios, the model tool should be used to support specific planning.

Acknowledgments

The presented work is a result of the KWB research project MIA-CSO and the EU FP7 project PREPARED. Modelling and monitoring efforts in Berlin are financially supported by Veolia Water, Berliner Wasserbetriebe and the Berlin Senate Department for Urban Development and the Environment. We kindly thank Volker Kirchesch and his colleagues at the German Federal Institute for Hydrology (BfG) for the constant development and support regarding the modelling software QSim.

Chapter 3: Understanding river processes and mitigation strategies

This study was published as:

Riechel, M., Matzinger, A., Pawlowsky-Reusing, E., Sonnenberg, H., Uldack, M., Heinzmann, B., Caradot, N., v. Seggern, D., Rouault, P. (2016). Impacts of combined sewer overflows on a large urban river – understanding the effect of different management strategies. *Water Research* 105, 264-273. https://doi.org/10.1016/j.watres.2016.08.017

This is the postprint version of the article. The final publication is available at SienceDirect via https://doi.org/10.1016/j.watres.2016.08.017.

Abstract

To support decision makers in planning effective combined sewer overflow (CSO) management strategies, an integrated modelling and impact assessment approach has been developed and applied for a large urban area in Berlin, Germany. It consists of an urban drainage model, a river water quality model and a tool for the quantification of adverse dissolved oxygen (DO) conditions in the river, one of the main stressors for urban lowland rivers. The coupled model was calibrated successfully with average Nash-Sutcliffe-efficiencies for DO in the river of 0.61 and 0.70 for two validation years. Moreover, the whole range of observed DO concentrations after CSO down to 0 mg L^{-1} is simulated by the model. A local sensitivity analysis revealed that in the absence of CSO dissolved oxygen principally depends on phytoplankton dynamics. Regarding CSO impacts, it was shown that 97% of the observed DO deficit can be explained by the three processes (i) mixing of river water with CSO spill water poor in DO, (ii) reduced phytoplankton activity due to CSO-induced turbidity and (iii) degradation of organic matter by heterotrophic bacteria. As expected, process (iii) turned out to be the most important one. However, depending on the time lag after CSO the other processes can become dominant. Given the different involved processes, we found that different mitigation schemes tested in a scenario analysis can reduce the occurrence of critical DO deficits in the river by 30-70%. Overall, the study demonstrates that integrated sewer-river-models can be set up to represent CSO impacts under complex urban conditions. However, a significant effort in monitoring and modelling is a requisite for achieving reliable results.

3.1 Introduction

Diffuse urban pollution, in particular from combined sewer overflows (CSO), can seriously impair the ecological quality of urban rivers (e.g. Chambers et al., 1997; Marsalek, 2003). Of particular concern are short-term impacts (Table 3.1), which can lead to acute stress or even lethal conditions for aquatic organisms (Harremoës, 1982; Whitelaw and De Solbe, 1989; Newcombe and Jensen, 1996; Rauch and Harremoës, 1996a). Current legislation in Europe or North America requires mitigation measures to reduce these impacts from a river perspective (e.g. US EPA, 1994; EU, 2000). The river perspective makes mitigation requirements case-specific, since type and extent of CSO impacts depend on the local situation (Table 3.1).

Table 3.1: Short-term CSO impacts, expected in different river types, adapted from Matzinger et al. (2012a).

Impacts	**small streams** [a]	**medium to large streams** [b]	**regulated, slow-flowing lowland rivers** [c]	**Available technical guidelines by country**
Hydraulic stress	✓			Germany (Borchardt et al., 2001), Switzerland (Krejci et al., 2004a; VSA, 2007), Austria (ÖWAV, 2007)
Turbidity	✓	✓		
Ammonia toxicity	✓		✓	Germany (Lammersen, 1997; Borchardt et al., 2001; BWK, 2007), Switzerland (Krejci et al., 2004a; VSA, 2007), Austria (ÖWAV, 2007), United Kingdom (FWR, 1998)
Oxygen deficits			✓	
Pathogens	✓ [d]	✓ [d]	✓ [d]	
Temperature	✓			Switzerland (Krejci et al., 2004a; VSA, 2007)

[a] Average runoff Q < 0.1 $m^3 s^{-1}$; width < 1 m; average flow speed v = variable
[b] Q > 0.1 $m^3 s^{-1}$; width > 1 m; v > 0.5 $m s^{-1}$
[c] Q > 0.1 $m^3 s^{-1}$; width > 5 m; v < 0.5 $m s^{-1}$
[d] only relevant if water body is used for bathing

To aid case-specific assessment, several countries have established non-binding technical guidelines (Table 3.1) that recommend water-quality goals for short-term exposure, for instance in the form of duration-concentration thresholds. Such water quality goals are important for the assessment of the current situation and, if action is required, for the planning of measures and the subsequent control of success.

Integrated deterministic models - representing both the source of CSO and the receiving rivers - are suggested for process understanding and planning of measures by the technical guidelines in Table 3.1 and by a number of scientific publications (House et al., 1993; Ellis and Hvitved-Jacobsen, 1996; Rauch et al., 2002a; Muschalla et al., 2009). Based on existing studies and recommendations the following three key rules for integrated modelling were identified:

a) The complexity of model components should be adapted to the impact(s) which require mitigation and the processes governing these impacts (Rauch et al., 1998a; BWK, 2007; Matzinger et al., 2012a).

b) The integrated model needs to be properly calibrated / validated (Muschalla et al., 2009), with a particular focus on the impacts, which should be mitigated. The calibration / validation step requires event-based measurements at sufficient temporal and spatial resolution to cover the dynamic nature of CSO.

c) The integrated model should be applied for an exemplary scenario analysis as a functionality and plausibility check (Muschalla et al., 2009). The results should be related to the defined water quality goal (BWK, 2007; VSA, 2007).

Several studies demonstrated the possibility of applying integrated models for CSO related questions, including either step (a) (Rauch et al., 2002b; Fu et al., 2009), step (b) (Even et al., 2007; Motta et al., 2010) or step (c) (Holzer and Krebs, 1998; Kashefipour et al., 2002). However, no studies were found that cover all the three steps. Moreover, many demonstrations follow a conceptualized approach, either by focusing on one sub-catchment or river stretch (e.g. Andrés-Doménech et al., 2010) or by simplifying the representation of either the sewer model (e.g. Even et al., 2007; Motta et al., 2010), the river model (e.g. Holzer and Krebs, 1998; Andrés-Doménech et al., 2010) or both (e.g. Rauch et al., 2002b).

So, on the one hand, legislation and technical guidelines recommend or demand integrated and reliable model tools to understand CSO impacts and to plan and implement impact-based CSO strategies. On the other hand, examples for integrated model applications that cover more than one of the above steps (a) to (c) are sparse. Accordingly, there is a major gap in integrated model studies that go all the way to the establishing of planning instruments. This is particularly true for complex urban drainage and river systems, as they occur in most large cities.

The following study presents such an integrated model study for a large city with a complex urban drainage system and a water quality goal that requires an ecosystem representation in the river model. The study aims at contributing towards the identified knowledge gaps by:

- Developing and demonstrating an integrated modelling approach for complex urban conditions,
- understanding of in-river processes that lead to negative impacts after CSO,
- studying the expected effect of different types of CSO mitigation measures and
- highlighting strengths and weaknesses of the chosen approach.

3.2 Material and methods

3.2.1 Study site

The combined sewer system (CS system) in central Berlin drains an impervious area of 64 km^2 (Table 3.2), of which 92% are included within the model boundaries (59 km^2, indicated by the dot-and-dashed line in Figure 3.1). During 30 to 40 rain events per year the CS system overflows to the River Spree and its side channels via 179 outlets (on average one outlet every ~200 m), amounting to an annual CSO volume of ~7 million m^3 containing ~0.6 million m^3 of untreated sewage (Weyrauch et al., 2010).

The River Spree is a lowland river with very low flow speeds of 0.15 ± 0.10 m s^{-1}. During major rain events the total CSO inflow reaches the same order of magnitude as the average flow of the river of 27 m^3 s^{-1} (Table 3.2), leading to a significant impact on river ecology (Riechel et al., 2010). A 12 km stretch of the River Spree and another 15 km of side channels are included within the model boundaries (indicated by the thick black line in Figure 3.1).

Table 3.2: Characteristics of the Berlin combined sewer system (city centre only) and the receiving River Spree.

Parameter	Unit	Value
Combined sewer system (from Weyrauch et al., 2010)		
Impervious area	km^2	64
Inhabitants	10^6	1.4
Total storage capacity	m^3	213,000
Average CSO volume per year	10^6 m^3	7
Average share of wastewater in CSO	%	9
Number of CSO discharge points	-	179
River Spree (from Matzinger et al., 2012b)		
Average flow [a]	m^3 s^{-1}	27 ± 18
River slope	%	0.009
Total phosphorus [a]	g m^{-3}	0.17 ± 0.07
Chlorophyll-a [a]	µg L^{-1}	27 ± 22
Biological oxygen demand in 5 days [a]	g m^{-3}	3.3 ± 1.0

[a] 10-year average ± standard deviation at flow km 16.9 (Figure 3.1)

3.2.2 Derivation of water quality goal and impact assessment

Of the expected CSO impacts for slow-flowing lowland rivers not used for bathing purposes (Table 3.1), ammonia toxicity could be ruled out, since ammonia remained below critical levels by a factor of 8 during two years of continuous monitoring in the River Spree (Matzinger et al., 2012b). In contrast, low concentrations in dissolved oxygen (DO) were found to be critical according to every impact assessment guideline in Table 3.1 (Riechel et al., 2010). DO can drop below 2 mg L^{-1} after CSO and lead to occasional fish kills (Matzinger et al., 2012b). Critical DO zones can prevail in the most affected 12 km long river section for several days, because of the river's low flow speed. In addition to acute CSO impacts, the River Spree is also under significant pressure from upstream pollution, which can lead to DO < 5 mg L^{-1} during extended periods in summer, even in the absence of CSO (Riechel et al., 2010).

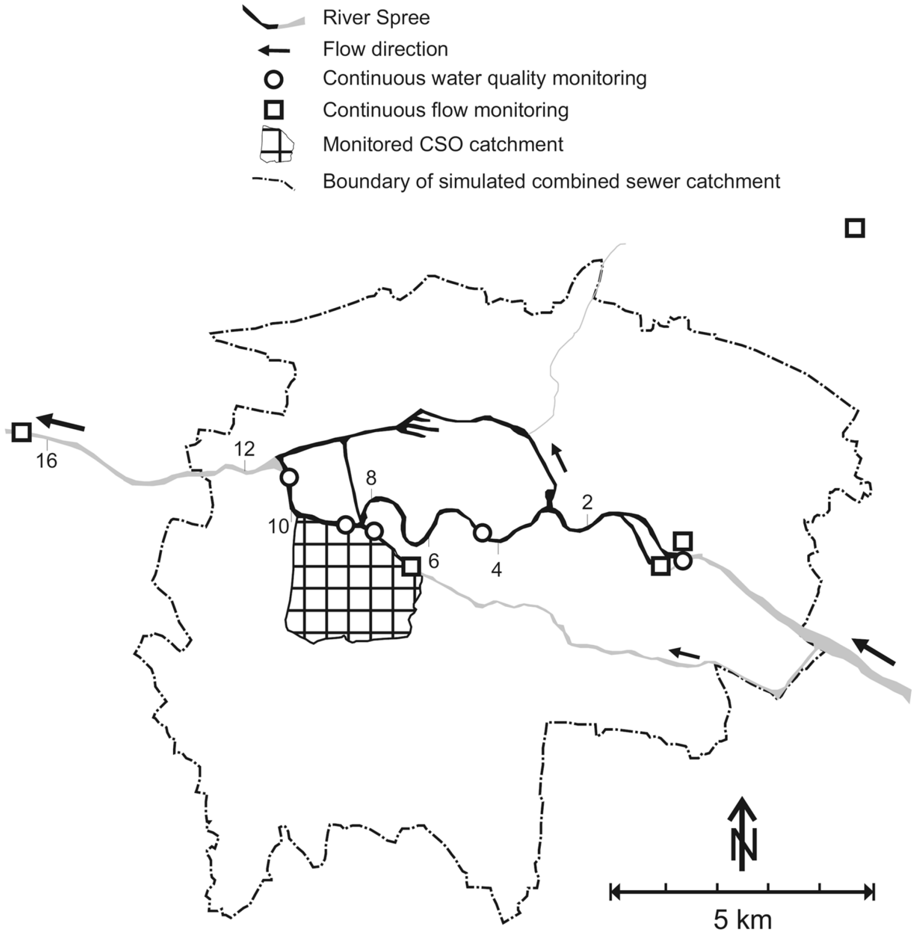

Figure 3.1: Map of the study area. The River Spree and its side channels are shown in grey, the central section covered by the presented model is shown in black. Flow kilometers are indicated along the river, starting at the upper model boundary. The combined sewer area affecting the simulated river stretch is indicated by the dashed line. River monitoring points are shown as squares or circles, the monitored sub-catchment of the sewer system is indicated by squared patterns (see legend).

For the evaluation of DO deficits within this modelling study, the duration-concentration-thresholds suggested by Lammersen (1997) were used, since they have a profound scientific basis and take temperature effects into account. The Lammersen-approach, which is also a basis for the German technical guideline (BWK, 2007), defines DO thresholds for eight different durations ranging from 10 min to 24 h. DO thresholds appear as a function of temperature considering that the oxygen demand of aquatic organisms increases with water temperature as described by Downing and Merkens (1957). Here, a time period is classified as an event with “suboptimal” DO conditions, when for a given temperature at least one of the eight concentration-duration-thresholds is violated. Events with low DO are separated by a recovery period of six hours following the recommendations of FWR (1998). In addition to these suboptimal conditions, that imply any kind of adverse effects, a threshold of 2 mg L^{-1} for ≥ 30 min was used in this work as a second quality standard for “critical” DO conditions. This value is derived from the lethal concentration LC50 of the asp (*Aspius aspius*), the most sensitive indigenous fish species of the River Spree (Wolter et al., 2003). Below 2 mg L^{-1} major fish kills can occur. If in turn DO remained above 2 mg L^{-1} the River Spree would allow major populations of the asp and the common barbell (*Barbus barbus*), two key species expected in this type of lowland rivers (C. Wolter, pers. comm., 2011).

3.2.3 Measurement data

A two-year monitoring program was carried out at a CSO outlet and in the River Spree to obtain model input data as well as data for model calibration and validation (Caradot et al., 2011).

CSO monitoring was performed at one major CSO outlet, to which roughly 10% of the entire CS system is connected (hatched area in Figure 3.1, connected impervious area: 8 km², inhabitants: 132,000). Flow during CSO was measured by two Doppler current profilers combined with an ultrasonic water level sensor installed in the overflow sewer. Total suspended solids (TSS), equivalent chemical oxygen demand (COD) and 5-day biological oxygen demand (BOD_5) were measured by a continuous UV-vis spectrometer installed in a bypass which was fed by a peristaltic pump. Ammonium nitrogen (NH_4-N) was measured with an ion-selective electrode. Online probes were calibrated and validated with analyzed grab samples taken by an autosampler during CSO. Overall uncertainty in measured BOD_5 and COD loads was found to be in the order of 30% (Caradot et al., 2013b).

River flow (Q) was measured at the weir upstream (river km 0) and downstream of the simulated river stretch (km 16.9) and on the major side channel (squares in Figure 3.1); water level is measured at the end of the stretch. Water quality was measured at a 15 min-interval at five monitoring stations (circles in Figure 3.1), each equipped with temperature (T) and DO probes. Additionally, monthly grab samples were analyzed for a range of chemical and biological parameters at the upstream model boundaries and at one point at km 8.3 within the simulated river stretch through the state-run monitoring program.

In addition to the conducted CSO monitoring campaign, operational data of the Berlin water utility was employed, covering water level measurements at the main pumping stations of the sewer network and water quality samples during dry weather for one exemplary sub-catchment. In terms of meteorological data, measurements from five rain gauges (unpublished data, Berliner Wasserbetriebe) as well as air temperature, wind speed, humidity, cloud cover and solar radiation data from one central Berlin station (Tempelhof, public data, Deutscher Wetterdienst) were used as boundary conditions for the sewer and river model.

3.2.4 Model setup

Urban drainage model

The goal variable DO in the river depends both on CSO hydraulics and substance loadings. As a result, an urban drainage model is needed which covers both aspects. In our case the hydrodynamic rainfall-runoff and pollutant load model InfoWorks CS (WSL, 2004) was selected. It solves the full Saint-Venant equations and simulates build-up and wash-off of particulate and dissolved pollutants. Quantity and quality of domestic wastewater is represented in form of hydrographs and pollutographs. Degradation processes in the sewer are not considered. Given the size of the Berlin CS system, a “coarse” grid of the sewer network with 5821 nodes and 6660 conduits was used for which conduits with a diameter < 0.3 m were neglected or lumped together.

The model was set up and initially calibrated by Pawlowsky-Reusing et al. (2006). Hydraulic calibration was conducted for each of the 15 modelled sub-catchments on the basis of water level measurements at the main pumping station for dry and wet weather conditions (typically four days each). Water quality during dry weather was calibrated with measurements at the main pumping station of one exemplary sub-catchment for four individual days (1-hour composite samples, 5-min sampling interval). Water quality during wet weather was finally calibrated on the basis of continuous measurements taken during five CSO events within the presented study (see section 3.2.3). Optimization criteria were total loads of BOD_5, COD, TSS, NH_4-N and the ratio between BOD_5 and COD, an important boundary condition for the subsequent river model.

Physicochemical variables which are not simulated explicitly (e.g. pH, DO or T) were derived from literature data, own measurements or assumptions. Regarding DO, existing measurements from a Berlin CSO structure (n = 2) were in the range of DO measurements in raw sewage (0.4 ± 0.9 mg L^{-1}, n = 88). Given the significant sampling uncertainty for DO in sewer systems and an expected DO concentration of 0 mg L^{-1} in raw sewage, we also assumed a constant value of 0 mg DO L^{-1} for CSO spill. This value is plausible, given the exceptionally low vertical gradients (median value: 0.2%) and the high residence time in the Berlin sewer system that leads to sedimentation and accumulation of reduced substances during dry weather periods (Schwarzböck et al., 2010) and considerable oxygen consumption in the sewer during wet weather. Model output uncertainties due to that assumption were estimated by varying the DO concentration in CSO spill within the observed range (0 – 2 mg DO L^{-1}) for all CSO events of the validation period, which resulted in a maximum deviation of DO in the river of 0.5 mg L^{-1}. Temperature in CSO was assumed to be equal to the river water temperature simulated upstream of the respective discharge point at each time step.

River water quality model

The river network (black river section in Figure 3.1) was simulated by the one-dimensional model Hydrax/QSim of the German Institute of Hydrology. The sub-model Hydrax simulates river hydraulics by solving the entire Saint-Venant equations (Oppermann et al., 2015). Input data are flow at the upper model boundaries and the CSO inlets (interface with InfoWorks CS) and water level at the lower boundary. One-dimensional model representation was chosen since no horizontal or vertical water quality profiles were observed during preliminary investigations. Water quality was calculated by the sub-model QSim (Kirchesch and Schöl, 1999) hydraulically driven by the sub-model Hydrax. Other model inputs are water quality data at the upper model boundaries including the simulated CSO outlets as well as meteorological data (see section 3.2.3).

QSim simulates quasi-closed material cycles for carbon (C), nitrogen (N), phosphorus (P), silicium (Si) and DO. Apart from the water column, a sediment compartment is calculated separately. The cycles include (i) growth, respiration and death of three groups of phytoplankton (Schöl et al., 2002; Quiel et al., 2011), (ii) growth through feeding, respiration and death of zooplankton (Schöl et al., 2002), (iii) heterotrophic decomposition of organic matter (Kirchesch and Schöl, 1999; see below) and (iv) nitrification and denitrification (the latter only in the sediment compartment). In addition, heat budget and acid-base equilibria are simulated (Kirchesch and Schöl, 1999). Available benthic modules (benthic algae and filter feeders, see Schöl et al. (1999) and Schöl et al. (2002)) were turned off in this application given the high turbidity and the lack of suitable substrates in the Berlin River Spree.

Processes with potential importance for DO are phytoplankton production, atmospheric exchange, decomposition of organic matter, nitrification and species respiration. The decomposition of organic matter is expected to be of primary importance when considering DO depressions from CSO. QSim considers two steps of hydrolysis from particulate through dissolved to monomeric organic C, followed by consumption by explicitly simulated heterotrophic bacteria under the use of DO. Particulate and dissolved organic C is each split into fast degradable, slowly degradable and inert fractions. The degradability, as well as the hydrolysis and degradation rates depend on the ratio between BOD_5 and COD, following the approach by Billen (1991). As a result, organic matter from raw sewage will decompose significantly faster than dead phytoplankton biomass.

Model coupling and validation

The sewer and river models simulate at a 1- and 15-min time step, respectively. Model coupling from the sewer to the river compartment is realized via export and import of text files for flow and water quality simulated at each CSO outlet. Formatting and temporal aggregation of sewer model output is done within a database application using structured query language (SQL) scripts. Hydraulic feedback from the river to the sewer model is considered by providing average measured water level in the river as constant sewer model input, given that water level in the River Spree is regulated with variations < 0.03 m.

Validation of the coupled sewer-river-model was based on two years of river flow measurements at km 16.9, continuous DO and T measurements at river km 4.8, 9.0 and 10.3, as well as Chlorophyll-a (Chl-a), nitrate (NO_3-N) and total phosphorus (TP) data from monthly grab samples at km 8.3 (see section 3.2.3).

3.2.5 Goodness-of-fit assessment

To assess coupled model performance three types of criteria were applied: i) qualitative criteria based on visual comparison, ii) quantitative criteria based on goodness-of-fit or error index statistics and iii) CSO impact assessment criteria:

Qualitative criteria are derived from graphical techniques such as time series plots. They account for the model representation of daily or annual variations and the occurrence of minimum or maximum peak values in the case of CSO events.

Quantitative criteria for model performance used in this study are the dimensionless goodness-of-fit indicator *NSE* (Nash and Sutcliffe, 1970) and the error index statistic *RMSE* which are calculated as follows:

$$NSE = 1 - \frac{\sum_{i=1}^{n}(m_i - s_i)^2}{\sum_{i=1}^{n}(m_i - \overline{m})^2} \qquad \textit{(Equation 3.1)}$$

$$RMSE = \sqrt{\frac{\sum_{i=1}^{n}(m_i - s_i)^2}{n}} \qquad \textit{(Equation 3.2)}$$

with m_i and s_i representing the measured and simulated value at time step *i*, *n* representing the total number of time steps and *m* representing the average measured value over n time steps. *NSE* quantifies the deviation between measured and simulated data relative to the variance in measured data, with values ranging between -∞ (no agreement) and 1 (perfect fit). *RMSE* quantifies the deviation between measured and simulated data in the units of the variable, with 0 being the optimum value. According to Jeong et al. (2010) and Moriasi et al. (2007), *NSE* values ≥ 0.65 and *RMSE* values lower than half the standard deviation of measured data can be considered as a good representation of measurements.

CSO impact assessment criteria quantify to which extent the model is able to reproduce the occurrence of acute CSO impacts that can harm fish and other aquatic organisms. In this study the annual frequency of suboptimal and critical DO conditions in the river (see section 3.2.2) is calculated for both measured and simulated data. The smaller the difference between measurements and simulations, the higher the model's capability to represent potential violations of impact based water quality goals.

