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ASSESSING THE IMPACT OF URBAN DRAINAGE MEASURES WITH REGARD TO THE WATER FRAMEWORK DIRECTIVE

DISSERTATION

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KURZFASSUNG

Die Bewertung von Maßnahmen in der Siedlungsentwässerung unter Berücksichtigung der europäischen Wasserrahmenrichlinie (WRRL) ist deutlich komplexer als die traditionellen Methoden. In der WRRL ist der kombinierte Ansatz festgelegt, das heißt, dass sowohl Emissions- als auch Immissionsgrenzwerte eingehalten werden müssen. Die Immissionsgrenzwerte sind jedoch noch nicht definiert, außer der Festlegung der WRRL, dass alle Oberflächenwasserkörper zumindest einen guten ökologischen Zustand erreichen müssen.

Das Ziel dieser Dissertation war die Bewertung der Auswirkung verschiedener Siedlungsentwässerungsmaßnahmen auf die Umwelt, besonders auf Fließgewässer. Zwei Messkampagnen wurden durchgeführt, eine zur Messung der verkehrsbedingten Verschmutzung von urbanem Schnee, die andere um die Auswirkung von Parkplatzabflüssen auf den Boden von Infiltrationsmulden zu bestimmen. Für die Bewertung von Auswirkungen der Siedlungsentwässerung auf Fließgewässer ist die WRRL heutzutage von grundlegender Bedeutung in Europa. Deshalb war ein Ziel dieser Dissertation den guten ökologischen Zustand anhand von Wasserqualitätsparametern zu beschreiben. Dieses ist jedoch heute aus verschiedensten Gründen noch nicht möglich (z. B. wegen der noch nicht identifizierten Ursache-Wirkungsbeziehungen zwischen biologischen Parametern und abiotischen Gewässerparametern). Deshalb wurde stattdessen die Identifikation geeigneter Immissionsgrenzwerte angestrebt, welche als Ersatz für den guten ökologischen Status verwendet werden können. An dem als Referenz für die untere Grenze des guten ökologischen Zustands für Tirol vorgeschlagenen Flussabschnitt, dem Fluss Drau, wurden die wichtigsten Gewässerqualitätsparameter für einen alpinen Fluss identifiziert. Von den in einer Literaturrecherche identifizierten Immissionsgrenzwerten wurden Parameter als Indikatoren für verschiedene Auswirkungen der Siedlungsentwässerung auf den Vorfluter ausgewählt. Diese Gewässerqualitätsindikatoren wurden verwendet, um die Aussagekraft verschiedener emissionsbasierter Leistungsindikatoren für Mischwasserüberläufe zu untersuchen. Es wurde festgestellt, dass keiner der traditionell eingesetzten Leistungsindikatoren für Mischwasserüberläufe etwas über akute Einwirkungen auf den Vorfluter aussagt. Das bedeutet, dass für die Bewertung akuter Mischwasserüberlaufauswirkungen der Einsatz von Gewässerqualitätsindikatoren, wie denen in dieser Dissertation festgelegten, unabdingbar ist. Die Gewässerqualitätsindikatoren wurden weiterhin in zwei Studien zur Bewertung von Maßnahmen der Siedlungsentwässerung eingesetzt. Die erste Studie beurteilte die Kosteneffizienz von Maßnahmen im Fluss zur Reduzierung hydraulischer Effekte. Es wurde festgestellt, dass Maßnahmen im Fluss oft effektiver hydraulische Auswirkungen reduzieren als eine Vergrößerung des Regenüberlaufbeckenvolumens. Die zweite Studie verglich zwei verschiedene Arten von Kanalsystemen, Misch- und Trennsystem, und fand heraus, dass vom Trennsystem, welches oft als die bessere Lösung angesehen wird, erhebliche Mengen von Schwermetallen direkt in den Vorfluter geleitet werden.