3.2.6 Model sensitivity and parameter identifiability

A local sensitivity analysis has been carried out for the goal variable DO to rule out potential errors in model formulation and identify dominant river processes for dry and wet weather conditions. 178 model runs with subsequent variation of 89 key parameters by ± 20% ("one-at-a-time") have been conducted and compared to the default simulation. Batch runs were realized with the calibration tool KALIMOD (Uhl and Henrichs, 2010). The model parameters were then ranked according to the difference in DO between simulations with and without parameter variation (quantified via the *NSE*, see Equation 3.1). The analysis was carried out at river km 8.8 for two different settings: i) without consideration of CSO inputs for the time period April to November 2010 by decoupling the sewer from the river model simulating background conditions and ii) under the impact of a major CSO event in July 2010.

Identifiability of the three most influential model parameters for both dry and wet weather conditions was analyzed for the month of July 2010 by comparing the respective DO time derivatives, i.e. the differences in simulated DO time series with and without parameter variation. For each possible pair of the six parameters the coefficient of determination r^2 of the respective DO time derivatives was calculated. High values of r^2 indicate that parameter behavior is correlated whereas low values of r^2 imply that parameters influence independent processes and hence are identifiable.

3.2.7 Process understanding

For further process understanding and later selection of impact mitigation measures, the specific processes that can lead to a decrease in dissolved oxygen were analyzed for two exemplary CSO events in July 2010. For this purpose, the model was run repeatedly, each time setting the following process-related CSO water quality variables at the interface of the sewer and the river sub-model to upstream river conditions:

- BOD_5 and COD for the degradation of organic matter,
- TSS for turbidity-induced change in DO production by phytoplankton,
- DO for the mixing of river water with CSO water poor in DO and
- NH_4-N for the impact of nitrification.

The relevance of these variables and their associated river processes was calculated from the difference in DO simulated with and without the respective CSO water quality variable in proportion to the overall DO deficit quantified after neglecting all CSO water quality variables. The analysis was done through a Lagrangian approach, following one water parcel downstream starting at a major CSO outlet at km 0.9 of the simulated river stretch.

3.2.8 Scenario analysis

The developed and validated integrated model was tested for selected CSO management scenarios to demonstrate its sensitivity to changed boundary conditions and its applicability as a planning instrument. Input data for all scenarios were derived from measurements of the year 2007 including rainfall, river water quantity and quality as well as meteorological data. The year

2007 was chosen as it represents one of the most severe periods regarding the occurrence of heavy rainfalls and adverse DO conditions in the river and hence is well suited to demonstrate the relative effect of CSO countermeasures. All simulations were limited to the eight-month time period April to November, as critical river impacts from CSO are not expected beyond that period (Riechel, 2009). Mitigation measures to be included in the scenario analysis were selected on basis of the processes identified as responsible for acute oxygen stress after CSO.

3.3 Results and discussion

3.3.1 Model calibration and validation

Calibration of the pollutant-build-up and wash-off parameters of the sewer sub-model yielded a good agreement between simulated and measured data. The relative error in total simulated pollutant loads for five CSO events is 5%, -2%, 8% and -13% for BOD_5, COD, TSS and NH_4-N, respectively. The simulated and observed BOD_5-COD mass ratios, particularly important for the model representation of acute DO depressions in the river, differ by only 2% in total. Total CSO volume for the same five events is simulated with an error of 3%.

Attempts for an individual calibration of the river sub-model on basis of genetic algorithms did not lead to a further improvement of model performance. Hence, the default parameter set as suggested by the model developers (Kirchesch and Schöl, 1999) was used.

Quantitative validation of the coupled sewer-river-model for two years of continuous measurements shows a good agreement for both river flow and water quality (Table 3.3). Average Nash- Sutcliffe model efficiencies *NSE* for Q, T and DO simulated and measured at a 15-min time step are 0.91, 0.99 and 0.66, respectively. Average *RMSE* values for the same variables are 6.7 $m^3 s^{-1}$, 0.38 °C and 0.96 mg L^{-1}, between 10 and 55% of the standard deviations of measured data. Hence, according to Moriasi et al. (2007) and Jeong et al. (2010) model performance can be considered as good (for DO) or very good (for Q and T). Model efficiencies for state variables measured at monthly intervals are lower but still reach *NSE* values of up to 0.75, 0.71 and 0.76 for Chl-a, NO_3-N and TP.

For the designated use of the model tool it is particularly important to properly simulate the occurrence of acute CSO impacts in the river. As shown in Figure 3.2 for a major CSO event in July 2010, the model simulates CSO discharges in good agreement with measurements despite a slight underestimation of the peak flow rate. At the same time, the resulting DO decrease simulated in the river follows the observations. Even though not all DO depressions are accurately reproduced by the model, simulation results generally follow the observed patterns. In particular, the whole range of observed DO concentrations after CSO down to 0 mg L^{-1} is simulated. Regarding the annual frequency of critical DO conditions in the river (DO ≤ 2 mg L^{-1} for ≥ 30 min) simulations and observations differ by one or two days per year, depending on the monitoring station, which can be considered as satisfactory.

Table 3.3: Goodness-of-fit indices for the validation periods April to November of the years 2010 and 2011.

	2010		2011	
	NSE	RMSE	NSE	RMSE
Q	0.96	6.1 $m^3 s^{-1}$	0.85	7.2 $m^3 s^{-1}$
T [a]	0.99	0.35°C	0.98	0.40°C
DO [a]	0.70	0.9 mg L^{-1}	0.61	1.0 mg L^{-1}
Chl-a	0.75	10.6 µg L^{-1}	-1.61	12.7 µg L^{-1}
NO_3-N	0.71	0.15 mg L^{-1}	0.13	0.42 mg L^{-1}
TP	0.61	0.03 mg L^{-1}	0.76	0.02 mg L^{-1}

[a] average indices for three different monitoring stations (Figure 3.1)

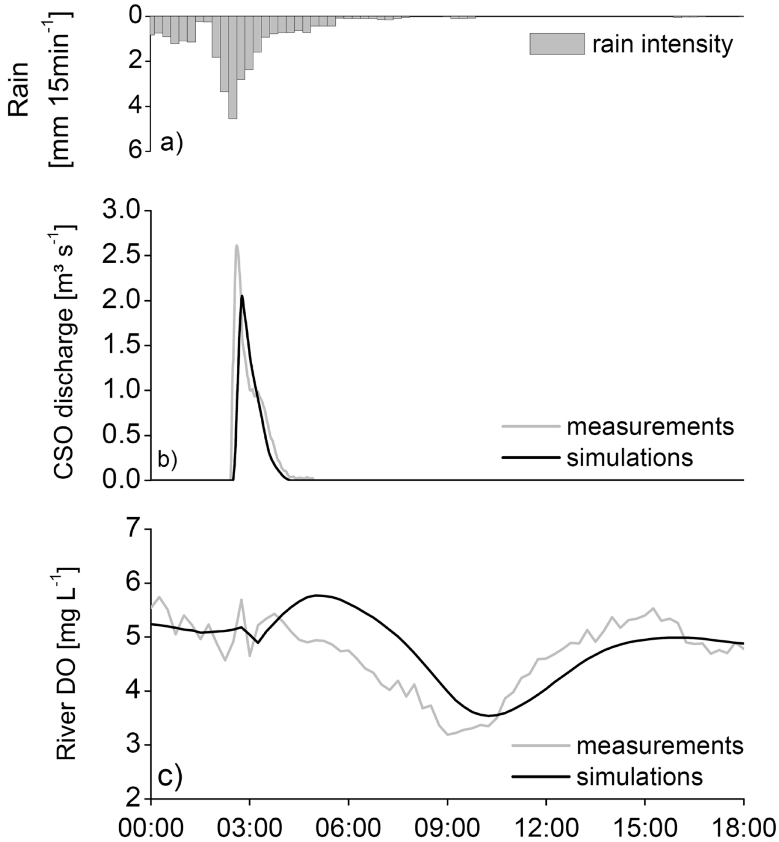

Figure 3.2: Measured rainfall intensity for an exemplary event of 34.8 mm rain depth (a), measured and simulated CSO discharge at the monitored CSO outlet (b) and measured and simulated DO concentration at river km 4.8 (c).

3.3.2 Model sensitivity and parameter identifiability

For dry weather conditions, the most influential model parameters regarding DO are the maximum growth rate of diatoms (*gro_max_dia*) and the Chl-a-biomass ratio of diatoms and cyanobacteria (*chl_bio_dia* and *chl_bio_cya*), the dominant phytoplankton groups in the River Spree. In contrast, for wet weather conditions the hydrolysis rate of easily degradable dissolved organic carbon (*hyd_eas_car*), the maximum uptake rate of monomeric carbon by heterotrophic bacteria (*car_upt_bac*) and the yield coefficient of heterotrophic bacteria biomass (*bac_bio_yld*) have the highest effect on DO. Thus, regarding the sensitivity of model parameters, dissolved oxygen in the absence of CSO principally depends on phytoplankton dynamics, while the most dominant process for oxygen depressions after CSO is the degradation of organic matter.

Figure 3.3 shows the derivatives of the simulated DO concentration with respect to time for the parameters *gro_max_dia* (Figure 3.3 b) and *hyd_eas_car* (Figure 3.3 c) for the month of July 2010. While the maximum growth rate of diatoms dominates the DO regime most of the time, its influence rapidly declines during wet weather when turbidity due to CSO increases and solar radiation is typically low. In contrast, the hydrolysis rate of organic carbon which plays a minor role during dry weather becomes the main driver for simulated DO in the case of CSO.

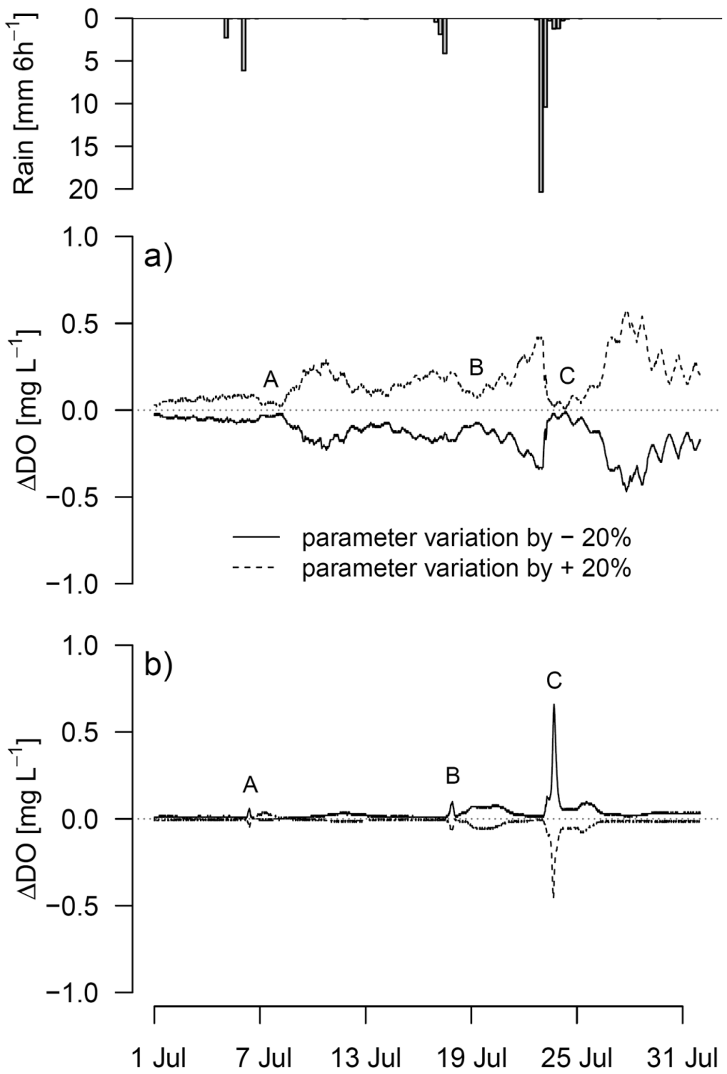

Figure 3.3: Rainfall intensity (a) and time derivatives of simulated DO for the variation of the maximum growth rate of diatoms (gro_max_dia) (b) and the hydrolysis rate of easily degradable dissolved organic carbon (hyd_eas_car) (c) for the simulation period July 2010 at river km 8.8. Letters "A" to "C" indicate the occurrence of CSO events.

Table 3.4 shows the correlation matrix of DO time derivatives for changes in the three most influential parameters for both dry and wet weather conditions. It reveals that the maximum growth rate and Chl-a-biomass ratio of diatoms are highly correlated (r^2 = 0.97) which means that they influence the DO concentration in a similar way. A weaker correlation (r^2 = 0.62) was found for the Chl-a-biomass ratios of diatoms and cyanobacteria since both phytoplankton species are dominant during different time periods but still depend both on nutrient and light availability. On the other hand, parameters that control the degradation of organic matter are completely independent of the phytoplankton activity ($r^2 \leq 0.03$) but show nearly-linear dependency among themselves ($r^2 \approx 0.95$). The results indicate that complex overparameterized models, such as QSim, contain a number of non-identifiable parameters among the most influential ones regarding DO. On the other hand, it is important to note, that parameters can become identifiable during specific situations, as exemplified by alternating dominance of diatoms and cyanobacteria in the present case.

Table 3.4: Correlation matrix for the most influential model parameters for dry weather (gro_max_dia, chl_bio_dia, chl_bio_cya) and wet weather conditions (hyd_eas_car, car_upt_bac, bac_bio_yld). Values in the matrix represent the coefficients of determination r^2 for the time derivatives of simulated DO for a parameter variation by -20%.

	gro_max_dia	*chl_bio_dia*	*chl_bio_cya*	*hyd_eas_car*	*car_upt_bac*	*bac_bio_yld*
gro_max_dia	1.00					
chl_bio_dia	0.97	1.00				
chl_bio_cya	0.71	0.62	1.00			
hyd_eas_car	0.01	0.00	0.02	1.00		
car_upt_bac	0.02	0.01	0.03	0.96	1.00	
bac_bio_yld	0.03	0.02	0.03	0.95	0.95	1.00

3.3.3 Process understanding

The time-dependent contribution of different processes to the observed DO deficit after a CSO event in July 2010 is shown in Figure 3.4 (see section 3.2.7 for methodology). For the exemplary event a maximum DO deficit of 3.5 mg L^{-1} was determined. As expected from sensitivity analysis (section3.3.2) and from literature, the degradation of organic matter turned out to be the most relevant process for oxygen depletions in the river after CSO (Streeter and Phelps, 1925; Harremoës, 1982; Cox, 2003; Marsalek, 2003). Overall, 97% of the DO deficit can be explained by the following three processes:

- mixing of river water with CSO spill water poor in DO,
- reduced phytoplankton activity due to CSO-induced turbidity and
- degradation of organic matter by heterotrophic bacteria.

Nonetheless, the relative contribution of the processes to the resulting DO deficit varies with time as shown in Figure 3.4.

The first hours after the CSO spill are typically impacted by the mixing of river water with large overflow volumes poor in DO (dashed line in Figure 3.4). This process is probably particularly relevant for large cities with a high degree of imperviousness and a flat sewer system that favors anoxic conditions.

Approximately ten hours after the CSO peak spill, degradation of organic matter becomes the dominant process of the oxygen budget. After a total of ~24 h biological oxygen demand tends to zero and the relative contribution of the process to the simulated DO deficit stagnates at 80% (point-dashed line in Figure 3.4).

The reduced phytoplankton activity due to CSO-induced turbidity gains importance with time and is the only process still slightly increasing 100 h after CSO, though never contributing more than 15% of the observed DO deficit (pointed line in Figure 3.4). This process is expected to be relevant for water bodies with a high productivity in which photosynthesis dominates the aquatic oxygen budget, such as shown for the River Spree during dry weather (section 3.3.2).

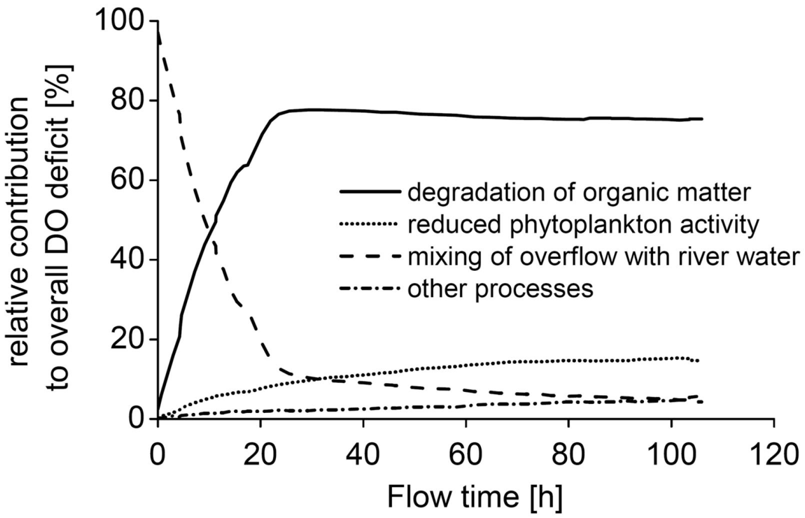

Figure 3.4: Relative contribution of different river processes to the simulated DO deficit after a CSO event in July 2010 as a function of the flow time.

The dependence on time indicates that different processes can be responsible for observed DO depressions at a given site, depending on flow speed. Regarding mitigation, the presented results imply that not only the increase of storage capacity to retain CSO volumes and organic pollutants can mitigate negative CSO impacts but also the removal of suspended solids or the aeration of the CSO spill at the outlet.

3.3.4 Scenario analysis

Selection of tested mitigation measures

Based on the identified in-river processes different CSO management scenarios S1 - S5 were defined. The starting point of the scenario analysis is the sewer status of the year 2010 (S1) with a specific storage capacity of 33 m^3 ha^{-1}. S2 tests an increase in storage capacity of 46% which is expected to affect all three DO-suppressing processes by reducing the volume of CSO spill water poor in DO and by that also pollutant loads that affect algae growth and degradation processes. To be as realistic as possible, S2 is based on specific measures planned within the Berlin sewer rehabilitation program implemented until the year 2020. The following hypothetical scenarios S3 to S5 use scenario S2 as a starting point. In S3, the impervious area of each sub-catchment is reduced by 20%. Similar to S2, S3 is expected to reduce CSO volume and thus all the three in-river processes. In contrast, increasing DO in CSO to 5 mg DO L^{-1} through aerators in S4 only tackles one of the three processes. Finally, end-of-pipe filters at the three major CSO outlets in S5 (assumed treatment capacity per outlet: 2 m^3 s^{-1}, removal efficiency for TSS and particulate fractions of BOD_5, COD, TN and TP: 50%) are expected to reduce negative effect of turbidity on phytoplankton growth and partly degradation of organic (particulate) matter, whereas dissolved fractions and water quantity are not affected.

Understanding of mitigation effects

Table 3.5 summarizes simulated CSO emissions and river impacts for all scenarios S1 - S5 for the eight-month simulation period. It reveals that the studied CSO management options can reduce highly fish-critical oxygen conditions to a varying extent from ~15 h yr^{-1} (status quo) down to 3 h yr^{-1}. However, none of the measures distinctly reduces suboptimal oxygen conditions on aquatic organisms, since their occurrence is mostly due to background pollution.

The increase in storage capacity by 46% (scenario S2) leads to a simulated reduction in CSO volumes by 17% and in pollutant loads between 21% for TSS and 31% for NH_4-N. The result shows that increased storage can retain the first, more polluted portion of the CSO volume. The simulation results are in line with measurements of higher concentrations in the first half of CSO events, though first flush according to the definitions of Diaz-Fierros et al. (2002) seldom occurs in Berlin (Caradot et al., 2013a). Moreover, wastewater related pollutants, e.g. NH_4-N, present in the sewer at the beginning of the rainfall event, can be retained more effectively than those compounds predominantly originating from surface wash-off, e.g. TSS. This is particularly true for smaller overflow events for which the mixing ratio of sewage to stormwater tends to be higher than for large events.

For the reduction of the impervious area by 20% (S3) a reduction in CSO volumes by 32% (compared to S2) has been simulated which is in the same range as the reduction in dissolved and particulate pollutant loads (32 - 33%). This means that, contrary to the effects of the storage capacity increase (S2), the mixing ratio of sewage and stormwater in CSO remains unaffected by surface runoff reduction.

Regarding river impacts, both scenarios S2 and S3 reduce the frequency of critical DO conditions by one third. Almost the same relative effect in the river can be achieved with an aeration of CSO spill (S4) although pollutant loads remain unaffected.

Partial treatment with end-of-pipe filters (S5) leads to a reduction in TSS loads by 18% and slightly lower reductions for the other pollutants, depending on their particulate fraction. This reduction results in a notable improvement in fish-critical DO conditions from 2.6 (S2) to 1.9 days per year (-27%) on average for the CSO impacted river stretch. In direct comparison, scenarios S4 and S5 have a similar effect in the more downstream part of the river (last 3 km), whereas S4 performs significantly better in the upstream part (first 8 river km), which highlights the delayed effect of biodegradation and reduced phytoplankton activity on dissolved oxygen (see section 3.3.3). Thus, the chosen measure type does not only define which DO-depleting process is reduced but can also have an impact on the place of occurrence of low DO in the river.

Table 3.5: Simulated CSO emissions and river impacts for the sewer status quo 2010 (S1) and four management scenarios (S2-S5).

	S1: Status quo (2010)	**S2:** Storage increase (2020)	**S3:** S2 + reduced imp. area	**S4:** S2 + CSO aeration	**S5:** S2 + CSO treatment
CSO emissions					
Number of events, V>10,000 m^3	34	32	29	32	32
Volume [10^6 m^3]	5.9	4.9	3.3	4.9	4.9
BOD_5 load [t]	354	273	182	273	226
COD load [t]	960	745	497	745	632
TSS load [t]	839	663	442	663	541
NH_4-N load [t]	12.2	8.4	5.7	8.4	8.4
TN load [t]	25.6	18.2	12.3	18.2	16.5
TP load [t]	4.1	3.0	2.0	3.0	2.7
River impacts					
Frequency of suboptimal DO conditions [d yr^{-1}]	27.5	26.9	25	25.4	25.6
Frequency of critical DO conditions [d yr^{-1}]	3.9	2.6	1.3	1.4	1.9
Total duration of critical DO conditions [h yr^{-1}]	15.3	10.4	3.2	4.1	6.9

Looking at single CSO events, fish-critical situations can mainly be reduced for small to moderate overflow volumes as shown in Figure 3.5 with return periods < 0.5 years. In contrast, the positive effect on the receiving river is negligible for the largest simulated rainfall event with a return period ≥ 15 years (not shown). The moderate event in Figure 3.5 with a total rainfall height of 26 mm and a peak rainfall intensity of 1.9 mm in 15 min produces sharp drops in DO at river km 7.3 immediately after the CSO flow peak. They are caused by the inflow of oxygen-poor CSO spill water at the CSO outlets located at short distance upstream of the observed river stretch. That first DO drop is followed by a second larger DO depression arriving ~12 h after the CSO spill at km 7.3 and after further flow time of ~6 h at km 10.3, which is primarily the result of oxygen depleting organic material entering the water body at different major CSO outlets further upstream. For the shown event, S2 and S3 lead to an elevation in minimum DO concentrations in the receiving river by approximately 1 mg L^{-1} each, no longer violating the critical value of 2 mg DO L^{-1} at km 10.3 (Figure 3.5). However, the example also shows the limits of using precise thresholds, since small deviations in DO simulation can decide on the final judgment whether a CSO is fish-critical or not; despite the fact that the situation is greatly improved.