Um verlässliche Vorraussagen über die Auswirkung von Siedlungsentwässerungsmaßnahmen auf den ökologischen Status machen zu können, ist weitere Forschung notwendig. Wenn das Verständnis der Beziehungen zwischen Wasserqualität und der aquatischen Flora und Fauna verbessert wird und so bessere Immissionsgrenzwerte definiert werden, wird dies die Kosteneffizienz in der Siedlungsentwässerung verbessern. Verlässliche Immissionsgrenzwerte zusammen mit Computersimulationen des integrierten Abwassersystems werden die Palette angewandter Maßnahmen vergrößern, indem sie es ermöglichen die Auswirkungen der Implementierung einer Maßnahme vorherzusagen.

ABSTRACT

The assessment of urban drainage measures with regard to the European Water Framework Directive (WFD) is characterised by considerably increased complexity compared to traditional assessment. In the WFD, the combined approach is specified, that is both emission limits and environmental quality standards have to be complied with. The ambient water quality standards however are not specified yet, except for the statement of the WFD that all surface water bodies have to achieve at least a good ecological status.

The aim of this dissertation was to assess the impact of different urban drainage measures on the environment, especially on rivers. Two measurement campaigns were performed, one to measure the pollution of urban snow caused by traffic, the other one to assess the impact of parking place runoff on the soil of infiltration swales. For the assessment of impacts from the urban drainage system on rivers, the WFD is fundamental nowadays in Europe. Therefore one aim of this dissertation was to describe the good ecological status of rivers in terms of water quality parameters. This is not possible today due to various reasons (e.g. the not yet identified cause-effect relations between biological parameters and abiotic water parameters). Therefore instead it was aimed at the identification of appropriate ambient water quality limits which could be applied as substitutes for the good ecological status. At the river stretch proposed as Tyrolean reference for the lower boundary of the good status, the river Drau, the most important water quality parameters for an alpine river were identified. From the ambient water quality limits identified in the literature review a set of parameters was chosen as indicators for different impacts from the urban drainage system on the receiving water. These receiving water quality indicators were used to evaluate the significance of different emission-based combined sewer overflow indicators, and it was found that none of the traditionally used emission-based CSO indicators was able to describe acute impacts. That means that for the assessment of acute CSO impacts, it is necessary to apply receiving water indicators as those identified in this dissertation. The receiving water indicators further were applied in two different studies for assessment of urban drainage measures. The first study evaluated the cost-effectiveness of in-stream measures to reduce hydraulic impacts and found out that in-stream measures are often more effective for mitigation of hydraulic impacts than an increase of combined sewer overflow basin volume. The second study compared two different types of sewer systems, the combined and the separate sewer system, and found that from the separate sewer system, which is often seen as the better solution, considerable amounts of heavy metals are directly discharged to the receiving water.

To make reliable predictions of the effect of urban drainage measures on the ecological status, further research is necessary. If the understanding of relations between aquatic biota and water quality is improved and thus better ambient water quality limits are established, it will greatly increase the cost-efficiency of urban drainage planning. Reliable water quality limits together with computer simulations of the integrated urban drainage system enlarge the scope of measures, as it will be possible to predict the improvements realised by the implementation of measures (also more uncommon measures as source control measures or in-stream measures can be tested numerically in advance).

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LIST OF ABBREVIATIONS

BOD	Biochemical oxygen demand
COD	Chemical oxygen demand
DWF	Dry weather flow
WWTP	Wastewater treatment plant
CSO	Combined sewer overflow
WFD	Water Framework Directive
N	Nitrogen
P	Phosphorus
PNEC	Predicted no-effect concentration
PE	People equivalent
LC	Lethal concentration
NOEC	No observed effect concentration
IUT	Unit of Environmental Engineering
CD4WC	Cost effective development of urban wastewater systems for Water Framework Directive compliance
MAR	Mean annual rain volume
NE	Number of rain events
ME	Maximum rain event
VQO	Mean annual overflow volume

1 INTRODUCTION

Urban drainage is concerned with the collection and conveyance of wastewater and stormwater from urban areas. The main goal of the first efforts was to improve the hygienic situation in the cities. The wastewater was discharged untreated into the rivers. During the 20th century it was recognised that beside the protection of the uses (like fishing), also the ecological quality of the watercourses is a valuable good which should be conserved. The number of wastewater treatment plants increased. Less and less sewers from settlements were directly connected to the receiving waters. The next step was the regulation of the amount of pollutants in the effluent of the wastewater treatment plants. This approach caused significant improvements of the water quality.