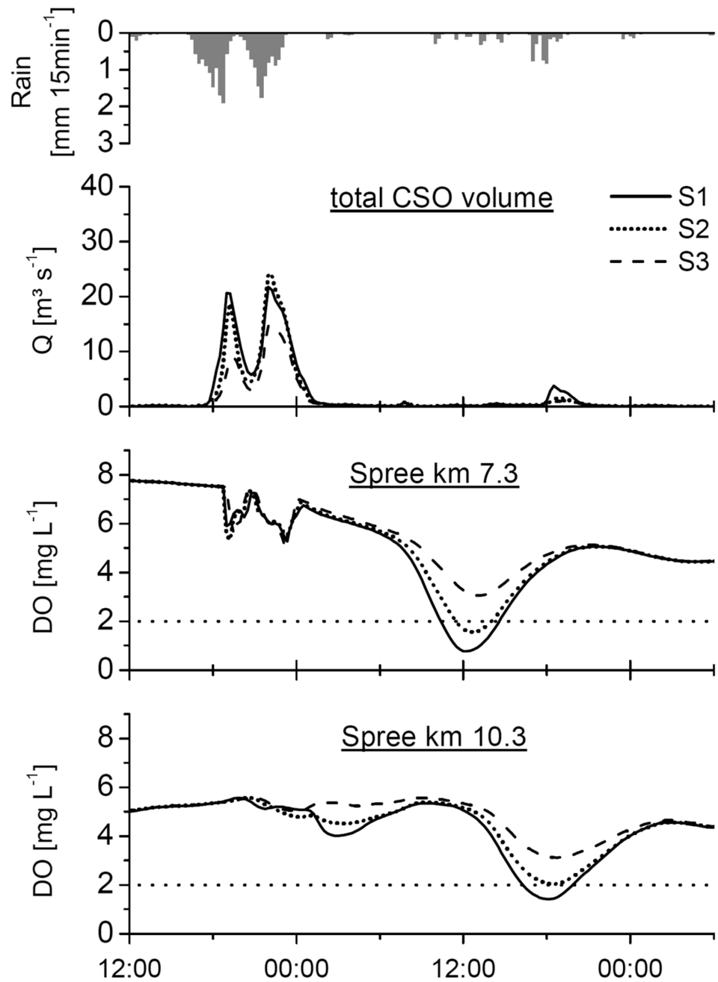

Figure 3.5: Simulated CSO impact under three scenarios – sewer status quo (S1), increased storage capacity (S2) and S2 + reduced impervious surface (S3) – for a rainfall event of 26 mm total rain depth. Horizontal dotted lines in the lower two panels represent the threshold for critical DO conditions.

Overall, the analysis shows that there are different effective mitigation strategies ranging from storage capacity increase or activation to different treatment options and decentralized stormwater management. Nonetheless, to fully rehabilitate urban water bodies and also reduce suboptimal DO conditions, CSO management strategies must go along with measures in the upstream catchment targeting background pollution.

3.4 Conclusions

In this study an integrated modelling and impact assessment approach for planning of CSO control measures has been presented, following the three identified key rules (a) model adaptation to required impact complexity, (b) calibration and validation with event-based monitoring data and (c) plausibility check via sensitivity and scenario analyses. In the following the major strengths of the chosen approach are highlighted:

- As demonstrated, integrated sewer-river-models can represent CSO impacts at city-scale under complex urban conditions in good agreement with measurements (rules a and b).
- The integrated model reacts sensitively to changes in the catchment (e.g. reduction of impervious area), in the sewer (e.g. increase in storage capacity), at the interface of the sewer and the river (e.g. CSO aeration, CSO treatment) and in the river itself (e.g. changes in rates for carbon uptake or hydrolysis) (rule c).
- Given the compliance of rules a to c, such models allow assessing the importance of in-river processes that lead to negative impacts. In the case of the River Spree, the degradation of organic matter, reduced photosynthesis due to turbidity increase and mixing with CSO spill water poor in DO were identified as major CSO-induced processes in the water body. The knowledge on processes allows selection of specific mitigation strategies.

However, the study also revealed some weaknesses of the chosen approach:

- The establishment of an integrated model involves a great effort from complex and large model setup (rule a), monitoring and calibration (rule b) to testing (rule c). A particular trap lies in an inappropriate simplification to save on effort. For instance, skipping rule (a) by choosing a simplified river model representation would have prevented proper understanding of processes that lead to critical DO situations. Without event-based monitoring data (rule b) validation of model representation of negative impacts would not be possible. Finally, without an exemplary sensitivity and scenario analysis (rule c) plausible reaction of the model to changes cannot be judged.
- As any modelling study, the presented work is subject to considerable uncertainties originating from model input, calibration and model structure (Deletic et al., 2012). Available techniques to quantify these uncertainties thoroughly (Dotto et al., 2012) are not applicable due to the high model complexity and the associated computational effort.
- Another source of uncertainty lies in the fact that the DO concentration in CSO spill water is treated as constant river model input although it depends on the mixing ratio of

stormwater and sewage as well as the amount and composition of sewer depositions, among others. Nonetheless, after variation of the model input in the range of observations, model output uncertainty was found to be comparably low.

- As typical for overparameterized mechanistic models, some input parameters are highly correlated indicating a lack in parameter identifiability for the presented model application. If the model was used for different boundary conditions, e.g. under climate change, it would be possible that parameters, that were unidentifiable in the validation period, become identifiable and lead to wrong predictions. However, in the presented study, we expect a moderate impact of this effect, since boundary conditions for validation and scenario analysis were varied to a limited extent.

Acknowledgements

The presented study was carried out at Kompetenzzentrum Wasser Berlin, financed by Berliner Wasserbetriebe and Veolia Water. Rain and sewer system data were supplied by Berliner Wasserbetriebe, river water quality data were supplied by the Berlin Senate Department for Urban Development and the Environment. River water quality modelling was thankfully supported by the German Federal Institute of Hydrology (BfG) and the consulting company "Ingenieurbüro für Wasser und Umwelt". Particular thanks go to Volker Kirchesch of BfG for ongoing adaptations of QSim. An earlier version of the manuscript profited significantly from suggestions by two anonymous reviewers.

Chapter 4: CSO impacts under climate change

This study was published as:

Riechel, M., Uldack, M., Heinzmann, B., Pawlowsky-Reusing, E., Matzinger, A. (2015). Demonstration of a planning instrument for integrated and impact based CSO control under climate change conditions. Book chapter in: *Climate Change, Water Supply and Sanitation: Risk assessment, management, mitigation and reduction*, 65-69. IWA Publishing. https://doi.org/10.2166/9781780405001

This is an adapted version of the original book chapter, complemented with an enhanced description of the climate change scenarios in section 4.2.2 (after Matzinger et al., 2012a).

4.1 Introduction

Combined sewer overflows (CSO), as observed in the city centre of Berlin, can be a significant source of pollution for urban surface waters. One of the main impairments for the Berlin River Spree is the acute depletion of dissolved oxygen (DO), mainly due to the degradation of organic pollutants entering the water body after intense rainfalls. According to long-term continuous measurements of the Berlin water authority, highly critical DO conditions for local fish species are observed on up to 59 days per year at a river stretch highly impacted by CSO.

To reduce negative CSO impacts and meet environmental objectives derived from the European Water Framework Directive (EU, 2000), extensive sewer rehabilitation measures are currently implemented in Berlin. However, an aggravation of ecological deficits must be expected from global climate change (changes in rainfall intensity, temperature increase), which may not only lead to more frequent CSO events but also increase the vulnerability of the ecosystem.

To support decision makers in planning further CSO control measures, a planning instrument has been developed following the methodology described in Matzinger et al. (2015). The planning instrument consists of (i) the commercial sewer modelling software InfoWorks CS (WSL, 2004) that enables to simulate CSO volumes and pollutant loads, (ii) the river water quality modelling software Hydrax/QSim (Kirchesch, 2004) that simulates the impact of CSO on the river and (iii) an impact assessment tool that quantifies adverse effects for aquatic organisms. Figure

4.1 visualises the schematic structure of the planning instrument and its application for the assessment of CSO impacts under a changing climate.

Goal of the demonstration was to test the developed planning instrument for selected CSO control strategies and predicted climate change scenarios. The performance of the different model components and the relative effect of changing boundary conditions was assessed. In the future, the planning instrument will help authorities and water utilities to optimise CSO management strategies and so effectively allocate investments in the wastewater infrastructure.

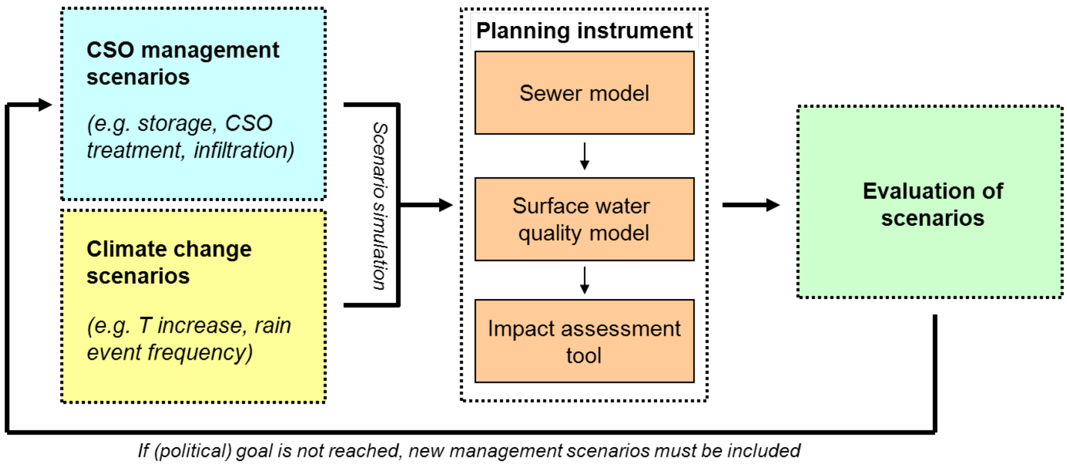

Figure 4.1: Schematic structure of the planning instrument and its application.

4.2 Findings

4.2.1 Model validation

The presented integrated modelling approach was validated with continuous measurements taken over three years at a combined sewer overflow structure and in the receiving river Spree (Caradot et al., 2011). Both the model performance during dry weather conditions as well as the capability to predict emissions and river impacts after CSO was assessed.

Simulated CSO emissions were verified for a representative overflow structure equipped with a flowmeter and online sensors for continuous water quality measurements. For the five studied CSO events of the validation period, emitted volumes and pollutant loads (TSS, BOD_5, COD) could be predicted with an absolute error of less than 10%.

Regarding river hydraulics, both peak flow during CSO and discharge during low flow periods can be very well predicted with the coupled sewer-river-model (model efficiencies according to Nash and Sutcliffe (1970) $\geq$ 0.85 at a 15-minute time step).

Regarding river water quality, the concentration of dissolved oxygen – the goal variable for CSO impact modelling in Berlin – can be simulated in good agreement with measured data. For three different river stations, Nash-Sutcliffe model efficiencies vary between 0.58 and 0.77. Not only the annual oxygen pattern, also the extent and duration of critical oxygen conditions after CSO agrees reasonably well with measured data. Figure 4.2 shows the comparison of measurements and simulations for both CSO emissions and river impacts.

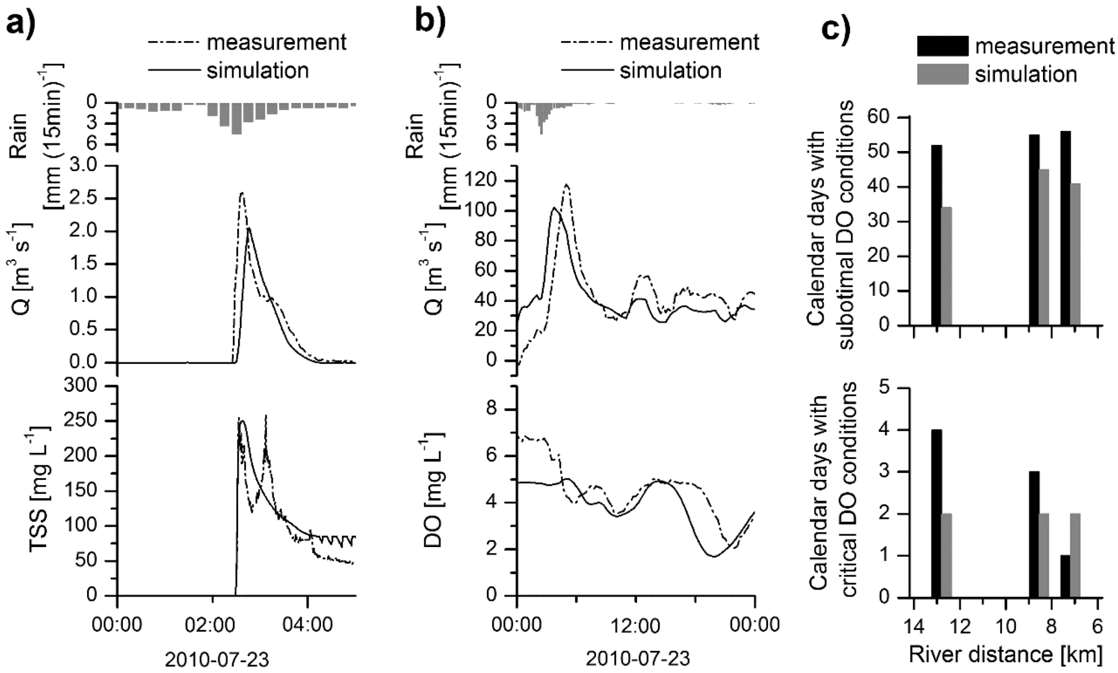

Figure 4.2: (a) Simulated and measured flow and TSS concentration at a major CSO outlet following a rain event on 23 July 2010, (b) simulated and measured flow and DO concentration at a highly CSO impacted river stretch and (c) occurrence of suboptimal and critical DO conditions simulated and measured at three different river stations for the year 2010. Suboptimal DO conditions imply any kind of oxygen stress whereas critical DO conditions refer to the lethal concentration of the most sensitive Spree fish.

4.2.2 Scenario analysis

The developed planning instrument was tested for selected CSO management and climate change scenarios. Main focus was to assess its sensitivity to changing boundary conditions, an important precondition for the future use for strategic planning. Four CSO management scenarios (S1 – S4) and three climate change scenarios (S5 – S7) were studied for input data (meteorological data, water quantity and quality data) of the exemplary year 2007, characterised by a high number of heavy rainfalls in summer:

- Scenario S1: Sewer status quo 2010;
- Scenario S2: Sewer status 2020 (corresponding to the status after implementation of planned rehabilitation measures of the sewer system);
- Scenario S3: S2 + storage volume of the sewer system increased by 20%;
- Scenario S4: S2 + impervious area reduced by 20%;
- Scenario S5: S2 + air and water temperature increased by 1.9°C;

- Scenario S6: S5 + rain intensity increased by 20% (multiplication with a factor 1.2);
- Scenario S7: S5 + rain intensity reduced by 20% (multiplication with a factor 0.8).

The temperature increase adopted for scenarios S5 - S7 is based on projections by Lotze-Campen et al. (2009), who predict an increase in surface air temperature by 1.9°C for summer months in Berlin, Germany (time period 2046-2055 compared to time period 1951-2006). Air and water temperature are assumed to develop similarly, as they are highly correlated for shallow and flow-regulated water bodies as the Berlin River Spree (see findings by Livingstone and Lotter (1998) and Kaushal et al. (2010)). Regarding rainfall, no reliable projections were made for Berlin, but generally both an increase and decrease in rainfall intensity can be expected (Grieser and Beck, 2002; Jonas et al., 2005). These uncertainties are reflected in scenarios S6 and S7 for which rainfall intensity of the status quo (year 2007) is scaled down and scaled up by 20%, respectively. Please note, that the climate change scenarios do not take into account hydrological or biogeochemical side effects outside the model boundaries (e.g. changes in river discharge or phytoplankton dynamics).

CSO management scenarios

The effect of the different CSO management scenarios on CSO emissions and river impacts is given in Figure 4.3 for an exemplary rain event of 26 mm depth. For the shown event, the sewer rehabilitation measures planned until 2020 (S2) lead to a decrease in CSO volume by 12% and in BOD_5 load by 15% compared to the sewer status quo 2010 (S1). Thanks to this decrease, critical DO conditions at the observed river stretch can be prevented for this event. A further increase of the storage volume by 20% (S3) allows to reduce 7% of discharged volume and 10% of BOD_5 loads compared to the reference scenario S2, which results in another slight increase of the minimum DO concentration. A significant improvement of oxygen conditions in the river can be achieved through a reduction of the impervious area by 20% (S4), preventing 35% of the CSO volume and 37% of the BOD_5 loads from being discharged into the river.

The effect of CSO control measures on the occurrence of suboptimal DO conditions that imply any kind of adverse conditions is negligible (Figure 4.3, right upper panel). They are primarily caused by low background levels of DO in combination with low flow velocities and high temperatures, in line with findings by Riechel (2009). In contrast, the frequency of highly critical oxygen conditions potentially leading to fish kills can be reduced by one third due to the planned sewer rehabilitation measures and by another third due to a 20% reduction of the impervious area (Figure 4.3, right lower panel). Table 4.1 summarises simulation results for CSO emissions and river impacts for the scenario year.

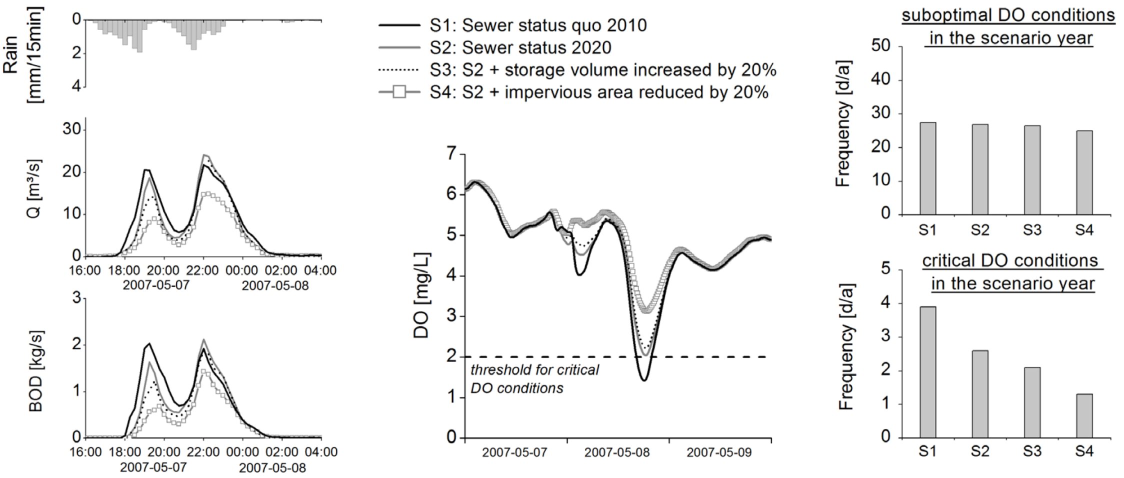

Figure 4.3: Simulated CSO volumes and BOD_5 loads for all 67 CSO outlets for an exemplary rain event (left panel), resulting DO concentration at a highly CSO impacted stretch of the Berlin River Spree (middle panel) and the frequency of suboptimal and critical DO conditions for the scenario year 2007 (right panel). The four different line types represent different CSO management scenarios.

Climate change scenarios

The increasing pressure due to climate change was analysed based on predicted changes in temperature and summer rainfall intensity. Figure 4.4 shows the simulated CSO volumes, pollutant loads and river impacts for the same rain event as above, visualising the effect of an increase in temperature and changes in rainfall intensity (both increase and decrease with regards to uncertainties in climate change predictions).

Figure 4.4 indicates that a temperature increase by 1.9°C (S5) has no direct effect on simulated CSO emissions. However, it negatively influences most processes that are relevant for the oxygen budget in the receiving river such as the oxygen flux into the sediment, oxygen production by algae or reaeration from the atmosphere. Moreover, the oxygen demand of fish increases with higher temperature. As a result, suboptimal oxygen conditions are simulated for 45 days of the scenario year, 18 days more than for the reference scenario S2.

While the higher temperature leads to an elevated background oxygen stress, changes in rainfall intensity have a particular effect on event-based impacts of CSO. For the event shown in Figure 4.4, a 20% higher rainfall intensity (S6) would augment CSO volumes and BOD_5 loads by 34% and 24% respectively, with significantly lower DO concentrations in the receiving river. The contrary effect is found for scenario S7 with 20% lower rain intensities, for which 35% of CSO volume and BOD_5 load can be prevented from entering the water body.

While the occurrence of suboptimal DO conditions is only slightly affected by changes in rainfall intensity, highly critical DO conditions potentially leading to fish kills are much more frequent if a 20% higher rainfall intensity is assumed (scenario S6). These findings indicate that in the negative case, climate change could negate the positive effect of implemented sewer rehabilitation measures (compare scenarios S1 with S6 in Table 4.1). In contrast, a 20% decrease in rainfall intensity (S7) would reduce the frequency of critical DO conditions by more than one third and therefore make measures more effective. The results of the conducted scenario analysis are summarized in Table 4.1.

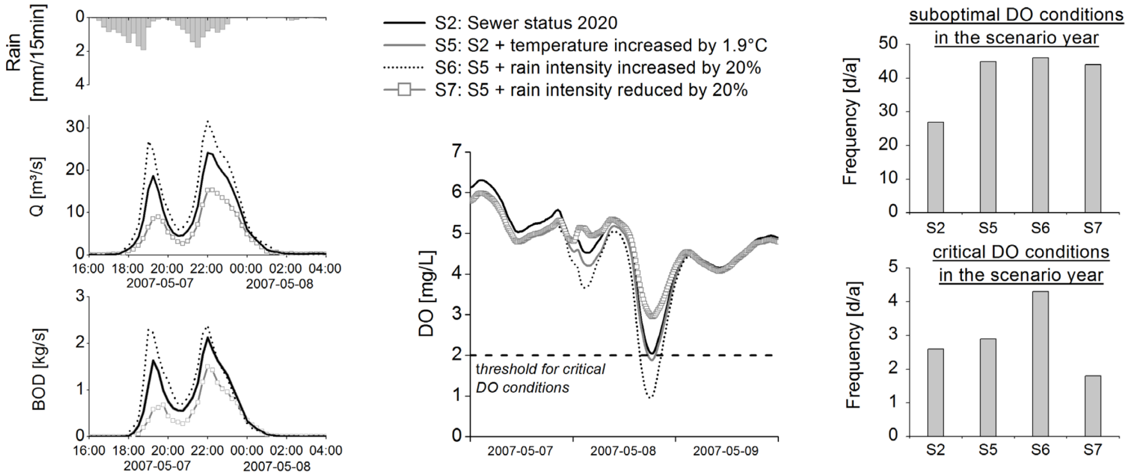

Figure 4.4: Simulated CSO volumes and BOD_5 loads for all 67 CSO outlets for an exemplary rain event (left panel), resulting DO concentration at a highly CSO impacted stretch of the Berlin River Spree (middle panel) and the frequency of suboptimal and critical DO conditions for the scenario year 2007 (right panel). The different line types represent different climatic conditions.