However, to sustain the biological integrity it is also necessary to consider the different features of the water bodies. The stresses a watercourse can handle without loosing its good ecological status depend highly on its characteristics. For the design and the evaluation of urban drainage systems this means that besides the sewer system and the wastewater treatment plant also the receiving water has to be taken into account. This so called integrated approach can help to assure the good ecological status of a water.

These new requirements have led to the need for new planning tools. Urban drainage modelling is a powerful method to design and analyse urban drainage systems. In the beginning the computer programs consisted of flow models for the sewer, today complex flow and water quality models for the integrated urban drainage system are available. These models allow system-wide analysis of water flow and pollutant fluxes (Butler and Davies, 2004).

The main features of the European Water Framework Directive (2000/60/EC) for urban drainage are the combined approach and the list of priority substances. Combined approach means that the more stringent of emission limits and environmental quality standards applies. However, until now environmental quality standards are only proposed for the priority substances and other relevant substances. Besides it is only specified that the water bodies have to achieve a good ecological status. For the evaluation of measures in different parts of the urban drainage system, the definition of the good ecological status given in the WFD is not sufficient. In this dissertation it is shown how it can be dealt with the WFD in urban drainage. The different features of the WFD are explained. Environmental quality standards and ambient water quality based approaches for the assessment of intermittent impacts are reviewed. Indicators for the assessment of urban drainage measures with regard to the WFD have been defined and applied in two case studies. Further two measurement campaigns are presented which determined the contribution of traffic to the pollution of snow and runoff.

As this dissertation has been elaborated during the work at the project CD4WC, many of the results have been developed in the CD4WC project team of the Unit of Environmental Engineering of Innsbruck University (IUT). Therefore the papers included in this dissertation often present the joint contributions of the team members.

1.1 EU WATER FRAMEWORK DIRECTIVE

The European Water Framework Directive (2000/60/EC) was enacted to create a framework for the European water policy. It had to be implemented by the EU member states into their national law until 2003. The overall aim for water bodies, both surface waters and groundwater, is to achieve a good status (Blöch, 1999). For surface water bodies at least a good ecological status (or good ecological potential for heavily modified or artificial water bodies, respectively) has to be achieved, the good status is to be defined using reference conditions (CIS-WG2.3, 2003). River basin management plans have to be prepared which illustrate with which measures the good status will be achieved (CIS-WG2.9, 2002). For this purpose each member state has to identify significant impacts and pressures on the water bodies (Borchardt and Richter, 2003). The WFD established a list of priority substances (2455/2001/EC), these are pollutants which are especially environmental hazardous and thus their emission into the environment are to be reduced. For this purpose environmental quality standards have to be prepared by the EU member states (Lepper, 2002). The WFD includes the combined approach, that means that both emission standards and environmental quality standards are valid and the more stringent applies (Achleitner *et al.*, 2005). Intercalibration procedures will ensure consistent requirements in all EU member states (CIS-WG2.5, 2002), intercalibration is planned among others for the environmental quality standards for priority substances or the reference stretches.

1.2 IMPACTS ON RECEIVING WATERS

Generally receiving waters for wastewater can be running waters, lakes, coastal waters and the sea. In this dissertation only running waters (from creeks to streams) are taken into account because the main proportion of the receiving waters in Central Europe are running waters. Running waters are characterised by the flow which has major influence on the aquatic biocoenosis. According to size and flow, running waters differ fundamentally from creeks to streams and mountain waters to lowland waters (Engelhardt, 1996). Respectively also the biocoenosis differs, the river continuum concept of Vannote *et al.* shows the relative changes of organisms in a river system from headwater to mouth (Wetzel, 2001). In Central Europe, running waters can be classified according to their fish biocoenosis into salmonid waters and cyprinid waters (Schwoerbel, 1999), a classification which is often used for ambient water quality standards (for example in the fish waters directive (78/659/EEC), the urban pollution management manual (FWR, 1998), the draft for an Austrian immission directive (AImVF, Draft 1995) and many others). Due to their different characteristics, rivers also react differently to pollution, for example rivers with high reaeration rate are less likely impacted by oxygen depletion. Based on such general characteristics, various authors developed classification schemes showing the general magnitude of impact of different pollutants for specific types of receiving waters (House *et al.*, 1993), (ATV, 1993), (Schilling *et al.*, 1997), (Borchardt and Sperling, 1997), (BWK, 2001), (Rossi *et al.*, 2004a). The impacts of urban drainage on the receiving water quality have different time and spatial scales (House *et al.*, 1993), they can be classified according to the time scale of their impact as acute, delayed, accumulating and long-term, or according to the type of impact as hydraulic/ hydologic, chemical, physical and bio-chemical (ATV, 1993), (Schilling *et al.*, 1997), (Borchardt and Sperling, 1997;Novotny and Witte, 1997). Wastewater contains a mixture of different pollutants, this can lead to additive, antagonistic or synergistic effects regarding the pollutants' toxicity (Welch and Jacoby, 2004). Only for very few pollutants such interactions are identified, for example for un-ionised ammonia and dissolved oxygen (Gammeter and Frutiger, 1990), (Borchardt, 1992), (FWR, 1998), nitrite and chloride (Orth *et al.*, 2003), (Wimmer *et al.*, 2003) or some heavy metals and water hardness (US EPA, 2002), (78/659/EEC), (Wimmer *et al.*, 2003).

1.3 INTEGRATED SYSTEM

The urban wastewater system consists of the sewer system and the wastewater treatment plant. The sewer system can be a combined system, a separate system or a mixture of the two. Via the sewer system, the wastewater is transported to the wastewater treatment plant. For optimal design of the urban drainage system an integrated approach, which also takes the receiving water into account, is necessary (Harremoës and Rauch, 1999;Hauger *et al.*, 2002), especially with regard to the Water Framework Directive.