Table 4.1: Summary of scenario results for the simulation period April to November 2007.

	S1: Sewer status quo 2010	**S2:** Sewer status 2020	**S3:** S2 + storage capacity increased by 20%	**S4:** S2 + impervious area reduced by 20%	**S5:** S2 + temperature increased by 1.9°C	**S6:** S5 + rain intensity increased by 20%	**S7:** S5 + rain intensity reduced by 20%
CSO volume [10^6 m³]	5.9	4.9	4.6	3.3	4.9	6.6	3.3
BOD_5 loads [t]	356	273	247	182	273	331	206
TSS loads [t]	840	663	605	442	663	801	503
NH_4-N loads [t]	12.2	8.4	7.3	5.7	8.4	10.9	5.7
CSO events *	34	32	32	29	32	34	29
DO suboptimal [d]	27.5	26.9	26.5	25.0	45.0	46.1	44.0
DO critical [d]	3.9	2.6	2.1	1.3	2.9	4.3	1.8

* with a total CSO volume of ≥10,000 m³.

4.3 Conclusions and recommendations

A planning instrument for integrated and impact-based CSO control under climate change conditions has been developed and demonstrated in Berlin. It enables to predict the impact of climate change on the sewer system or the river and assess the benefit of possible CSO control strategies (e.g., the construction of new storm water tanks). In the future, the instrument will be used for planning specific measures in the sewer system or on the catchment's surface.

Model validation shows that results of the coupled sewer-river-model fit well with measurements, both at dry weather as well as for CSO impacted periods. The performed scenario analysis further indicates that the model tool reacts sensitively to changes in boundary conditions. Thus, the planning instrument is a valuable tool to assess the relative effect of changes in storage capacity, the impervious area, air and water temperature or rainfall intensity.

The scenario analysis conducted with input data of the year 2007 shows that the sewer rehabilitation measures planned until the year 2020 can significantly reduce overall CSO volumes (−17%) and pollutant loads (−23% for BOD_5) and consequently improve oxygen conditions in the receiving water body (−33% calendar days with critical oxygen conditions). For a further improvement of water quality, the local infiltration of storm water appears to be very effective. However, expected climate change has a negative impact on the overall oxygen budget and can significantly increase the occurrence of acute oxygen depressions after CSO, so that the expected improvement through sewer rehabilitation measures could be counterbalanced.

Already during the development of the planning instrument, the end-users (the environmental authority, the operating water utility and a local engineering consultant) were closely involved. The operating water utility provided the model of the Berlin combined sewer system and conducted the sewer simulations for the scenario analysis. The environmental authority provided continuous river quantity and quality data for model calibration and validation. All end-users took an active role in scenario development and all other project decisions. They tested a beta-version of the planning instrument and gave valuable feedback to improve its practicability.

Recently the impact-based planning instrument has been transferred to the end-users in form of a DVD, supporting them in finding optimal CSO management strategies under a changing climate. It will help to effectively allocate investments in the wastewater infrastructure and maximise the environmental benefit of measures. The basic structure of the planning instrument can also be adopted by other cities, requiring adaptation of the selected modelling and impact assessment approaches to the local conditions.

Chapter 5: Sustainable urban drainage systems for CSO impact mitigation

This study was published as:

Riechel, M., Matzinger, A., Pallasch, M., Joswig, K., Pawlowsky-Reusing, E., Hinkelmann, R., Rouault, P. (2020). Sustainable urban drainage systems in established city developments: Modelling the potential for CSO reduction and river impact mitigation. *Journal of Environmental Management* 274, 111207. https://doi.org/10.1016/j.jenvman.2020.111207

This is the postprint version of the article. The final publication is available at SienceDirect via https://doi.org/10.1016/j.jenvman.2020.111207.

Abstract

Sustainable urban drainage systems (SUDS) can significantly reduce runoff from urban areas. However, their potential to mitigate acute river impacts of combined sewer overflows (CSO) is largely unknown. To close this gap, a novel coupled model approach was deployed that simulates the effect of realistic SUDS strategies, developed for an established city quarter, on acute oxygen depressions in the receiving river. Results show that for an average rainfall year the SUDS strategies reduce total runoff by 28%–39% and peak runoff by 31%–48%. Resulting relative reduction in total CSO volume ranges from 45%–58%, exceeding annual runoff reduction from SUDS by a factor of 1.5. Negative impacts in the form of fish-critical dissolved oxygen (DO) conditions in the receiving river (< 2 mg DO L^{-1}) can be completely prevented with the SUDS strategies for an average rainfall year. The realistic SUDS strategies were compared with a simpler simulation approach which consists in globally downscaling runoff from all impervious areas. It indicates that such a simple approach does not completely account for the positive effect of SUDS, underestimating CSO volumes for specific rain events by up to 13%. Accordingly, global downscaling is only recommended for preliminary planning purposes.

5.1 Introduction

Diffuse pollution from urban drainage systems and in particular from combined sewer overflows (CSO) can seriously impair the ecological quality of receiving water bodies. Long-term negative impacts range from sediment contamination of river beds (Wu et al., 2019) to eutrophication in the downstream sea (Barone et al., 2019). Of special concern, however, are short term impacts such as hydraulic stress (Kominkova et al., 2017), ammonia toxicity (Krejci et al., 2004b), and acute oxygen depressions (Riechel et al., 2016) which can lead to lethal conditions for aquatic organisms.

A large potential for mitigating these impacts is seen in the implementation of sustainable urban drainage systems (SUDS), e.g. green roofs, infiltration swales or other nature-based solutions (Dietz, 2007; Ahiablame et al., 2012; Eckart et al., 2017). SUDS retain stormwater and attenuate runoff peaks (Palla et al., 2017; Hou et al., 2019; Liu and Chui, 2019), thus reducing hydraulic loadings to drainage systems. Some SUDS also remove specific pollutants associated with stormwater runoff, e.g. suspended solids, phosphorus or heavy metals (Dunne et al., 2012; Drake et al., 2014; Lim et al., 2015). These positive effects are often combined with additional benefits, such as augmentation of biodiversity (Pille and Saeumel, 2017; Monberg et al., 2019) or mitigation of urban heat islands (Norton et al., 2015; Rosso et al., 2019). By that, SUDS can make cities more liveable and more resilient to climate change impacts, addressing the sustainable development goals of the United Nations (2015).

Different modelling approaches exist that allow to simulate hydraulic effects of single or combined SUDS, ranging from simple tool boxes (Liu et al., 2015; Fu et al., 2019) to more complex hydrodynamic rainfall-runoff models (Paule-Mercado et al., 2017; Hamouz and Muthanna, 2019). A review is given by Jayasooriya and Ng (2014). These applications usually simulate the retention, attenuation and delay of runoff via SUDS at small spatial scale, e.g. building or city quarter level. However, to the best of our knowledge, none of these approaches scales up these effects and propagates them to the receiving river to quantify the potential to mitigate CSO impacts.

On the other hand, there are modelling approaches that simulate the impacts of stormwater management on river water quality at catchment or city scale. However, these classical approaches usually lack a detailed representation of SUDS and only simulate global relative reductions of the impervious area without spatial differentiation (Borris et al., 2013; Riechel et al., 2016; Wang et al., 2019). Such global approaches can give insights into the general reaction of the entire system but neglect (i) the dynamic nature of the hydraulic performances of the individual SUDS, (ii) the spatially diverse feasibility of SUDS implementation and (iii) the acceptance of the extent and type of SUDS by authorities and water utilities. As a result, it is questionable whether these global model approaches provide an adequate estimate of the mitigation potential and therefore a reasonable basis for strategic planning of SUDS for urban river rehabilitation.

The present study aims at assessing a realistic impact of SUDS on river water quality after CSO, taking the above points (i) to (iii) into account, and at comparing this "realistic impact" with a classical approach based on a global reduction of the impervious area. In particular, the following questions are discussed:

- What is the potential of realistic SUDS strategies to reduce CSO emissions and mitigate acute impacts in a receiving water body?
- To which extent would surface runoff have to be reduced to achieve a specific effect on CSO and reach defined water quality goals?
- How different are realistic SUDS strategies from simpler approaches using a global reduction of impervious area in terms of simulated CSO emissions and which model complexity is needed for strategic impact-based planning?

To answer these questions, a detailed rainfall-runoff model, including components for a multitude of SUDS, was combined with a sewer model and a water quality model of the receiving river. The analysis is conducted for a 17 km^2 established city area in Berlin, Germany, which is drained by a combined sewer system and where occasional fish-kills are observed in the receiving river after CSO. For this area - scaled up from a selected city quarter - detailed SUDS strategies were developed on the basis of stakeholder participation and layout by engineers and urban planners. Impact assessment focusses on the mitigation of acute oxygen depressions, the main stressor for the local aquatic biocenosis (Riechel et al., 2010).

5.2 Material and methods

5.2.1 Study area

SUDS strategies were developed and dimensioned for the city quarter “Alt-Schöneberg” (total area: 104 ha, imperviousness: 70%) located in the city of Berlin and characterised by mainly residential land use (Figure 5.1). The city quarter has 16,505 inhabitants. Its surface is composed of 26% roofs, 32% streets (including sidewalks), 13% yards and 29% green structures. Runoff rate for an average rainfall year is 48% (see Figure A.8, Appendix).

The studied city quarter “Alt-Schöneberg” is part of Berlin’s biggest combined sewer catchment “Wilmersdorf” (Figure 5.1). The catchment drains stormwater from a total area of 1,651 ha (imperviousness: 56%) and sanitary sewage of ~265,000 inhabitants by gravity to a main pumping station. Water is pumped to the wastewater treatment plant at a maximum capacity of 1.5 m^3 s^{-1}, which is twice the peak dry weather flow. Total daily dry weather volume is ~40,000 m^3. The diameter of the sewer pipes ranges from 80 mm to 4200 mm (median: 300 mm). Median pipe slope is 0.4%. The sewer network disposes of four storage tanks and has three CSO outlets to the river. The specific storage capacity of the tanks and the sewer pipes is 41 m^3 ha^{-1}. Annual CSO volume is ~700,000 m^3 for an average rainfall year (see Table A.6, Appendix).

The CSO outlets of the combined sewer catchment discharge into the “Landwehrkanal”, a side-channel of the Berlin River Spree (Figure 5.1). The Landwehrkanal is a regulated lowland river with a length of 11 km, a mean discharge of 3 m^3 s^{-1} and an average flow speed of 9 cm s^{-1}. The main observed CSO impacts are deficits in dissolved oxygen (DO) leading to sporadic fish kills during summer months. Critical oxygen conditions, quantified as described in section 5.2.3, are observed on at least five days per year.

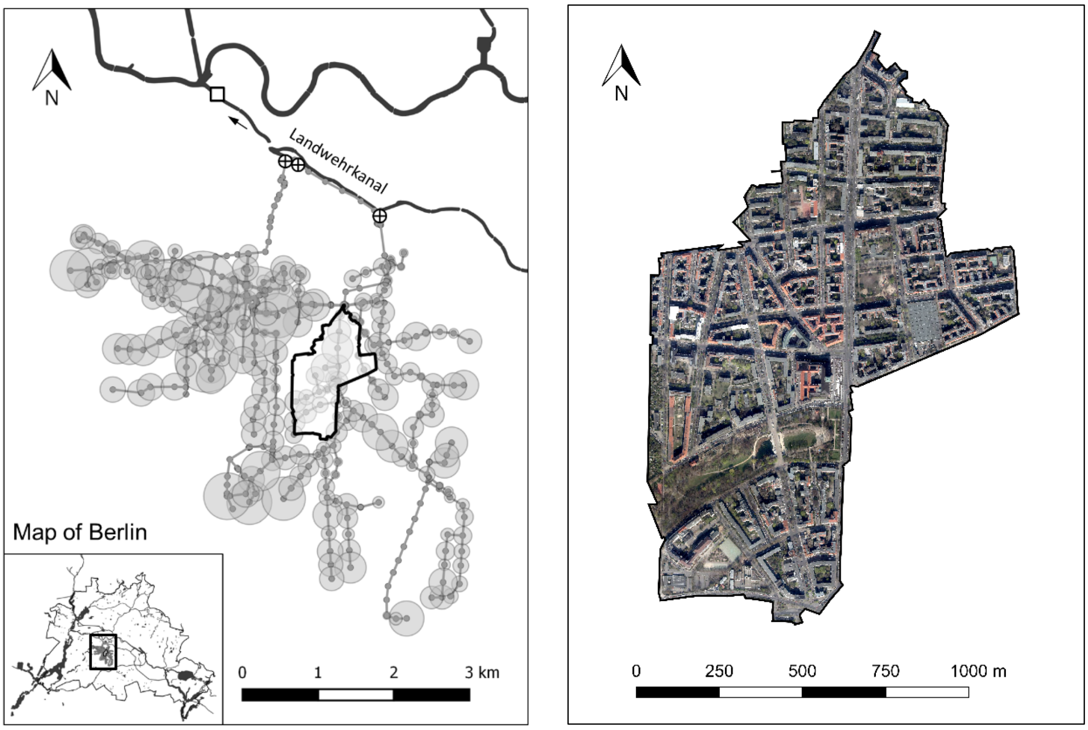

Figure 5.1: Left: Schematic map of the study area consisting of i) the city quarter "Alt-Schöneberg" (semi-transparent with black border), ii) the combined sewer system "Wilmersdorf" with its subcatchments (light grey, as represented in the sewer model, see section 5.2.4) and iii) the receiving river "Landwehrkanal" (dark grey, labelled). Crossed circles are the CSO outlets to the river, the arrow shows flow direction and the white-filled square indicates the point at which river impacts were assessed. Right: Aerial image of the studied city quarter "Alt-Schöneberg", adapted from Geoportal Berlin.

5.2.2 Stormwater strategies and runoff scenarios

Realistic SUDS strategies

Three different SUDS strategies were developed for the city quarter "Alt-Schöneberg" to demonstrate their potential for CSO impact mitigation (and other environmental and human health effects not part of this study). The strategic planning process was organised in form of a workshop and involved representatives of the local authorities, water utilities, researchers, engineers and urban planners. Workshop participants were divided into three heterogenous groups A to C that independently developed one SUDS strategy each.

For each strategy A, B and C, SUDS were selected from a portfolio of 19 established solutions including green roofs, green facades, stormwater harvesting, infiltration basins and swales, tree-trenches, and artificial ponds, among others. Suitable SUDS were selected and placed in the city quarter after cross-checking their individual effects, costs, and technical feasibility against the local deficits and goals defined by the authorities (e.g. protection of surface waters or increase of biodiversity). The required information was derived from surveys, extensive literature reviews and own measurements, the latter taken during one-year monitoring campaigns (e.g. Schubert et al., 2015). Technical feasibility of SUDS was accounted for by considering spatial data on roof slopes (in case of green roofs) and field capacity of soils (in case of infiltration swales), as two examples. SUDS were finally implemented in numerical models in agreement with German design standards. A more detailed description of the fictive planning process is presented by Matzinger et al. (2017b) and Nickel et al. (2016).

The resulting strategies cover a total of 13 different SUDS with a strong focus on extensive green roofs, green facades, stormwater harvesting and tree-trenches. Also, combinations of SUDS are employed, e.g. with an extensive green roof connected to an infiltration swale. Table 5.1 gives an overview of the three SUDS strategies, which show differences regarding the total connected area and the types of deployed SUDS (see Appendix, section A, for more details).

It is important to note that the three strategies were essentially developed for demonstration purposes as part of a simulation game and, up to now, are not planned to be fully implemented. Nevertheless, they are considered as ambitious but realistic.

Table 5.1: Characteristics of the three SUDS strategies

	Strategy A	Strategy B	Strategy C
Total area connected to SUDS (portion of city quarter's impervious area)	21.6 ha (30%)	32.6 ha (45%)	31.9 ha (44%)
Top 3 SUDS ranked by their respective connected area *	Green facades (9.8 ha), Tree-trenches (8.0 ha), Extensive green roofs (7.7 ha)	Stormwater harvesting (17 ha), extensive green roofs (9.7 ha), tree-trenches (6.7 ha)	Green facades (11 ha), extensive green roofs (7.6 ha), stormwater harvesting (6.2 ha)

* Some surface elements are connected to two SUDS in series and are therefore counted twice

Global runoff reduction scenarios

In addition to the elaborate SUDS strategies that require a complex rainfall-runoff model with a high spatial resolution, a simpler approach in the form of global runoff reduction scenarios was investigated. For these scenarios, which were directly derived from the status quo simulation without implemented SUDS, runoff from all surface elements was proportionally scaled down in 10% steps down to 0% runoff. By that, a general reduction in the catchment's impervious area is simulated without spatial differentiation and without considering the dynamic effect of SUDS on surface runoff.

The scenarios were used to i) highlight the general relationship between runoff reduction and CSO volumes (linear, exponential, etc.), ii) estimate to which extent surface runoff would have to be reduced to achieve a certain effect in the river and iii) verify if computationally expensive modelling of individual SUDS gives more realistic information on CSO reduction potential than global runoff reduction scenarios. The latter was done by comparing the two approaches regarding their simulated CSO volume for a given total runoff reduction.

5.2.3 Assessment metrics

The performance of the SUDS strategies was assessed at three scales:

- at city-quarter level via the reduction in total runoff and peak runoff rate (after combining runoff from the different surface elements),
- at sewer catchment level via the reduction in CSO frequency, CSO volume and BOD_5 pollutant loads (after combining emissions from the different CSO outlets), and
- in the receiving river via the reduction in the frequency of fish-critical oxygen conditions, simulated 1.5 km downstream of the CSO outlets.

Frequency of fish-critical oxygen conditions was quantified as the number of calendar days with DO concentrations below the lethal concentration LC_{50} of the most sensitive endemic fish species, the asp, which is 2 mg DO L^{-1} (Wolter et al., 2003). Improvements beyond that critical value and effect variance between rainfall events were investigated via the absolute increase in minimum DO concentration after CSO. All indicators were quantified for a representative rainfall year (see section 5.2.5) with regards to the status quo.

For the global runoff reduction scenarios and their comparison with the SUDS strategies, rainfall, runoff, and CSO volumes were evaluated for the individual events of the representative rainfall year, separated with a six-hour dry weather period. Volumes pumped to the wastewater treatment plant were used to calculate water balances and determined for a time interval starting one hour before and ending six hours after the CSO event. The impact on river water quality was assessed via the total duration of DO below thresholds of 2, 3 and 4 mg L^{-1}.

5.2.4 Model setup

The developed modelling approach for the assessment of stormwater strategies consists of different sub-models for rainfall-runoff, the sewer system, and the receiving river. The chosen approach extends the coupled sewer-river model by Riechel et al. (2016) with a detailed rainfall-runoff component which is able to represent SUDS. The extended model approach was applied for the realistic SUDS strategies. In turn, the approach by Riechel et al. (2016) without SUDS representation was directly applied for the global runoff reduction scenarios.

Rainfall-runoff model

The deterministic hydrological rainfall-runoff model STORM (IPS, 2007) was used to simulate surface runoff and associated pollutant loads for the studied city quarter. It was chosen since it has a detailed SUDS representation in accordance with German design standards and covers all main components of the hydrological cycle (e.g. runoff, infiltration, evapotranspiration), required to calculate water balances and assess other effects of SUDS not part of this study (e.g. on groundwater recharge).

The model setup for the studied city quarter consists in 2,463 surface elements, distinguished into four surface types. Runoff from impervious areas (surface types “roof” and “street”) is simulated via interception and depression storages as well as two different runoff coefficients effective before and after filling of the depression storage. By that it is possible to simulate runoff already before depression storages are completely filled. The parameters can be adjusted to mimic the behaviour of different slopes and roof materials, on the one hand, or different pavement characteristics and vegetative street covers, on the other hand. The annual vegetation pattern is mimicked by dynamic interception storage. Hydrological behaviour of pervious areas (surface types “yard” and “green structure”) is computed based on a soil water balance model which considers natural soil characteristics (soil type, hydraulic conductivity, field capacity, etc.) and density of vegetation. Long-term series of temperature, relative humidity, wind speed, and sunshine-duration are used to calculate actual evapotranspiration according to Penman-Monteith (Monteith, 1965). The simulation time step is 5 minutes.

Pollutant wash-off is simulated in the form of constant concentrations for each of the four surface types as derived from literature (see Table A.4, Appendix). Considered pollutants are total suspended solids (TSS), the chemical and biological oxygen demand (COD, BOD_5), total phosphorus (TP), total nitrogen (TN), and ammonium nitrogen (NH_4-N).

Hydraulic retention and pollutant removal capacity of the individual SUDS was derived from an extended literature review as well as own measurements in case of green roofs and retention soil filters (in total > 350 datasets, overview in Matzinger et al. (2016)). An example of runoff reduction potential of the considered SUDS is given in Figure A.2 (Appendix).

Sewer model and upscaling of runoff

The hydrodynamic pollutant load model InfoWorks CS (WSL 2004) was used to simulate flow and pollutant transport in sewer pipes, as described in Riechel et al. (2016). Flow and pollutant loadings originating from sanitary sewage are simulated via hydrographs and pollutographs that represent the typical daily distribution of wastewater quantity and quality during dry weather. Stormwater runoff and associated pollutant concentrations are simulated with the rainfall-runoff model STORM and then passed to the sewer model as inflow to the manholes. The sewer model also simulates effects of existing sewer-based control measures, e.g. variable weirs or stormwater tanks, on hydraulics and pollutant concentrations. Degradation processes in the sewer are not considered. The simulation time step is 5 minutes.

A conceptual model of the catchment’s sewer network was provided by the Berlin water utility and re-calibrated for this study. It includes 257 lumped subcatchments, 390 main sewer pipes and 385 manholes. To account for the diverging scales of the rainfall-runoff model (city quarter scale, 73 ha impervious area) and the sewer model (catchment scale, 921 ha impervious area), runoff was scaled up at the interface of the two models. The upscaling of runoff and SUDS effects from the city quarter to the sewer catchment is crucial for the simulation of CSO emissions and existing sewer-based control measures. Details on the upscaling method are given in section B of the Appendix.

River water quality model

Impacts of CSO in the receiving river were simulated with the hydrodynamic river water quality model Hydrax-QSim, version 13.01 (Kirchesch and Schöl, 1999; Oppermann et al., 2015), as described in Riechel et al. (2016). It simulates one-dimensional longitudinal river flow and reactions of all major water quality parameters, including the above mentioned (TSS, COD, BOD_5, TP, TN, NH_4-N) as well as different species of phytoplankton and zooplankton. CSO discharges simulated with the sewer model are incorporated as river inflows. The simulation time step is 15 minutes.

Among the included processes, phytoplankton production, atmospheric exchange, nitrification, species respiration and especially the decomposition of organic matter are of primary importance for the oxygen budget of the river. For the latter, Hydrax-QSim considers two steps of hydrolysis: i) from particulate to dissolved and ii) from dissolved to monomeric organic carbon, which is then consumed by explicitly simulated heterotrophic bacteria under the use of dissolved oxygen. A model representation of the receiving river existed from a previous study (Riechel et al., 2016).

Model coupling, calibration and validation

The three sub-models are coupled on an output-input basis with a unidirectional flow of information. Data transfer is realised via comma separated text files, applying subroutines for data formatting and time step aggregation (from 5 to 15 minutes between the sewer and river water quality model).

Hydraulic calibration and validation of the coupled rainfall-runoff and sewer model was conducted on basis of water level measurements in the sewer network, specifically at the main pumping station, two overflow crests and two stormwater tanks. For calibration, runoff parameters for impervious and pervious areas as well as the dry-weather hydrographs were used as tuning variables. Model validation, carried out for rainfall events of different intensities, yielded Nash-Sutcliffe efficiencies (Nash and Sutcliffe, 1970) between 0.64 and 0.96 for the different locations (mean: 0.86), indicating a good to very good agreement between simulations and measurements (Moriasi et al., 2007). The comparison of simulations and measurements for an exemplary rainfall event is visualised in Figure A.4 (Appendix).

Regarding water quality, the dry-weather pollutographs of the sewer model were calibrated based on two-hour mixed water quality samples taken during five different days at the main pumping station. Pollutant concentrations in stormwater runoff were derived from a literature review (see Table A.4, Appendix). For validation of the coupled rainfall-runoff and sewer model, simulated pollutant concentrations in CSO (TSS, COD, BOD_5, TP, TN, and NH_4-N) were compared with literature data (Brombach and Fuchs, 2003; Caradot et al., 2011; Gasperi et al., 2012) and unpublished data by the Berlin water utility. Here, annual mean concentrations were in the range of observations, except for TN which is of minor importance for acute DO deficits in the river (Figure A.5, Appendix).

Validation of the coupled sewer and river water quality model was done in the framework of a previous study (Riechel et al., 2016), yielding Nash-Sutcliffe efficiencies between 0.61 and 0.70 for the goal variable dissolved oxygen (DO).

5.2.5 Rainfall scenario and other model input data

The developed SUDS strategies and runoff reduction scenarios were analysed for a one-year representative rainfall series for the local climate conditions (year 1990, rain gauge Berlin-Dahlem, 5-minute time step, see section D of Appendix for details). A long-term simulation for e.g. 30 years was not feasible due to the enormous computational effort, especially regarding river impact modelling.

Hydraulic and weather input data for the river water quality model, e.g. upstream river flow, downstream water levels, wind speed, and global radiation, were taken from the same year to correspond with the rainfall period. Regarding river water quality, more recent measurement data were used to account for the current water quality conditions. Here, measured concentrations of TSS, COD, BOD_5, TP, TN, and NH_4-N as well as temperature and phytoplankton measurements were used at a monthly interval to account for the typical seasonal variations.

5.3 Results and discussion

5.3.1 Evaluation of realistic SUDS strategies

Surface runoff

For the status quo without implementation of SUDS a total runoff of 2949 m^3 ha^{-1} (equivalent to 295 mm) is simulated for the one-year period. The three analysed strategies A to C reduce total runoff by 28% (A), 38% (B) and 39% (C) (Figure 5.3). The strategy with the lowest degree of implementation (strategy A) has the smallest relative effect on total runoff. In general, percentage of runoff reduction is slightly smaller than the percentage of impervious area connected to SUDS (30%, 45% and 44%; Table 5.1), since most measures do not retain 100% of the runoff (Figure A.2, Appendix). Results on the strategies' effect on the urban water balance can be found in section F of the Appendix.

Reductions in peak runoff rates are 31%, 38% and 48% for strategies A, B and C (Figure 5.3), respectively, and thus slightly higher than reductions in total runoff (with the exception of strategy B which has a strong focus on stormwater harvesting). The main reason for this is the attenuating and retarding effect of many SUDS, in particular green roofs and trough-trench infiltration, which positively impact hydraulic loadings to the sewer system.

Figure 5.2 visualises the hydraulic performance of an extensive green roof, through-trench infiltration and stormwater harvesting for an exemplary winter rainfall. Both the extensive green roof and the trough-trench infiltration show an important reduction (by 67% and 87%, respectively) and delay (by 11 and 29 hours, respectively) in peak runoff compared to the reference roof as indicated by the flatter cumulative runoff curves. The positive effect is most pronounced for the trough-trench infiltration, which has a large-dimensioned retention volume. The effect of stormwater harvesting, on the other hand, depends on the cistern volume, the initial filling and the extraction rate and is marginal here (building with a 1219 m² roof, a cistern volume of 30 m³, an initial filling of 90% and an extraction rate of 0.6 m³ d^{-1}).

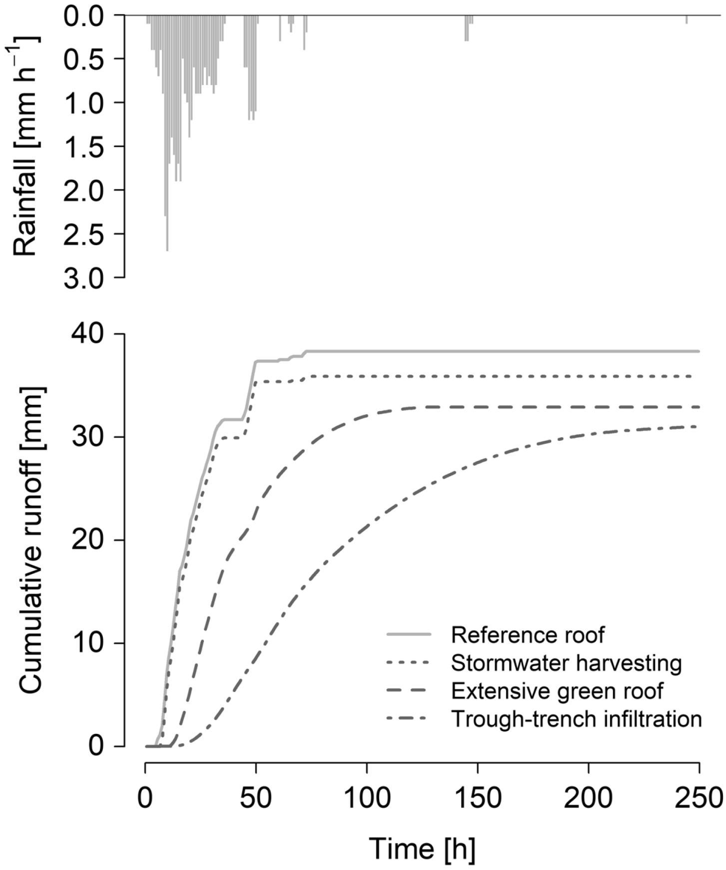

Figure 5.2: Hydraulic performance of selected SUDS and a reference roof (without SUDS) for an exemplary rainfall period in December 1990 (42 mm rainfall with an effective duration of 29 h). Upper panel: rainfall intensity, lower panel: cumulative runoff. The SUDS receive runoff of the reference roof.

For all three strategies the reduction in pollutant loads associated with stormwater runoff is generally smaller than the reduction in volume, since a relatively high portion of roofs with a relatively low degree of pollution is connected to SUDS (see Table A.4, Appendix). As a result, average pollutant concentrations in stormwater runoff are slightly increased for the tested SUDS strategies (e.g. between 12 and 21% for BOD_5). The results indicate that it would be beneficial to primarily implement SUDS downstream of streets with a relatively high degree of pollution, to efficiently reduce overall pollutant loads from stormwater runoff. This is particularly relevant for separate sewer systems where stormwater runoff is directly discharged to the receiving water body without treatment.

CSO emissions

All three stormwater strategies significantly reduce overflow volumes discharged via CSO outlets. Relative reductions are 45%, 57% and 58% for strategies A, B and C (Figure 5.3) and thus 1.5 to 1.6 times higher than reductions in total runoff, which is in line with the findings by Riechel et al. (2016). For single rain events leading to CSO, relative reductions in CSO volume exceed relative runoff reductions by a factor between 1.4 and 4.1 (discussed in detail in 5.3.2).

Pollutant loads of BOD_5, the main driver for DO deficits in the river (Riechel et al., 2012), are reduced by 39% for strategy A and by 53% for strategies B and C (Figure 5.3). For all strategies, pollutant reduction is slightly smaller than volume reduction which can be explained by a marginally higher wastewater ratio in CSO after SUDS implementation and by the fact that a majority of SUDS receives runoff from roofs with a minor degree of pollution.

The number of CSO events is reduced by 47% for strategy A and by 63% for strategies B and C (Figure 5.3), with only 10 or 7 of 19 CSO events remaining. Compared to volume reduction, CSO frequency reduction is slightly more pronounced, as some small events with a minor contribution to the overall volume can be eliminated completely by the SUDS strategies.

River impacts

As a result of the reduction in CSO emissions by around one half, all fish-critical DO conditions in the receiving river are eliminated by each of the three SUDS strategies (Figure 5.3). Lowest simulated DO concentrations for strategies A, B and C are 2.3, 2.7 and 3.1 mg L^{-1} (status quo: 1.0 mg L^{-1}) no longer violating the threshold defined for the protection of the local biocenosis from acute impacts (2 mg DO L^{-1}). Hence, realistic SUDS strategies are able to reduce acute CSO impacts to a tolerable level in an average rainfall year. Nonetheless, it must be assumed that a higher effort would be required to continuously maintain DO above 2 mg L^{-1} and compensate ongoing urbanisation and climate change impacts.

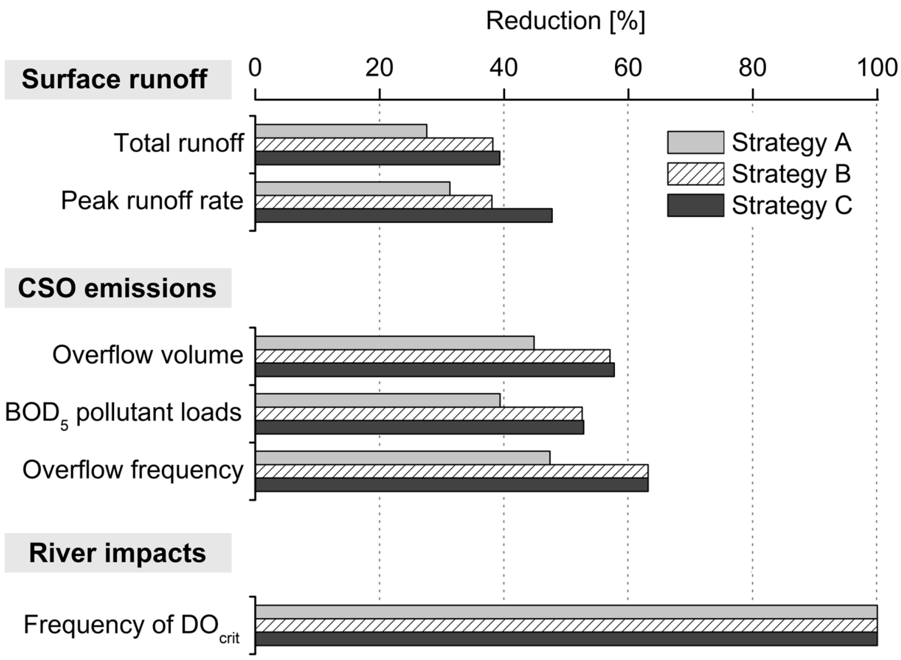

Figure 5.3: Simulated effects of SUDS strategies on surface runoff, CSO emissions and river impacts compared to the status quo.

Although, under the tested boundary conditions, all three strategies comply with the selected threshold for acute river impacts, effects beyond that threshold vary between strategies. Figure 5.4 shows their absolute effect on the minimum DO concentration simulated for each CSO event, distinguishing three rainfall classes (rainfall depth < 10 mm, 10 – 20 mm and > 20 mm). The biggest absolute effects with DO increases of up to 3.8 mg L^{-1} are achieved for strategies B and C, which also yield the biggest reductions in CSO emissions. Strategy C, which is mainly characterised by green roofs and green facades, performs particularly well for summer events when evapotranspiration is high (Berndtsson, 2010).

In general, the effect of SUDS on DO in the river increases with rain depth since absolute reductions of CSO volumes are commonly higher for large rain events (see Figure A.6, Appendix). Besides the rainfall, the antecedent dry weather period (observed range: 0.3 to 12 days) is a major factor for the varying measure effect since it controls the recovery capacity of many SUDS. Comparably small improvements are generally observed for the second of two consecutive rain events when soils are saturated and the retention volume of SUDS is limited. Other important factors for the variance of effects in the river are:

- River water temperature which defines oxygen solubility (Weiss, 1970) and degradation kinetics (Lønborg et al., 2018),
- river water quality upstream of the CSO outlets, e.g. concentrations of DO, organic matter and phytoplankton, which defines background conditions and
- river discharge which defines the dilution capacity for CSO.

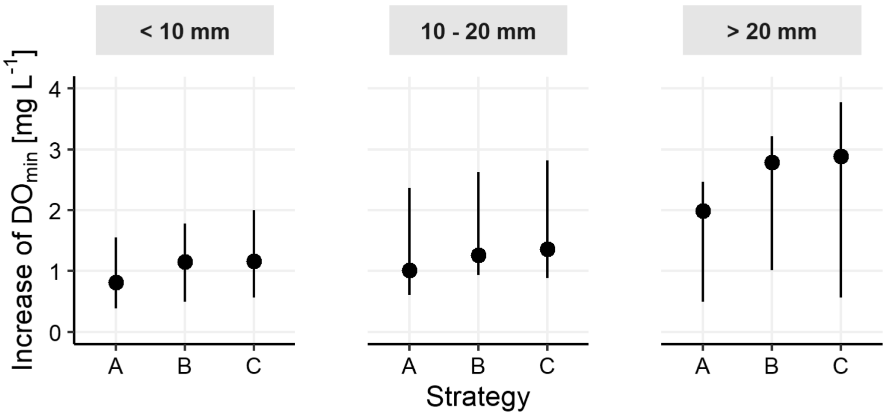

Figure 5.4: Increase of minimum DO concentration for the three SUDS strategies and three classes of rain events: rainfall depth < 10 mm (n = 7, left), rainfall depth: 10 – 20 mm (n = 7, centre) and rainfall depth > 20 mm (n = 5, right). Points show median, whiskers show minimum and maximum values.

5.3.2 Evaluation of global runoff reduction scenarios

CSO emissions

As shown in the previous section, detailed SUDS strategies reduce CSO volumes in general more effectively than surface runoff. The specific relationship between runoff and CSO reduction has been studied at event scale by means of global runoff reduction scenarios. Figure 5.5 (upper panel) shows the discharged CSO volume relative to the status quo situation plotted against the relative runoff reduction for three classes of rain events (rainfall depth < 10 mm, 10 – 20 mm and > 20 mm). For the first 80% of CSO reduction, an almost linear relationship with runoff reduction is observed for all considered rain events. Mean CSO-runoff-reduction slopes for the three rainfall classes are -5.4, -3.9 and -1.9, respectively (dashed lines in Figure 5.5). The required runoff reduction for an 80% decrease in CSO volumes ranges between 10 and 30% for rainfall depths < 10 mm and between 30 and 60% for rainfall depths > 20 mm.

The relatively high rates of CSO volume reduction can be explained with a simple volume balance equation, for didactical reasons ignoring temporal dynamics.

The CSO volume V_{CSO} for a static urban drainage system can be calculated as follows:

$$V_{CSO} = V_{Rain} - V_{WWTP} - V_{Storage} \qquad \textit{(Equation 5.1)}$$

with V_{Rain} being the rainfall volume leading to runoff, V_{WWTP} being the volume pumped to the wastewater treatment plant and $V_{Storage}$ being the volume stored in sewer pipes and storage tanks.

The CSO reduction rate η_{CSO} for a given runoff reduction can be estimated as follows, assuming that V_{WWTP} and $V_{Storage}$ remain constant:

$$\eta_{CSO} = 1 - \frac{V_{CSO\_new}}{V_{CSO}} = 1 - \frac{V_{Rain} \times (1 - \eta_{Runoff}) - V_{WWTP} - V_{Storage}}{V_{Rain} - V_{WWTP} - V_{Storage}} \quad \text{(Equation 5.2)}$$

with V_{CSO_new} being the CSO volume after runoff reduction and η_{Runoff} being the runoff reduction rate.

After transformation of Equation 5.2, the ratio of CSO and runoff reduction rates calculates as follows:

$$\frac{\eta_{CSO}}{\eta_{Runoff}} = \frac{1}{1 - \frac{V_{WWTP} + V_{Storage}}{V_{Rain}}} \geq 1 \quad \text{(Equation 5.3)}$$

For the simplified example, the relationship between the two reduction rates can be expressed as a linear function with a slope ≥ 1. This means that, for a given rainfall, relative CSO reduction is always higher than runoff reduction as long as V_{WWTP} and $V_{Storage}$ remain constant. The larger the rainfall the smaller the slope, i.e. the ratio of the two reduction rates.

The linearity and the reduction of the ratio with larger rain events is also illustrated in Figure 5.5, indicating that non-linear effects are of minor importance for a large range of events and scenarios. The relationship between CSO and runoff reduction only turns non-linear, when storage and pumping capacities remain unexploited while marginal CSO spills persist (shaded areas in Figure 5.5). In that case the simplification of Equation 5.2 is not valid and the CSO reduction rate can be smaller than the runoff reduction rate (slope < 1).

As this is partly the case in the investigated drainage system, a particular effort in runoff reduction is required to eliminate the last 20% of CSO volume (Figure 5.5, upper panel). As a consequence, a 70% to 80% runoff reduction is necessary to avoid all CSO events of the investigated rainfall year.

The fact that CSO events occur without having exploited the full capacity of the pumping station or storage tanks indicates potential for improved sewer management in the studied catchment. An elevation of selected overflow crests or an enhanced real-time control strategy as suggested by Kroll et al. (2018) could help to further increase the efficiency of runoff reduction measures but is beyond the scope of this study.

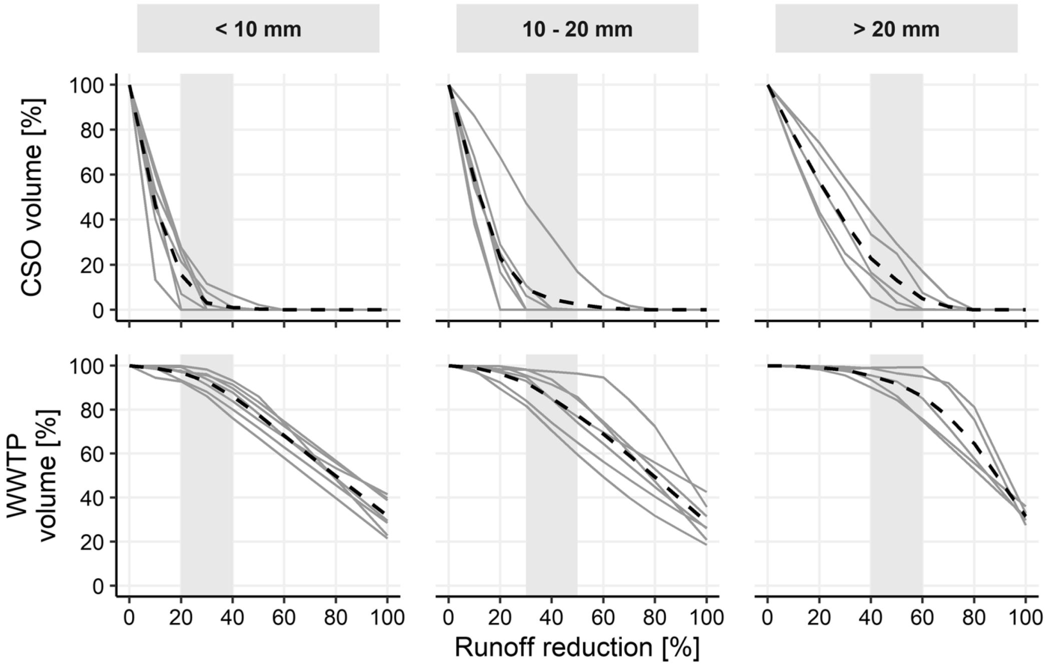

Figure 5.5: Effect of runoff reduction on CSO volume reduction (upper panel) and volumes pumped to the wastewater treatment plant (WWTP, lower panel) for three classes of rain events: rainfall depth < 10 mm (n = 7, left), rainfall depth: 10 – 20 mm (n = 7, centre) and rainfall depth > 20 mm (n = 5, right). Values for CSO volume, WWTP volume and runoff reduction are normalised to the status quo situation. Dashed lines represent mean values for the respective rainfall class. Shaded areas show approximate runoff reduction ranges where pumping capacity decreases while CSO spills remain.

River impacts

Regarding the river perspective, a 30% runoff reduction would be required to maintain DO concentration above a value of 2 mg L^{-1} (Figure 5.6) and to avoid fish kills in the receiving river, which is in line with the results shown above (section 5.3.1). For more ambitious water quality goals of 3 or 4 mg DO L^{-1}, that may support more demanding fish species, a 40% or 70% runoff reduction would be necessary, respectively. In contrast to CSO emissions (Figure 5.5), the reaction in the river (Figure 5.5) is not linearly related to runoff reduction. The reason lies in the different river conditions at the time of CSO (discussed in section 5.3.1), which highlights the necessity of impact-based modelling approaches.

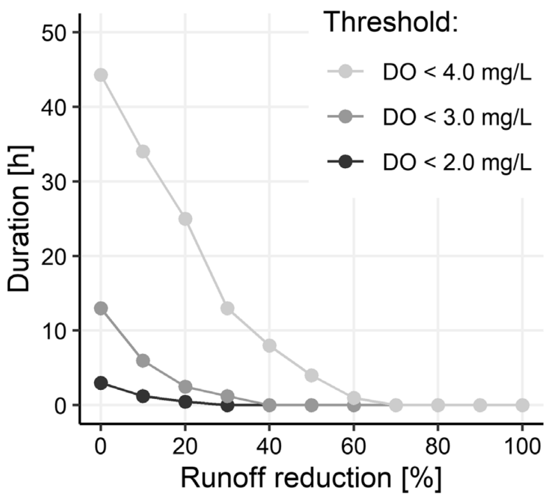

Figure 5.6: Total duration of critical DO concentrations in the receiving river (three different thresholds) for the status quo and runoff reductions from 10% to 100%.

Comparison with realistic SUDS strategies

A comparison of the realistic SUDS strategies and the global runoff reduction scenarios shows that the latter can give a good approximation of the measure effect on CSO emissions. For a given reduction in total runoff, the realistic SUDS strategies only yield marginally higher reductions in CSO volume than the global runoff reduction scenarios (mean: 2%, after linear interpolation of the latter). Exceptions to this are relatively short and intense rainfalls, for which the realistic strategies can outperform simple runoff reduction scenarios by up to 13% CSO volume reduction. Nonetheless, this effect may increase if strategies had a stronger focus on green roofs or other SUDS that attenuate and delay runoff peaks. On the other hand, the global runoff reduction scenarios tend to overestimate performance of SUDS for the second of two consecutive rainfall events, when retention capacities of SUDS are often exhausted. A detailed comparison of the two approaches for the 19 CSO events of the studied rainfall year is given in Figure A.7 (Appendix).

5.4 Conclusion

Three realistic SUDS strategies - developed under consideration of local deficits, feasibility and individual SUDS performance - were analysed regarding their effects on surface runoff, CSO emissions and river impacts. In addition, global runoff reduction scenarios were investigated that range from status quo to a complete retention or separation of stormwater. The following conclusions can be drawn with regards to the three starting questions:

- The investigated SUDS strategies effectively reduce surface runoff by 28% to 39% and CSO volumes by 45% to 58%, exceeding the effect on surface runoff by a factor of ~1.5. Acute CSO impacts in the receiving river are mitigated to a tolerable level by all three strategies. Consequently, SUDS can help to protect aquatic organisms and re-establish populations of indicator species. The difference between the three strategies is comparably small, which implies a large scope of possible actions for stakeholders when it comes to the selection of SUDS.

- A linear relationship with a slope > 1 was found between CSO volume and surface runoff reduction, which highlights the great benefit of any runoff reduction measure. However, to avoid all CSO events for a regular rainfall year and maintain the DO concentration in the river above a level of 4 mg L^{-1}, a huge effort, namely a 70% to 80% runoff reduction, would be necessary. For even more ambitious water quality goals, additional measures upstream of the combined sewer system would have to be taken.
- A simple simulation approach based on a proportional reduction of runoff instead of a detailed SUDS representation gives a generally good estimation of the potential of stormwater management for CSO prevention. However, it underestimates the CSO reduction potential of SUDS by up to 13% for individual rain events. Further, such global runoff reduction scenarios ignore the feasibility of measures and the acceptance of stakeholders. As a consequence, they are only recommended for preliminary planning purposes or the definition of general runoff reduction goals.

Although these findings fill an important knowledge gap, in particular for planners and local authorities, the present study has some limitations:

- Firstly, SUDS effects were scaled up from a specific city quarter to a larger sewer catchment without again verifying technical feasibility of SUDS for all surface elements. However, this simplification can be justified with a relatively homogenous land use and building structure found within the catchment's boundaries. Secondly, calibration and input data uncertainties, e.g. due to the limited spatial resolution of rainfall data and the assumption of constant, site-specific pollutant concentrations in surface runoff, must be mentioned. Nevertheless, model validation confirms a generally good agreement with measured data.
- Another shortcoming of this study is that no reliable statement on the variability of SUDS effects among different rainfall years could be made, given the high computational costs of the chosen modelling approach. As an alternative, a much simpler statistical emulation model could be established for a first estimation of CSO emissions, using the linearity found between runoff and CSO reduction for a range of rain events. Nonetheless, a detailed river water quality model as used for this study is essential to cover the non-linear nature of river impacts.

Acknowledgements

The presented results were established within the project KURAS, mainly funded by the German Ministry of Education and Research (BMBF, grant number 033W013B) in the framework of the FONA (Research for Sustainable Development) initiative. The project is co-financed by contributions of the participating SMEs and large-scale enterprises. KWB received co-financing from Veolia Water and Berliner Wasserbetriebe. The deployed software for river water quality modelling was thankfully provided by the German Federal Institute of Hydrology (BfG).

Chapter 6: Identifying CSO hotspots for pathogen emissions

This study was published as:

Riechel, M., Seis, W., Matzinger, A., Pawlowsky-Reusing, E., Rouault, P. (2019). Relevance of Different CSO Outlets for Bathing Water Quality in a River System. Book chapter in: New trends in Urban Drainage Modelling: UDM 2018, 859-863. Editor: Giorgio Mannina, Springer. https://doi.org/10.1007/978-3-319-99867-1

This is the postprint version of the book chapter.

Abstract

Combined sewer systems are one of the major sources of microbiological contamination in urban water bodies. However, identification of hotspots for pathogen emissions is not straightforward, especially in large and complex drainage systems. To determine the relevance of different CSO outlets for bathing water quality a simple tracer approach which uses wastewater volume as a proxy for pathogen emissions has been developed and tested for the city of Berlin, Germany. The approach reveals that the average wastewater ratio in CSO varies largely between different river outlets (0 to 15%). Hence, the outlets with the largest CSO volumes are not automatically the greatest wastewater emitters and assumed hotspots for pathogen contamination do not coincide with hydraulic hotspots. This is verified with own measurements that show enormous differences in pathogen concentrations between waste and stormwater of 4 orders of magnitude. As a result, wastewater which represents only 5% of the CSO volume contributes > 99% of the pathogen loadings to the river. The study highlights the relevance of wastewater volumes for the identification of point sources for the hygienic impairment of water bodies.

6.1 Introduction

The revised Bathing Water Directive (EU, 2006) requests that member states take appropriate measures to achieve at least sufficient bathing water quality at European bathing sites. To comply with the regulation and allocate measures effectively it is necessary to first identify the sources of microbiological contamination.

Different authors found that combined sewer overflows (CSO) are one of the major reasons for hygienic impairment of urban water bodies (Ellis and Wang, 1995; Rechenburg et al., 2006; Passerat et al., 2011). However, it is not straightforward to find the actual hotspots of pathogen emissions, especially in large combined sewer systems with a multitude of different CSO outlets. This is assumed to be partly due to the enormous differences in pathogen concentrations between waste and stormwater and the uneven distribution of wastewater flows in many combined sewer networks. The latter is particularly the case when parts of the combined sewer system also receive stormwater runoff of the separate sewer system as in Berlin.

This paper presents a simple tracer approach to identify the relevance of different CSO outlets for bathing water quality in the urban river system of Berlin, Germany. The tracer approach is implemented in a hydrodynamic model of the city's combined sewer system and is verified with measurements of *Escherichia coli* and *somatic coliphages*.

6.2 Material and Methods

6.2.1 Study site

The combined sewer system (CSS) of the city of Berlin covers an impervious area of 66 km² and drains wastewater of 1.5 million inhabitants. The CSS has a total length of approx. 2000 km and is connected to the river system via 176 CSO outlets, discharging in case of heavy rainfall. The CSO outlets are located at the River Spree, its side-channels and the Lower Havel along a total distance of ~ 50 km. While the River Spree and its side channels are mostly used for shipping and recreation, the Lower Havel has different declared bathing sites ~ 5 km downstream of the CSS. The typical flow time for that river stretch is 1 to 2 days.

6.2.2 CSO emission model

A hydrodynamic model of the combined sewer system was set up in the software InfoWorks and initially calibrated by Pawlowsky-Reusing et al. (2006). Since then the model was constantly refined and recently validated by Riechel et al. (2016). The model consists of 3210 subcatchments, 4810 conduits and 4621 manholes. For this study the InfoWorks ICM software package (Innovyze, 2017; version 7.0) was used. InfoWorks solves the full St. Venant equations and thus accounts for backwater effects and reverse flow, both of which occur in the Berlin sewer system. Surface runoff is simulated with the Desbordes routing model under consideration of initial losses and evaporation. Domestic and commercial wastewater flow is represented in form of hydrographs.

6.2.3 Tracer calculation

Assuming a highly diverging microbiological contamination potential of the waste and stormwater portions of CSO, a wastewater tracer was simulated as a proxy for fecal pathogens.

The tracer is transported advectively with the flow and is neither subject to sedimentation nor degradation. A concentration of 100 mg L^{-1} in domestic and commercial wastewater and of 0 mg L^{-1} in stormwater runoff was assigned to the tracer as constant values. The wastewater ratio at any point in the combined sewer network can be directly derived from the simulated tracer concentration (Example in Figure 6.1).

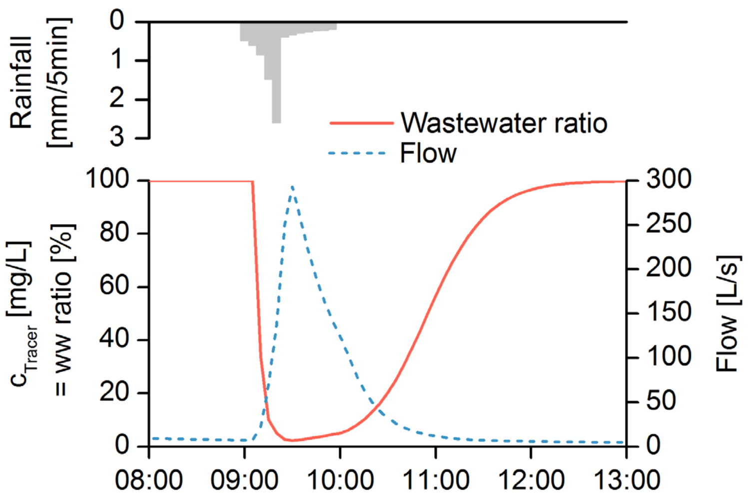

Figure 6.1: Rainfall, flow and wastewater ratio in a combined sewer.

To confirm the relevance of the wastewater portion in CSO for the microbiological contamination of urban rivers, total loads of Eschericia *coli* and *somatic coliphages* originating from the waste and stormwater portion of CSO were calculated. Own measurements in stormwater runoff (Seis et al., 2016) and in raw wastewater (unpublished data) were used.

CSO volumes and wastewater ratios were simulated for the summer period of the year 2016 (May to October). Precipitation data of 9 rain gauges at a 5-min interval was considered with total rainfall ranging between 199 and 261 mm.

6.3 Results and discussion

For the summer period 2016 a total CSO volume of 2.9 million m³ with 144,000 m³ discharged wastewater (5% of the CSO volume) is simulated. Wastewater ratios vary largely between different CSO outlets (Figure 6.2). At 20 outlets, primarily those that receive additional surface runoff of the separate sewer system, only marginal wastewater portions (< 1%) are simulated. On the other hand, 12 outlets have a mean wastewater ratio > 10%, typically in catchments with relatively large populations and a small specific storage capacity. Temporal variability is also very distinctive at some outlets. Highest wastewater ratios are typically observed in the beginning of a rainfall event when sewers are still filled with dry-weather flow. Figure 6.2 shows the frequency distribution of the wastewater ratio at three exemplary CSO outlets with mean wastewater ratios of 0.2% (a), 5.3% (b) and 13.1% (c).

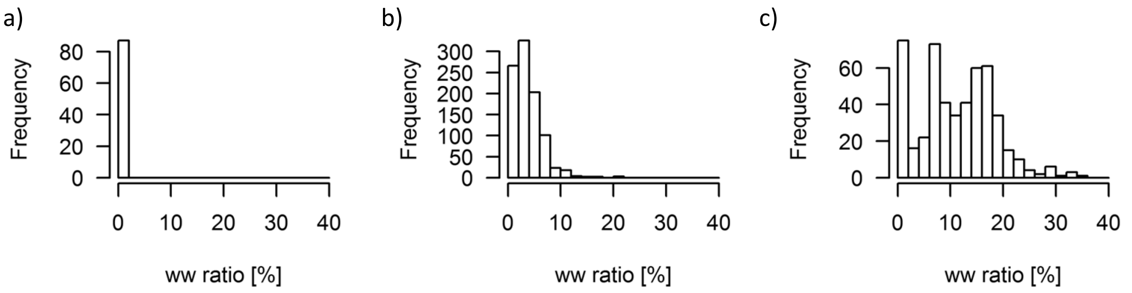

Figure 6.2: Distribution of simulated wastewater ratios at three exemplary CSO outlets located at Landwehrkanal km 6.3 (a), Landwehrkanal km 1.7 (b) and Spree km 6.0 (c).

Following the quantification of wastewater ratios and associated volumes, hotspots for wastewater discharges and potential microbiological contamination were mapped. Results show that the outlets with the largest CSO volumes (Figure 6.3, left) are not necessarily the ones with the highest wastewater volumes (Figure 6.3, right). Vice versa, the two outlets with the highest wastewater contribution (in sum 26%) discharge only 13% of the total CSO volume and would eventully not have been detected with a solely hydraulic simulation approach.

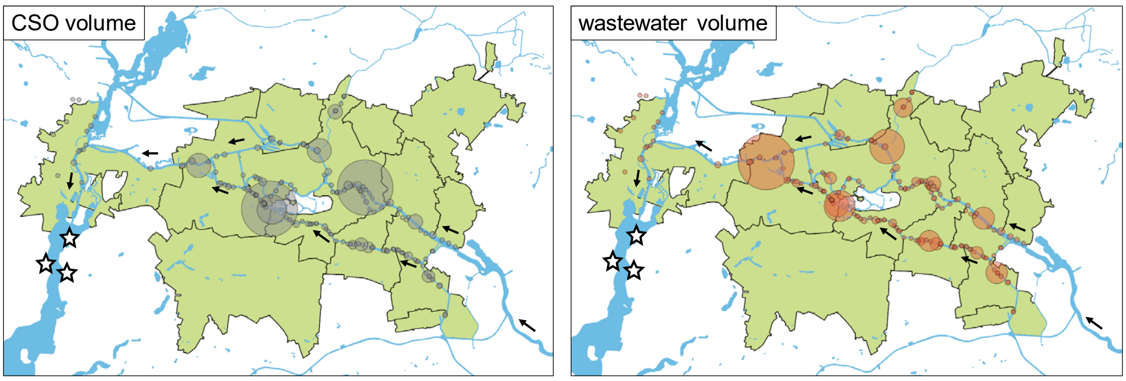

Figure 6.3: Locations of total CSO (left) and wastewater discharges (right) of the Berlin combined sewer system. The size of the circles is proportional to the simulated CSO and wastewater volumes. Arrows indicate the flow direction. The stars represent the downstream bathing sites.

Measurements indicate that pathogen concentrations in wastewater are approx. 4 orders of magnitude higher than in stormwater. As a consequence, wastewater which represents only 5% of the CSO volume contributes > 99% of the total pathogen loadings to the river (Table 6.1). This highlights the importance of simulating wastewater volumes when identifying point sources for the hygienic impairment of water bodies.

Table 6.1: Simulated CSO volume, measured concentrations for E. coli and somatic coliphages and resulting pathogen loadings via the storm and wastewater portions of CSO. Measurements refer to event mean concentrations for stormwater (n = 9) and single samples for wastewater (n = 11).

		E. coli		*S. Coliphages*	
	Simulated volume [m³]	Concentration [MPN/100 mL]	Loading [MPN]	Concentration [PFU/100 mL]	Loading [PFU]
Wastewater	1.4×10^5	1.4×10^8	2.1×10^{17}	2.8×10^5	4.1×10^{14}
Stormwater	2.7×10^6	8.9×10^3	2.4×10^{14}	5.1×10^1	1.4×10^{12}

6.4 Conclusions

The following key conclusions can be drawn:

- A simple wastewater tracer can be used as a proxy for pathogen emissions from a combined sewer system.
- Hotspots for microbiological contamination do not necessarily correspond with the CSO outlets that discharge the largest volumes.
- For the improvement of bathing water quality mitigation measures may be implemented at different locations than measures to tackle other water quality goals, e.g. the elimination of severe oxygen deficits.
- The presented approach can be employed also for other contaminants which are primarily associated with wastewater such as paracetamol or ibuprofen (Weyrauch et al., 2010).

Acknowledgments

The presented paper highlights results of the project FLUSSHYGIENE funded by the German Federal Ministry of Education and Research (BMBF, grant number 02WRS1278A) and the Berlin water utility (BWB).

Chapter 7: Synthesis

7.1 Summary and conclusions

Discharges from urban drainage systems, and combined sewer overflows (CSO) in particular, can have severe adverse effects on aquatic organisms. One of the main stressors for many urban rivers are deficits in dissolved oxygen (DO), mainly due to the degradation of organic matter. Resulting impacts for fish and invertebrates can range from behavioural changes to death. In addition to the ecological impairments, CSO events contribute to poor microbial water quality and hence pose a risk for public health if receiving water bodies are used for bathing. To reduce CSO emissions and mitigate negative impacts, storage tanks, real-time control strategies, and sometimes treatment techniques are implemented in many sewer systems. In the past years, sustainable urban drainage systems (SUDS) that retain and attenuate stormwater runoff, among other benefits, have also been added to the portfolio of CSO mitigation measures. However, the effects of these measures on short-term CSO impacts in the river is scarcely investigated.

In this thesis, different setups of integrated models for the simulation of river impacts from CSO have been used for a large urban area. Extensive monitoring data from the sewer and the river as well as different approaches for model validation were deployed to thoroughly assess model performance. The models were used to provide new insights into the dynamic river processes that lead to oxygen deficits in the river. Additionally, the potential of different global mitigation strategies and realistic SUDS scenarios to reduce CSO emissions and mitigate oxygen deficits in the river was quantified. Global runoff reduction and specific SUDS scenarios were compared to explore the required model complexity and give recommendations for simulating SUDS effects on CSO. Further, the impact of climate change on severe oxygen deficits after CSO and background oxygen conditions during dry weather was highlighted and put into relation with the effect of mitigation measures. Finally, a novel tracer approach for identifying hotspots of pathogen emissions in large combined sewer networks was developed and demonstrated. The approaches can assist in rehabilitating flow-regulated rivers impacted by CSO and support the allocation of measures to improve bathing water quality in complex urban systems. Based on this work, the research questions defined in Chapter 1.4.2 could be answered.

Before the specific outcomes of the thesis are summarized in this chapter, some general conclusions can be drawn:

- The simulation of CSO impacts on receiving rivers is possible in good accordance with measurements, even for complex urban water systems as the Berlin River Spree and for highly interdependent variables as DO.
- Coupled sewer-river models can be used to evaluate the benefits of mitigation strategies and analyse the impact of climate change giving valuable support for strategic planning.
- Sewer-based CSO control measures and SUDS have significant positive effects on CSO emissions and resulting river impacts and can help to protect aquatic organisms.
- A simple wastewater tracer can be used as a proxy for pathogen emissions from a combined sewer system and for identifying the hotspots of microbiological contamination.

In the following, the main outcomes of the thesis are summarised and conclusions are drawn with regards to the research gaps identified in Chapter 1.4.1.

7.1.1 Impact-based modelling for strategic planning

- An integrated modelling approach for the urban drainage system and the receiving river was used to investigate CSO impacts at city scale under complex urban conditions. The model was calibrated and validated with measurements collected during a two-year monitoring campaign at a major CSO outlet and at various river sections. A local sensitivity analysis for the 89 model parameters of the river water quality model was conducted. The effect of changes in boundary conditions (e.g. rainfall, temperature, impervious area, storage capacity) as well as variations at the interface of the sewer and the river were investigated.
- It was shown, that total CSO volumes and pollutant loads can be simulated with relative errors in the range of 5% and 10%, respectively, aggregated over several CSO events. Regarding river water quality, the integrated model can properly simulate annual variations of all major water quality variables as well as the goal variable DO for the specific events of interest. In particular, the whole range of observed DO concentrations is represented by the model. The frequency of critical DO conditions in the river (DO < 2 mg L^{-1} for ≥ 30 min) is simulated with a deviation of one or two calendar days per year, depending on the monitoring station. As a conclusion, the integrated model can represent CSO emissions and impacts in good agreement with observations.
- The integrated sewer-river model reacts sensitively to changes in the catchment (e.g. the reduction of the impervious area), in the sewer (e.g. increase in storage capacity), and in the river (e.g. changes in rates for carbon uptake or hydrolysis), which is an important plausibility check and prerequisite for subsequent scenario analysis. The sensitivity analysis for the river water quality model further revealed the important factors to be included in the calibration process. However, as typical for complex models, some highly correlated parameters were identified (e.g. the maximum growth rate and Chlorophyll-a-

biomass ratio of diatoms), which, in turn, complicates model calibration by giving equally good model results for different parameter sets.

- Different validation approaches for the integrated model were tested and compared. As an important outcome, model validation for CSO impacts should not solely rely on graphical methods or error metrics for the simulated and observed time series, since they can fail in case of time lags or inadequately selected validation time periods. Instead, it is recommended to also compare model output and measurements with regards to legislative goal functions, e.g. the frequency of adverse DO conditions. This is particularly important to verify suitability of models for strategic planning and impact-based decision making.
- Throughout this thesis, the integrated sewer-river model was extended with a detailed rainfall-runoff model including components for a multitude of SUDS to simulate effects of realistic SUDS strategies on the receiving river. The standard and the extended modelling approach were compared regarding simulated CSO volumes for different runoff reductions. It was shown, that a detailed hydrological SUDS model is required to adequately cover the dynamic effects on runoff. The comparison revealed that SUDS, that also attenuate and delay runoff peaks, outperform the global downscaling of the impervious area in terms of CSO reduction. Nevertheless, a SUDS simplification which simply downscales the impervious area can still be acceptable for preliminary planning purposes.
- A simple but powerful upscaling method was developed to overcome the diverging spatial scales between SUDS (building or city quarter scale) and the river (catchment scale). The proposed method extrapolates SUDS effects from an exemplary city quarter to the entire catchment in proportion to the impervious area. By that, the computational costs of extensive rainfall-runoff simulations at the spatial resolution of individual buildings and the high effort of SUDS representation for an entire catchment can be avoided, while preserving the capability to mimic the dynamic runoff behaviour of SUDS.

7.1.2 Process understanding

- The processes affecting the oxygen budget of the river were investigated by two means. First, the results from a local sensitivity analysis were used to distinguish the processes relevant during dry and wet weather conditions. Further, different water quality variables at the interface of the sewer and the river model were varied to identify the major processes that lead to a DO decrease in the river after CSO. Here, concentrations of i) BOD_5 and COD, ii) TSS, iii) DO, and iv) NH_4-N simulated at each CSO outlet were set equal to upstream river conditions one at a time, deactivating the respective effect of CSO on the associated river process. The differences in DO simulated with and without the respective CSO water quality variable were put into relation to the overall DO deficit quantified after neglecting all CSO water quality variables at once. The analysis was done through a Lagrangian approach, starting at a major CSO outlet and following one water parcel downstream.

- Annual DO dynamics during dry weather mainly depend on the two dominant phytoplankton species, diatoms and cyanobacteria, which highlights the importance of trophic background pollution in the Berlin River Spree (Köhler et al., 2002). This is also reflected in long periods with suboptimal DO conditions that imply any kind of impairment and which are almost independent of simulated CSO emissions. In contrast, DO concentrations after CSO are dominated by the degradation of organic matter by heterotrophic bacteria, either directly in the water column or delayed in the sediments (Harremoës, 1982).
- Besides the degradation of organic matter by heterotrophic bacteria, two other processes have been found to contribute to DO deficits after CSO: i) the mixing of river water with CSO spill water poor in DO and ii) the inhibition of the phytoplankton activity due to CSO-induced turbidity. While the first process has already been mentioned sporadically in literature (Harremoës, 1982; Hvitved-Jacobsen, 1982), the second has remained largely unrecognised, to date. The findings provide insights for process formulation in future integrated modelling studies.
- The mixing of overflow water poor in DO leads to an immediate DO sag in the river and is the dominant process during the first ten hours after the CSO peak. It is subsequently exceeded by the degradation of organic matter, which fully evolves after 24 hours of flow time and then makes up about 80% of the DO deficit. The inhibition of phytoplankton activity generally gains importance with time, though never contributing more than 15% of the observed DO deficit. The dependence on time indicates that different processes can be responsible for observed DO depressions at a given river section, depending on flow speed.
- The identified processes are influenced by different types of mitigation measures which, in turn, can determine where along the river positive effects will unfold. As an example, the removal of particles via CSO treatment reacts on the degradation of organic matter as well as on the inhibition of photosynthesis and, in accordance with the process dynamics, has a delayed effect fully evolving after 24 hours of flow time. As a consequence, benefits are expected to be more pronounced in the downstream part of the river. The aeration of overflow water, in contrast, inhibits the immediate DO sag after CSO and therefore can be a viable option for sensitive river sections in close distance to large CSO outlets. Lastly, the increase of storage capacity and runoff reduction via SUDS impact all three identified processes by reducing loadings in organic matter, in suspended solids, and CSO volumes poor in DO. As a consequence, these measures positively affect the DO budget along a comparably long river stretch.

7.1.3 Effects of mitigation measures

- Different strategies for CSO mitigation, that include sewer-based CSO control measures as well as runoff reduction techniques, were tested for the Berlin combined sewer system. Runoff reduction techniques were simulated at different levels of complexity ranging from a global downscaling of the impervious area to individual SUDS elements localised in a specific city quarter. Measure effects were evaluated with regards to the reduction of CSO emissions and the mitigation of impacts on two Berlin lowland rivers, the River Spree and the Landwehrkanal. In addition, the effect of different climate change scenarios was investigated for a selected mitigation strategy.

- It was found that the current implementation of a CSO control program, which includes several new storage tanks and real-time control (RTC) components with in total almost 50% additional storage capacity, leads to a reduction in CSO volumes by 17% for a relatively intense rainfall year. Relative reduction in pollutant loads exceed relative reduction in volume by factors of 1.2 for TSS (21% reduction), 1.4 for BOD (23% reduction), and 1.9 for NH_4-N (31% reduction). This highlights that storage tanks and RTC strategies for in-sewer storage primarily retain the first, typically more polluted portion of CSO. In reverse, the stormwater portion in the remaining overflow volume is slightly increased. As a consequence, the retention of combined sewage can also be a viable option to reduce other quality-related CSO impacts, e.g. ammonia toxicity.

- An important positive effect on CSO volumes was simulated for measures that reduce surface runoff. As an example, a 20% runoff reduction, evenly distributed over the entire combined sewer catchment, would yield a reduction in total CSO volume by 33%. Similar CSO volume to runoff reduction ratios (between 1.3 and 1.9) were found for different runoff reduction and SUDS scenarios investigated for Berlin's biggest combined sewer catchment "Wilmersdorf". Pollutant loads emitted via CSO are reduced at the same rate as CSO volume (independent of the substance), indicating that the mixing ratio of stormwater and wastewater in CSO remains unaffected by runoff reduction measures.

- For individual rainfall events, a linear relationship between CSO and runoff reduction with a slope > 1 was found over a large range of scenarios, which confirms the high efficiency of runoff reduction to reduce CSO emissions. Nonetheless, the reduction efficiency decreases when storage and pumping capacities in the drainage system remain unexploited while marginal CSO spills persist, mainly due to suboptimal sewer management. As a consequence, a particular effort in runoff reduction would be required to eliminate the last 20% of CSO volume for the studied drainage system.

- Positive effects in the river were found to be more pronounced than effects on CSO volumes and pollutant loads across all studied scenarios and strategies. As an example, the 23% reduction in BOD_5 pollutant loads, achieved with an increase in storage capacity, results in a reduction in frequency and duration of critical DO concentrations by about one third, as an average across different sections of the Berlin River Spree. Similar effect ratios were observed for a global reduction of the impervious area and CSO treatment at the three major CSO outlets (assumed treatment capacity per outlet: 2 $m^3 s^{-1}$, removal rate for particulate matter: 50%). In absolute numbers, the frequency of fish-critical DO conditions in the Berlin River Spree can be reduced from about 4 to less than 3 calendar days per year for these scenarios, considering a relatively intense rainfall year.

- River effects of SUDS strategies, investigated for a highly impaired side channel of the Berlin River Spree and a moderate rainfall year, were even more pronounced. No fish-critical oxygen conditions remained after runoff reductions of 28%, simulated for the least extensive of the investigated SUDS strategies. In conclusion, SUDS can mitigate acute CSO impacts to a tolerable level and hence help to protect aquatic organisms and re-establish populations of indicator species. In fact, all three investigated SUDS strategies equally managed to maintain DO above the critical threshold of 2 mg L^{-1}, which implies a relatively large scope of possible actions when it comes to the selection of SUDS. In addition to the positive effects in the river, SUDS produce an important shift in the local water balance with evapotranspiration becoming the dominant hydrological factor.
- In contrast to CSO volumes, no linear relationship was found between runoff reduction and resulting river impacts. Instead, severity and duration of river impacts depend on a number of additional factors, e.g. the discharge of the river which defines its dilution capacity, the water temperature which influences oxygen solubility and degradation kinetics, and lastly river water quality upstream of the CSO outlets, in particular background DO levels. The results highlight the demand for a river water quality representation in integrated modelling, which cannot be substituted by simple first order approximations as it is partly the case for CSO volumes.
- Although frequency and duration of highly critical DO conditions can be significantly reduced with the investigated measures, periods with suboptimal DO conditions that imply any kind of impairment remain largely unaffected. To also moderate suboptimal DO conditions, mostly associated with background pollution effects, additional measures upstream the combined sewer catchment must be taken.
- A full rehabilitation of the Berlin River Spree, however, is challenging since expected climate change puts further pressure on oxygen conditions both during dry weather and after CSO. The projected increase in temperature, as an example, negatively influences most processes relevant for the oxygen budget in the river, such as the oxygen flux into the sediment, oxygen production by algae, or reaeration from the atmosphere. In addition, it leads to a decreased oxygen solubility and an increased oxygen demand of aquatic organisms. As a result, the frequency of suboptimal oxygen conditions in the Berlin River Spree would increase by 67% compared to current climate conditions with 45 instead of 27 affected calendar days per year (average over different river sections). An increase in rainfall intensity by 20% would lead to one to three additional days per

year with highly fish-critical oxygen conditions after CSO, depending on the river section. The effect would counterbalance the improvements achieved through a 50% storage capacity increase.

7.1.4 Identification of pathogen hotspots

- A conservative tracer was implemented in a hydrodynamic urban drainage model to quantify and compare the potential impact of different CSO outlets on bathing water quality. The tracer approach uses the quantity of wastewater in CSO as a proxy for pathogen emissions, given the large differences of pathogen concentrations in wastewater and stormwater. With a constant tracer concentration of 100 mg L^{-1} in domestic and commercial wastewater and of 0 mg L^{-1} in stormwater, the wastewater volume emitted via CSO and so the microbial pollution potential can be directly calculated from the simulated CSO volume and the associated tracer concentration. The approach was tested for the entire city of Berlin with 176 individual CSO outlets, discharging into different rivers and side channels, to determine the hotspots of microbiological contamination.
- A wastewater ratio in total CSO volume of 5% was simulated for the entire Berlin combined sewer system. Since concentrations of *E. coli* in wastewater were found to be four orders of magnitude higher than in stormwater (about 10^8 vs. 10^4 MPN 100 mL^{-1}), more than 99% of the *E. coli* load in CSO can be attributed to wastewater, despite its comparably small volume.
- The wastewater ratio and thus the *E. coli* concentration in CSO are subject to high spatial and temporal variation. Mean wastewater ratios for the 176 CSO outlets that discharge into Berlin River Spree, its side channels, the River Panke, and the River Havel over a total flow distance of 50 km, range between 0 and 22%. The ratio is particularly low at outlets that also receive stormwater runoff from a separate sewer system, as it is partly the case in Berlin. In contrast, the wastewater ratio in CSO is particularly high in densely populated areas with a comparably small storage capacity. In terms of temporal variation, the microbial load is usually highest at the beginning of a CSO event, when the dry weather flow is pushed away by the inflowing stormwater.
- A large part of the CSO emissions is discharged to the receiving waters via only a few hotspots. For both CSO volume and the wastewater fraction in CSO about half of the total volume can be attributed to only five different outlets. Nonetheless, hotspots for microbiological contamination do not necessarily correspond with CSO volume hotspots. While the largest CSO volumes are emitted to the Landwehrkanal and the upstream section of the Berlin River Spree, the largest wastewater volumes are discharged into the downstream section of the Berlin River Spree and into the River Panke. As a result, mitigation measures for an improved bathing water quality must not necessarily target the same areas as measures to mitigate fish-critical oxygen conditions, given that a

relevant part of DO consuming substances can also originate from the stormwater fraction of CSO (Wicke et al., 2015).

- The findings show the importance of distinguishing different fractions in CSO when looking for the CSO hotspots for microbial contamination of bathing waters. Even though the developed tracer approach neglects the dynamic effects of dry weather duration and rainfall intensity on pathogen loadings (McCarthy et al., 2012), it can still give an approximation of the contamination potential of different CSO outlets in a large sewer network. As a result, it can pinpoint suitable areas for the allocation of mitigation measures. The presented approach can also be employed for other contaminants which are primarily associated with wastewater such as paracetamol or ibuprofen (Weyrauch et al., 2010).

7.2 Recommendations for stakeholders

In the following, recommendations for stakeholders, primarily water authorities and utilities, are given and transferability of methods and conclusions to other cities is discussed.

Effective CSO control measures are required to mitigate acute CSO impacts to a tolerable level and will be indispensable to counteract the effects of global climate change or ongoing urbanisation. The results of this thesis highlight that any action that reduces stormwater runoff from the catchment or that retains or treats combined sewage has a positive effect on the river. This includes the mitigation of oxygen deficits and most likely also other negative impacts from CSO, e.g. ammonia toxicity. A particular improvement of ambient water quality can be expected from SUDS, which also have other positive side effects, such as an increase in biodiversity and the mitigation of urban heat islands. When it comes to the selection of SUDS, there is a large scope of possible combinations that can help to achieve a certain water quality goal. Particularly high positive effects on CSO emissions can be expected from SUDS that also attenuate and delay runoff peaks, e.g. green roofs or through-trench-infiltration.

If decision makers aim at avoiding all CSO events and resulting river impacts, a complete shift in the urban water balance must be made. At best, runoff reduction measures should be combined with an optimised management of the drainage system. To also moderate suboptimal oxygen conditions during dry weather, CSO control measures must be combined with additional measures upstream of the combined sewer catchment that target background pollution effects. This will become particularly relevant given the expected increase in temperature associated with global climate change, which will negatively affect the oxygen budget of the river. To finally re-establish populations of indicator species, improvements in river morphology are needed, in addition. Lastly, it is pointed out that measures to reduce pathogen emissions and improve bathing water quality might need to focus on different parts of the drainage system than measures to improve oxygen conditions in the river. To find the hotspots of pathogen emissions in large and complex combined sewer systems, the simulation of a wastewater tracer is recommended.

Regarding required model complexity for strategic planning, global runoff reduction scenarios without an explicit representation of SUDS may be sufficient to get a first estimate of SUDS' potential to mitigate CSO impacts. In fact, for the exclusive estimation of CSO volumes, even a statistical approach based on the linearity found between runoff and CSO reduction for a large range of situations could be used. However, it must be considered that, depending on rainfall dynamics, effects on CSO volume will be underestimated when the specific hydrological effects of SUDS are not accounted for. In case of a detailed SUDS representation, in turn, obstacles caused by diverging spatial scales between SUDS (building or city quarter scale) and the river (catchment scale) can be overcome with the upscaling method proposed in this thesis. In any case, a detailed river water quality model is required to adequately represent CSO impacts in the river.

The developed methods, general conclusions, and also part of the results, e.g. the pronounced relative effect of runoff reduction on CSO emissions, can be transferred to other cities with similar river types and climatic conditions. Nonetheless, the developed model tools would need to be adapted to the local situation. Regarding the identified CSO-related processes, the mixing of overflow water poor in dissolved oxygen is expected to be particularly important for cities with a high degree of imperviousness and a flat sewer system that favours anoxic conditions. The inhibition of phytoplankton activity, in turn, is expected to be relevant in water bodies in which photosynthesis dominates the aquatic oxygen budget.

7.3 Limitations and outlook

Although these findings fill some important knowledge gaps, few questions remain unanswered. In the following, the main limitations of this thesis and ideas for future research are outlined.

First, uncertainties in model predictions, which are an inherent part of any mathematical model and of integrated models in particular (Moreno-Rodenas et al., 2019), could not be thoroughly investigated within this thesis. In the following, important sources of uncertainties which propagate from the upstream to the downstream model are summarised:

- *Input and calibration data uncertainties:* A major source of uncertainty for this study is the spatial variation of rainfall, the main input of the urban drainage model, which is usually not sufficiently covered by point rainfall measurements (Moreno-Rodenas et al., 2017). In addition, uncertain assumptions on the DO concentration in CSO discharges had to be made, as this variable is not explicitly simulated by the urban drainage model. Instead, it was treated as a constant, ignoring dependencies on the residence time in the sewer and the wastewater ratio in CSO as explored by Huisman et al. (2004). Lastly, measurement data used for calibration or as model input can be subject to errors, e.g.

related to sampling, storage, and analytical methods (Dotto et al., 2014) and so contribute to model uncertainties.

- *Model parameter uncertainties:* All model parameters - whether quantified in physical experiments or adjusted during model calibration - are subject to uncertainties (Dotto et al., 2012). As the river water quality model QSim counts almost 100 configurable model parameters, many of them relevant for the DO budget, large overall uncertainties must be assumed. As typical for complex models, some of the parameters are non-identifiable, e.g. parameters that describe phytoplankton growth or the degradation of organic matter. These identifiability issues reveal unnecessary model complexity and can complicate model calibration processes by giving equally good model results for different parameter sets (Freni et al., 2009).
- *Model structure uncertainties:* Structural model errors can consist in an inaccurate representation of the physical system to be modelled, ill-posed model equations, or a missing or inadequate representation of processes (Deletic et al., 2012). Model structure uncertainties are very hard to evaluate but are inherent in any deterministic model. As an example, degradation processes in the sewer network were not considered in this study and are probably compensated to some extent by the river model. Further, build-up and wash-off processes on the catchment's surface, which can lead to large temporal variations of pollutant concentrations (Wicke et al., 2021) were not considered for the investigation of SUDS effects.

A first important step for future research activities on model uncertainties would be the quantification of the importance of the different uncertainty sources, e.g. with methods proposed by Beven and Binley (1992), Vrugt et al. (2003), and Dotto et al. (2014). In a second step, uncertainties associated with the limited spatial resolution of rainfall input data could be attempted to be reduced with the integration of radar sensor data or the use of microwave networks as suggested by Pastorek et al. (2019). In addition, further investigations on DO in CSO spill water for the local boundary conditions are recommended. A large potential for a more accurate representation of pollutant loads in the urban drainage model is also seen in the establishment of data-driven methods which, besides rainfall, take into account land use, road traffic, tree vegetation, or cleaning of streets, among others. Lastly, special effort should be put on the reduction of model complexity, especially in the river water quality model, which will make future calibration processes easier and avoid simulation errors due to non-identifiable model parameters.

As another limitation of this study, all investigations have been done for selected rainfall and river boundary conditions of individual years. For example, a relatively intense rainfall year was chosen for the investigation of global mitigation strategies and river processes, an average rainfall year was chosen to investigate specific SUDS effects on the river, and a relatively dry rainfall year was selected for determining hotspots of pathogen emissions. Consequently, no statistically proven statement on the variability of measure effects under different rainfall and river conditions as well as long-term assessment of measure performance could be made. This limitation is mainly associated with the high computational costs of integrated modelling studies in large urban areas. To overcome this limitation, it is suggested to focus future studies on events with severe CSO impacts in the river and use the corresponding rainfall and river boundary data, e.g. of the past ten years, to build event-based time series for sewer and river model input. This approach would avoid the costly simulation of time periods without special relevance for the river and facilitate an assessment of the long-term performance of mitigation measures from a river perspective. As an alternative, high performance computers could also be deployed for long-term simulations. Lastly, measure effects on CSO emissions could also be estimated to some extent with a simple statistical emulation model, based on the linearity found between runoff and CSO volume for a range of rainfall events. Nonetheless, a detailed river water quality model or at least a more complex data-driven approach would be required to cover the non-linear nature of river impacts.

Finally, SUDS in this thesis have been designed and parameterised to reproduce the median runoff and pollutant reduction found in literature. In this context, the dependency of the SUDS' performance on system age and its specific implementation has not been accounted for. Further, the capability of SUDS to cope with extreme storm events or droughts has not been thoroughly examined. To overcome these limitations, it is recommended to dedicate future work to the alteration of SUDS' performance over their operational lifetime and further consider risks of system failure or malfunction. In addition, differences between individual designs of specific SUDS should be accounted for. Such investigations on the robustness of measures would significantly improve the knowledge basis of decision makers and help in strategic planning of SUDS. Further, it is recommended to adapt or extend the developed upscaling method for SUDS simulation to the entire city of Berlin and optimise strategies with regards to the location of SUDS within the city. This could be done by defining river-based goal functions that take into account the achieved oxygen levels or the river length at which noticeable improvements are observed.

Chapter 8: Supplementary contributions

The chapter gives an overview of supplementary scientific work related to the topic of this thesis and published in non-peer-reviewed journals, books or as conference proceedings.

Publications in non-peer-reviewed journals and book chapters (in chronological order):

Riechel, M., Matzinger, A., Uldack, M., Caradot, N., Sonnenberg, H., Rouault, P., Pawlowsky-Reusing, E., v. Seggern, D., Heinzmann, B. (2012). Immissionsorientierte Mischwasserbewirtschaftung. *wwt - Wasserwirtschaft Wassertechnik, Modernisierungsreport* 2012/2013, 65-67.

Matzinger, A., **Riechel, M.**, Petersen, S., Heinzmann, B. & Pawlowsky-Reusing, E. (2015). A planning instrument for an integrated and recipient/impact based CSO control under conditions of climate change. Book chapter in: *Climate Change, Water Supply and Sanitation: Risk assessment, management, mitigation and reduction*, 312-317. IWA Publishing. https://doi.org/10.2166/9781780405001

Conference papers (in chronological order):

Riechel, M., Matzinger, A., Rouault, P., Schroeder, K., Sonnenberg, H., Pawlowsky-Reusing, E., Leszinski, M., 2010. Application of stormwater impact assessment guidelines for urban lowland rivers - the challenge of distinction between background pollution and impacts of combined sewer overflows. pp. 8. *7th International Conference on Sustainable Techniques and Strategies in Urban Water Management - Novatech*, Lyon, France.

Riechel, M., Matzinger, A., Meier, I., Caradot, N., Stapf, M., Sonnenberg, H., Pawlowsky-Reusing, E., Heinzmann, B., Rouault., P. (2011). Towards an Impact-based Planning Instrument for Combined Sewer Management in Berlin, Germany, pp. 2. *2nd International Conference on Integrated Water Resource Management*. Dresden, Germany.

Riechel, M., Schwarzmüller, H., Pallasch, M., Sieker, H., Säumel, I., Taute, T., Köhler, M., Kaiser, D., Heise, S., Bartel, H., Heinzmann, B., Joswig, K., Rouault, P., Matzinger, A. (2014). Bewertung von Maßnahmen der Regenwasserbewirtschaftung am Beispiel von Umwelteffekten, pp. 3. *Aqua Urbanica* 2014. Innsbruck, Österreich.

Riechel, M., Stapf, M., Philippon, V., Hürter, H., Pawlowsky-Reusing, E., Rouault, P. (2015). A Holistic Assessment Approach to Quantify the Effects of Adaptation Measures on CSO and Flooding. pp. 4. *10th International Urban Drainage Modelling Conference*. Quebec, Canada.

Philippon, V., **Riechel, M.**, Stapf, M., Sonnenberg, H., Schütze, M., Pawlowsky-Reusing, E., Rouault, P. (2015). How to find suitable locations for in-sewer storage? - Test on a combined sewer catchment in Berlin. pp. 4. *10th International Urban Drainage Modelling Conference*. Quebec, Canada.

Riechel, M., Pallasch, M., Matzinger, M., Sommer, H., Heinzmann, B., Joswig, K., Pawlowsky-Reusing, E., Rouault, P. (2016). A modelling approach for assessing acute river impacts of realistic stormwater management strategies. pp. 4. *8th International Conference on Sewer Processes and Networks*. Rotterdam, The Netherlands.

Matzinger, A., **Riechel, M.**, Schmidt, M., Corral, C., Hein, A., Offermann, M., Strehl, C., Nickel, D., Sieker, H., Pallasch, M., Köhler, M., Kaiser, D., Möller, C., Büter, B., Lessmann, D., Günther, R., Säumel, I., Pille, L., Winkler, A., Heinzmann, B., Joswig, K., Reichmann, B., Sonnenberg, H., Remy, C., Schwarzmüller, H. and Rouault, P. (2016). Quantification of multiple benefits and cost of stormwater management. pp. 4. *9th International Conference on Sustainable Techniques and Strategies in Urban Water Management - Novatech*, Lyon, France.

Riechel, M., Matzinger, M., Pallasch, M., Heinzmann, B., Joswig, K., Rouault, P. (2017). Gewässerschutz durch kombinierte dezentrale und zentrale Maßnahmen der Regenwasserbewirtschaftung - Modellstudie am Beispiel Berlins. pp. 13. *Aqua Urbanica* 2017. Graz, Österreich.

Appendix: Supplementary material for Chapter 5

This is supplementary material for the following publication:

Riechel, M., Matzinger, A., Pallasch, M., Joswig, K., Pawlowsky-Reusing, E., Hinkelmann, R., Rouault, P. (2020). Sustainable urban drainage systems in established city developments: Modelling the potential for CSO reduction and river impact mitigation. *Journal of Environmental Management* 274, 111207. https://doi.org/10.1016/j.jenvman.2020.111207

The publication itself can be found in Chapter 5 of this thesis ("Sustainable urban drainage systems for CSO impact mitigation"). The supplementary material can also be found online at https://doi.org/10.1016/j.jenvman.2020.111207.

A. SUDS strategies

Table A.1: SUDS and deployed combinations for strategy A

No. ranked	Measure (combination)	Connected area [ha]	Portion of city quarter's impervious area
1	Tree-trenches	8.0	10.9%
2	Extensive green roofs + green facades	6.7	9.1%
3	Green facades	2.8	3.9%
4	Stormwater harvesting	2.8	3.8%
5	Extensive green roofs	1.1	1.5%
6	Stormwater harvesting + green facades	0.3	0.4%

Table A.2: SUDS and deployed combinations for strategy B

No. ranked	Measure (combination)	Connected area [ha]	Portion of city quarter's impervious area
1	Stormwater harvesting	7.5	10.2%
2	Extensive green roofs + stormwater harvesting	4.7	6.5%
3	Extensive green roofs + tree-trenches	4.1	5.6%
4	Permeable pavements	3.8	5.2%
5	Decentralised treatment + stormwater harvesting	2.7	3.8%
6	Tree-trenches	2.7	3.7%
7	Permeable pavements + Trough-trench infiltration	1.9	2.6%
8	Stormwater harvesting	1.3	1.7%
9	Green facades + stormwater harvesting	1.1	1.5%
10	Extensive green roofs + artificial pond	0.9	1.2%
11	Green facades	0.7	1.0%
12	Artificial pond + infiltration swales	0.6	0.9%
13	Others	0.7	1.0%

Table A.3: SUDS and deployed combinations for strategy C

No. ranked	Measure (combination)	Connected area [ha]	Portion of city quarter's impervious area
1	Green facades	9.0	12.3%
2	Extensive green roofs	7.6	10.4%
3	Stormwater harvesting	6.2	8.5%
4	Tree-trenches	4.0	5.4%
5	Artificial stream + artificial pond	2.9	4.0%
6	Green facades + infiltration swale	1.9	2.6%
7	Others	0.4	0.5%

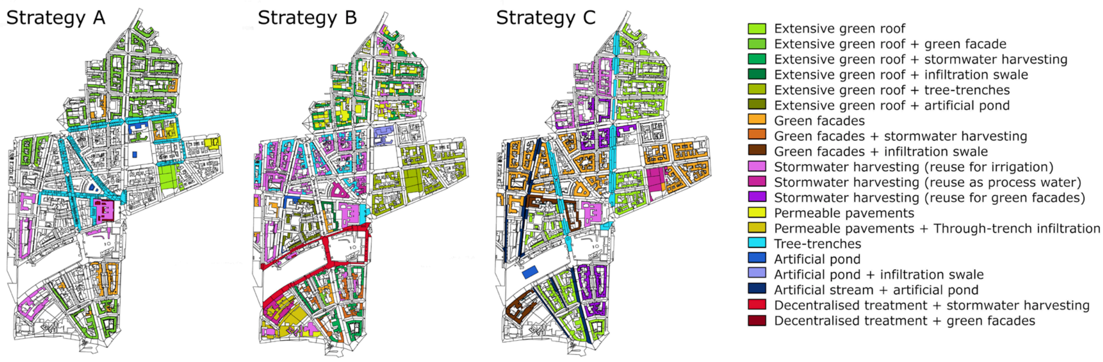

Figure A.1: SUDS strategies A, B and C, adapted from Matzinger et al. (2017a). Colours indicate the type of SUDS the respective surface elements are connected to.

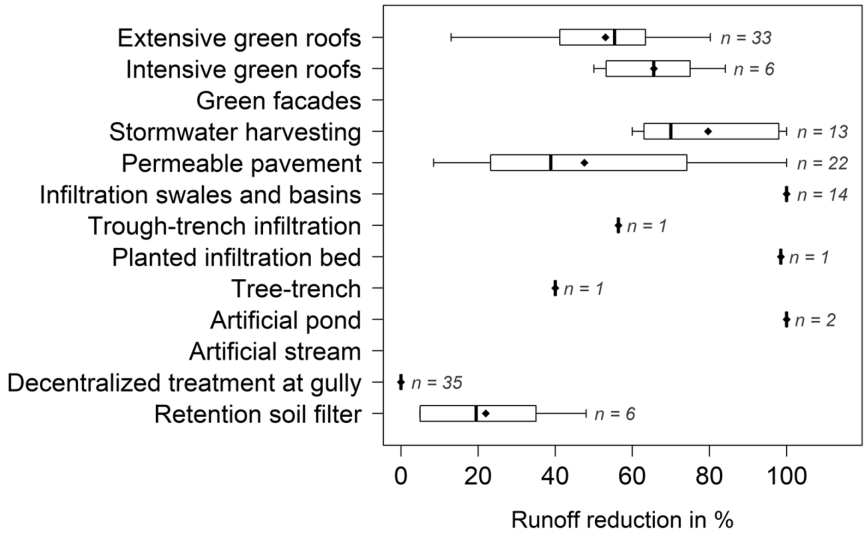

Figure A.2: Runoff reduction potential of SUDS derived from literature data and own measurements (in case of green roofs and retention soil filters, unpublished data).

B. Model setup and upscaling of runoff

Table A.4: Mean pollutant concentrations in stormwater runoff derived from literature and adopted in the rainfall-runoff model. References a to f are given below.

Surface type	TSS [mg L^{-1}]	COD [mg L^{-1}]	BOD_5 [mg L^{-1}]	NH4-N [mg L^{-1}]	TN [mg L^{-1}]	TP [mg L^{-1}]
Roof	40 [a]	40 [d]	12 [a]	0.1 [e]	2 [f]	0.03 [e]
Street	200 [b]	120 [d]	45 [c]	0.6 [c]	3 [f]	0.80 [c]
Yard	86 [c]	70 [a]	45 [c]	0.1 [a]	3 [f]	0.20 [a]
Green structure	12 [a]	19 [a]	2 [a]	0.8 [a]	3 [a]	0.10 [a]

References: [a] Göbel et al. (2007), [b] Schmitt et al. (2011), [c] Wicke et al. (2015), [d] Sommer (2007), [e] Schubert et al. (2015), [f] Scheid et al. (2013)

Upscaling of runoff: To account for the diverging scales of the rainfall-runoff model (city quarter scale, 73 ha impervious area) and the sewer model (catchment scale, 921 ha impervious area), runoff was scaled up at the interface of the two models. This was realised with a three-step procedure:

1. Spatial aggregation of runoff simulated for the 2,463 surface elements of the city quarter to one single time series,
2. Calculation of the portion of each of the 257 subcatchments of the sewer catchment in relation to the impervious area of the city quarter,
3. Multiplication of the aggregated runoff time series (step 1) with each of the calculated scaling factors (step 2) and assignment of the newly generated runoff time series to the corresponding subcatchments.

A graphical illustration of the upscaling method is given in Figure A.3.

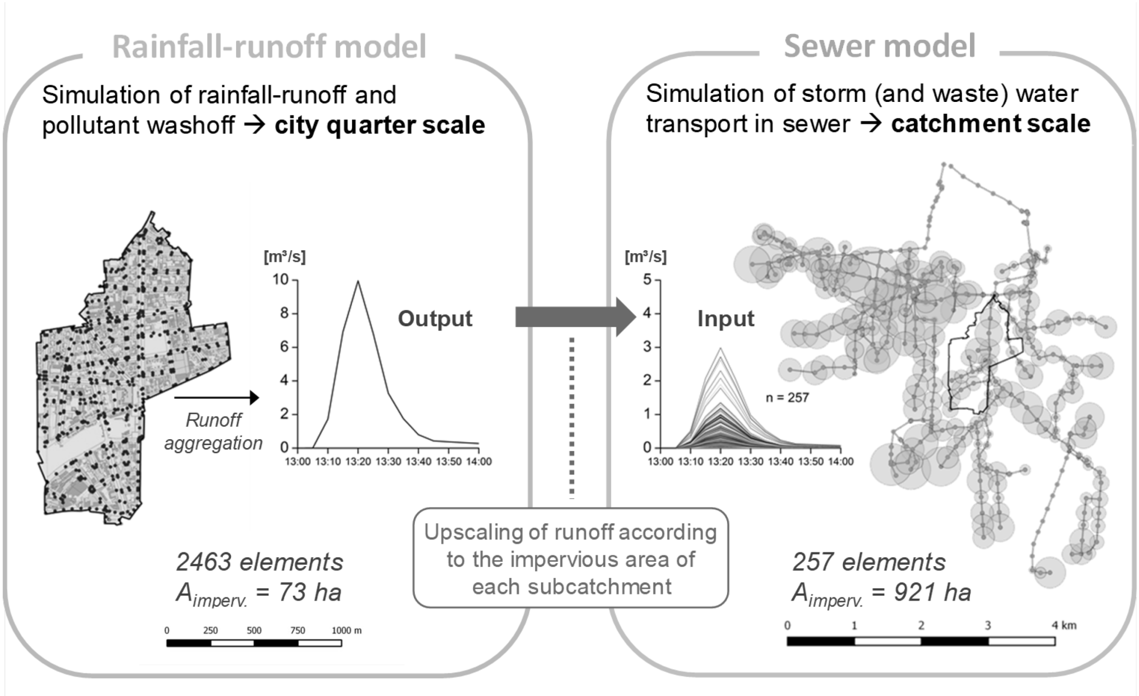

Figure A.3: Upscaling from the rainfall-runoff model (city quarter scale) to the sewer model (catchment scale)

C. Model validation

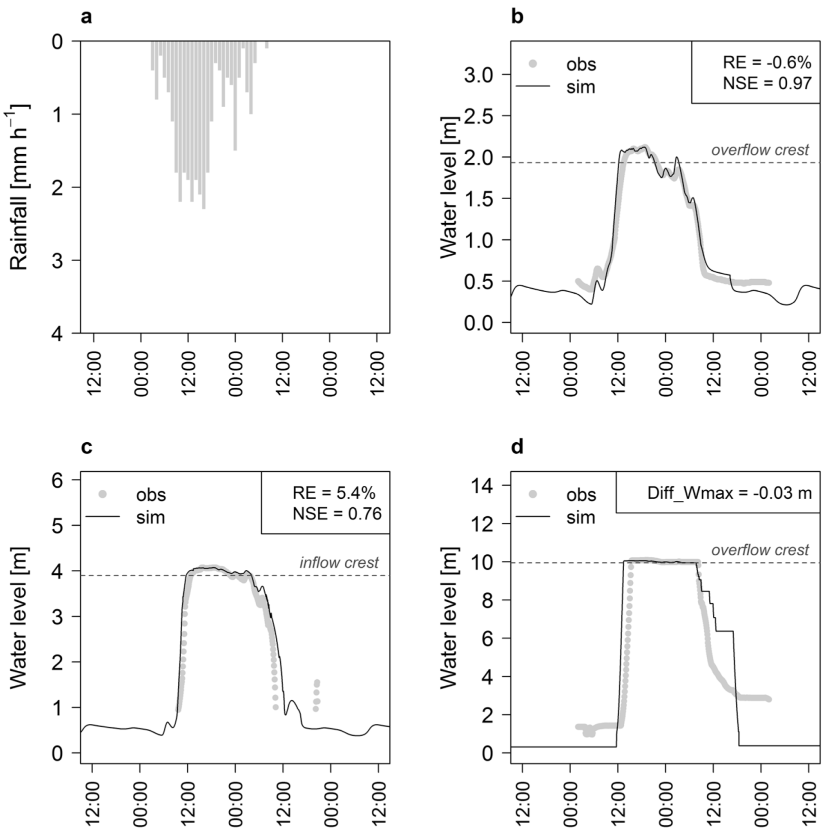

Figure A.4: Comparison of observed ("obs") and simulated ("sim") water levels for an exemplary rainfall event of 38 mm in 31 h. Plot panels show rainfall intensity (a) and water levels at a major overflow structure (b), at the inflow of a major stormwater tank (c) and in the tank itself (d). RE = relative error, NSE = Nash-Sutcliffe efficiency (Nash and Sutcliffe, 1970), Diff_Wmax = difference in maximum water level.

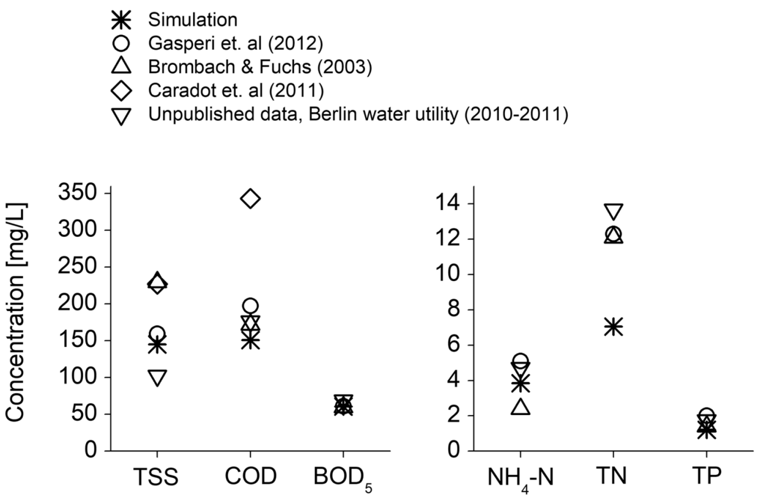

Figure A.5: Comparison of mean simulated pollutant concentrations in CSO with literature data (Brombach and Fuchs, 2003; Caradot et al., 2011; Gasperi et al., 2012) and unpublished data by the Berlin water utility.

D. Rainfall scenario

The developed stormwater strategies and runoff reduction scenarios were analysed for a one-year representative rainfall series for the local climate conditions. To find this representative rainfall year, six rainfall metrics (e.g. total rainfall and frequency of intense rain events, see Table A.5) were calculated for each year of a continuous 30-year rainfall series (rain gauge Berlin-Dahlem, 5-minute time step, time period: 1981-2010) and then compared to the long-term average. The year 1990 has the smallest mean relative deviation from the long-term averages and hence was selected as representative model input. A long-term simulation for e.g. 30 years was not feasible due to the enormous computational effort, especially regarding river impact modelling.

Table A.5: Rainfall characteristics of the climatic reference period 1981-2010 and the selected year 1990 with its relative deviations from the long-term averages (T = return period).

Rainfall metric	Annual average (1981-2010)	Year 1990
Total rainfall	582 mm	612 mm (+5%)
Summer rainfall (May - Oct.)	328 mm	342 mm (+4%)
Winter rainfall (Nov. - Apr.)	253 mm	270 mm (+7%)
Freq. of rainfall ≥ 7.5 mm h^{-1} (T = 3 months)	3.9	4 (+3%)
Freq. of rainfall ≥ 29.2 mm h^{-1} (T = 5 years)	0.2	0 (-)
No. of dry weather periods with duration ≥ 5 d	20	21 (+5%)

E. Results for SUDS strategies and global runoff reduction scenarios

Table A.6: Surface runoff, CSO emissions and river impacts for status quo simulation and stormwater strategies A to C. Relative effects of stormwater strategies compared to status quo are given in parentheses.

	Status quo	Strategy A	Strategy B	Strategy C
Total runoff [m^3 ha^{-1}]	2,949	2,136 (-28%)	1,823 (-38%)	1,790 (-39%)
Peak runoff rate [L s^{-1} ha^{-1}]	96.2	66.1 (-31%)	59.6 (-38%)	50.3 (-48%)
Overflow volume [m^3]	690,716	381,054 (-45%)	296,903 (-57%)	292,065 (-58%)
BOD_5 pollutant loads [t]	42.0	25.5 (-39%)	19.9 (-53%)	19.8 (-53%)
Overflow frequency [1] [-]	19	10 (-47%)	7 (-63%)	7 (-63%)
Frequency of DO_{crit} [-]	5	0 (-100%)	0 (-100%)	0 (-100%)

[1] CSO events with volumes < 100 m^3 not counted (only one occurrence for strategy A)

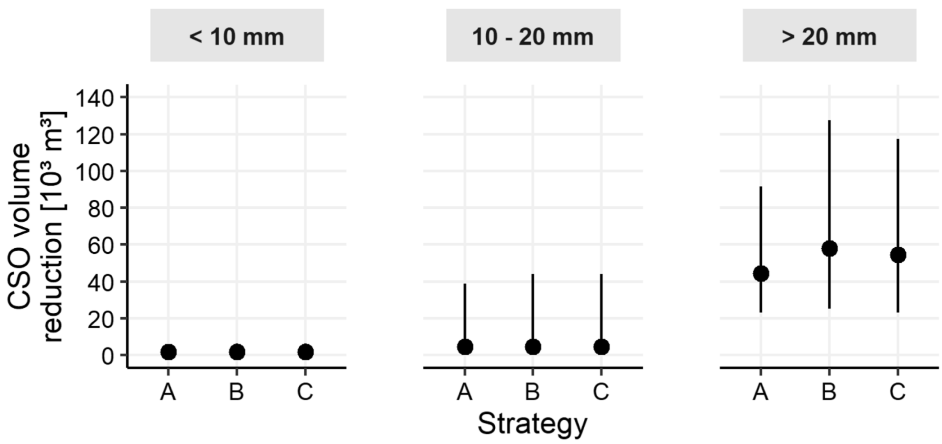

Figure A.6: Reduction in CSO volume for the three SUDS strategies and three classes of rain events: rainfall depth < 10 mm (n = 7, left), rainfall depth: 10 – 20 mm (n = 7, centre) and rainfall depth > 20 mm (n = 5, right). Points show median, whiskers show minimum and maximum values.

Table A.7: Rainfall characteristics for the 19 CSO events of the studied rainfall year

Event	Start	End	Duration [h]	Total rainfall [mm]	Max. rainfall intensity [mm 5 min^{-1}]	Max. rainfall intensity [mm h^{-1}]
1	1990-01-25 13:00	1990-01-25 21:40	8.8	11.0	1.1	2.9
2	1990-02-14 01:35	1990-02-14 07:15	5.8	8.2	0.4	3.5
3	1990-02-28 21:20	1990-03-01 05:40	8.4	11.6	0.5	3.2
4	1990-03-06 17:25	1990-03-07 09:25	16.1	14.3	0.4	3.0
5	1990-05-09 15:30	1990-05-09 19:40	4.2	6.2	1.7	6.0
6	1990-05-12 13:05	1990-05-13 03:25	14.4	18.4	7.9	17.5
7	1990-06-08 01:50	1990-06-09 06:50	24.3	30.2	2.7	5.8
8	1990-06-09 11:00	1990-06-10 17:05	30.2	69.3	1.1	7.2
9	1990-06-18 18:50	1990-06-19 02:55	8.2	10.3	1.2	5.5
10	1990-06-20 20:50	1990-06-21 08:15	11.5	16.8	1.8	8.7
11	1990-06-30 00:45	1990-06-30 03:00	2.3	7.9	1.2	6.8
12	1990-08-09 18:10	1990-08-10 14:35	20.5	14.0	1.6	6.9
13	1990-08-14 19:00	1990-08-14 19:55	1.0	7.3	2.2	7.3
14	1990-08-20 06:10	1990-08-20 11:45	5.7	9.6	1.3	6.0
15	1990-08-30 23:40	1990-08-31 00:30	0.9	9.7	4.3	9.7
16	1990-08-31 21:55	1990-09-01 02:40	4.8	25.0	7.0	13.3
17	1990-09-07 15:00	1990-09-08 07:00	16.1	8.4	2.2	6.8
18	1990-11-17 05:15	1990-11-19 21:50	64.7	29.1	0.7	3.2
19	1990-12-10 10:25	1990-12-11 20:40	34.3	33.7	1.0	2.8

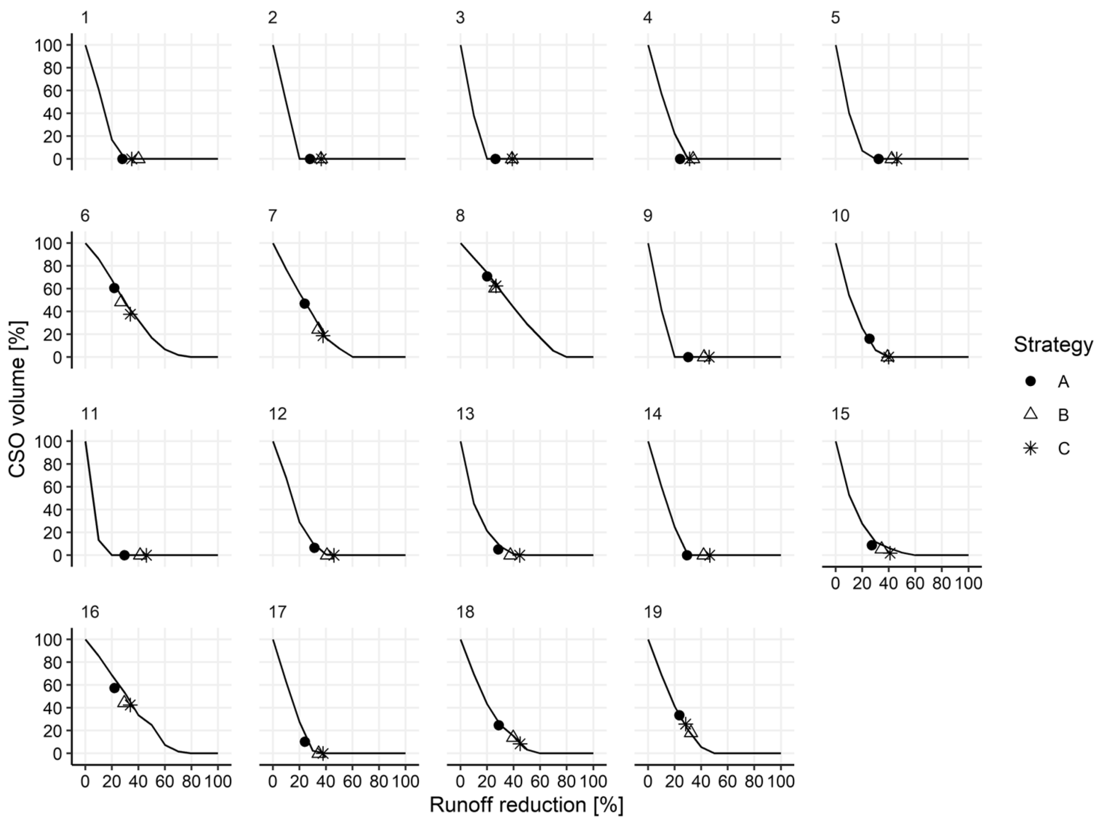

Figure A.7: Effect of runoff reduction on CSO volume for global runoff reduction scenarios (black lines) and realistic strategies (point symbols, not considering sewer-based measures) for the 19 CSO events of the studied rainfall year 1990.

F. Effect of SUDS strategies on the urban water balance

The overall reduction in surface runoff achieved with the three SUDS strategies (between 28% and 39%) implicates an important relative increase in annual infiltration (54% to 83%) and, to a lesser extent, in evapotranspiration (13% to 17%, Table A.8). The annual shares of infiltration in the total water balance (Figure A.8) increase by 6.7%, 8.9% and 10.1% and those of evapotranspiration by 5.9%, 8.5% and 8.1% for strategies A, B and C, respectively. For all three strategies, evapotranspiration supersedes surface runoff as the dominant hydrological factor which points to an important shift in the local water balance (Figure A.8). Nonetheless, the simulated evapotranspiration rates (46% to 49% of annual rainfall) still differ largely from those expected under natural conditions. Glugla et al. (1999) reported evapotranspiration rates of 75% for grasslands and 84% for forests under local climate conditions. Differences between the three strategies are relatively small and mostly originate from the different total areas connected to SUDS.

Table A.8: Urban water balances for the status quo simulation and the SUDS strategies A to C for the investigated average rainfall year 1990 (total rainfall: 612 mm). Relative effects of SUDS strategies compared to the status quo are given in parentheses. Storage refers to the volume remaining in cisterns, green roofs, infiltration swales, etc. at the end of the simulation.

	Status quo	Strategy A	Strategy B	Strategy C
Surface Runoff [mm]	294	213 (-28%)	182 (-38%)	178 (-39%)
Infiltration [mm]	71	110 (+54%)	122 (+72%)	130 (+83%)
Evapotranspiration [mm]	247	278 (+13%)	290 (+17%)	290 (+17%)
Storage [mm]	-	11 (-)	18 (-)	14 (-)

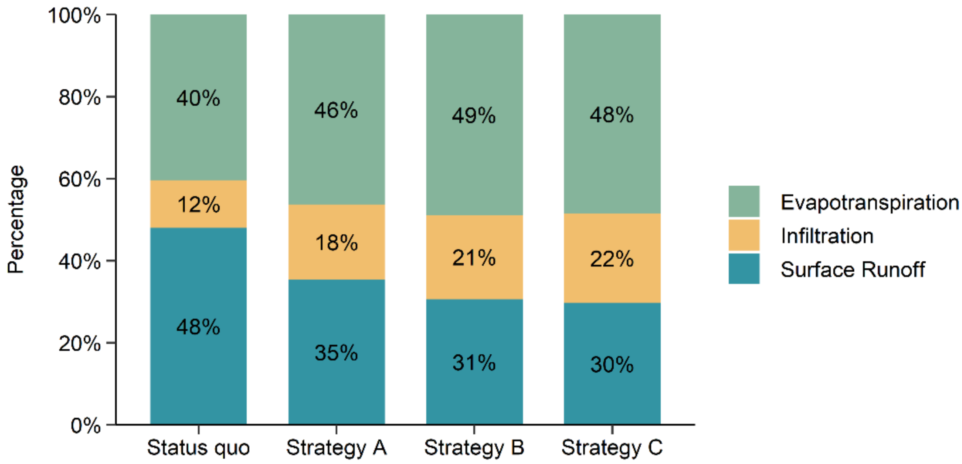

Figure A.8: Shares in the urban water balance simulated for the status quo and the three SUDS strategies A, B and C for an average rainfall year. Volumes stored in SUDS at the end of the simulation are not considered here.

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