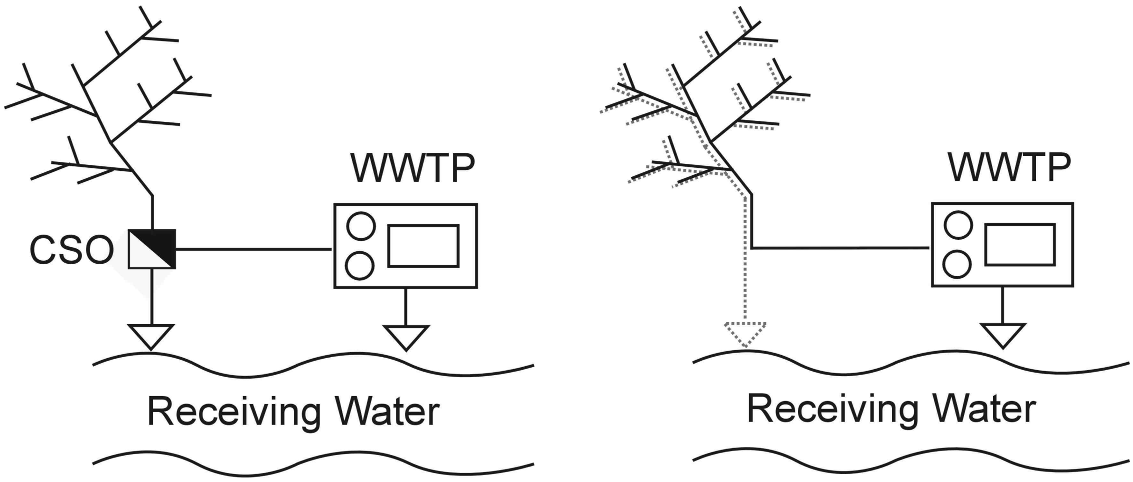

Figure 1.1: Schematic plan of a combined (left) and a separate sewer system (right), after Butler and Davies (2004).

Combined sewer systems are the traditional type of sewer system, here both wastewater and stormwater are conveyed in the same pipe. It is not feasible to design the combined sewer systems to transport the whole wastewater/ stormwater mixture at all times to treatment. Therefore in case of large storm events, the combined system can be hydraulically overloaded. For such cases overflow structures are built into the combined sewer system via which the exceeding water can be released into the next watercourse to prevent flooding of urban areas. Such a discharge event (but also the structure itself) is called combined sewer overflow (CSO). The overflow structures can be channel-like or include a basin in which some of the wastewater can be stored before the basin is filled and the water is discharged. This is an important difference because the wave travels faster than the water itself (Krebs *et al.*, 1999). This means that in case of a rain event, the wave peak reaches the overflow structure before the rainwater that caused the peak, causing an overflow of nearly undiluted wastewater. Further details regarding different types of CSO and their design can be found in Butler and Davies (2004) ATV A 128 (1992) or ÖWAV Regelblatt 19 (1987;Draft 2003). Although CSO spills can discharge significant amounts of pollution, it does also not make sense to transport all wastewater to the wastewater treatment plant because during large storm events, the wastewater treatment plant's performance is decreased by the high hydraulic load and the highly diluted wastewater (Holzer and Krebs, 1998;Rauch and Harremoës, 1996, 1997).

Separate sewer systems consist of one pipe for the sanitary wastewater and one pipe for the stormwater. This system has the advantage that it is not necessary to treat mixed rain and wastewater at the sewage treatment plant. On the other hand, the stormwater is released directly to the receiving water, usually without treatment. Further details can be found in (Butler and Davies, 2004).

The dry weather flow (DWF) is the wastewater that is discharged to the wastewater treatment plant on days without rain. It consists of the domestic wastewater and depending on the catchment also of varying amounts of parasite water (e.g. infiltrating groundwater, fountains or small creeks) and industrial wastewater. Concentrations of various pollutants in dry weather flow have been collected by Brombach *et al.* (2005) in a literature review on worldwide measurement data. The results for Biochemical Oxygen Demand (BOD), Chemical Oxygen Demand (COD), nitrogen, phosphorus, cadmium, lead, copper and zinc are listed in both the **papers IV** and **VI**. The dry weather flow can contain hundreds of different chemicals only from household wastewater. These substances and the resulting risk has been investigated by Eriksson *et al.* (2002). Pollution of stormwater has been measured by various authors (see literature review of Brombach *et al.* (2005), some results are given in the **papers IV** and **VI**, the heavy metal concentrations in surface flow of streets and highways have been measured by Boller (2004), Boller and Häfliger (1996), Dierkes and Geiger (1999), Barbosa and Hvitved-Jacobsen (1999) and reviewed by Welker and Dittmer (2005) (summarised in **paper II**) and roof runoff by Förster (1996;1999), Odnevall Wallinder *et al* (2000), or Boller (1997)). Eriksson *et al.* (2004;2005) and Baun *et al.* (in press) investigated the pollutants in stormwater and the associated risks and developed tools for identification of priority substances in stormwater.

1.4 COST EFFECTIVE DEVELOPMENT OF URBAN WASTEWATER SYSTEMS FOR WATER FRAMEWORK DIRECTIVE COMPLIANCE (CD4WC)

The project "Cost effective development of urban wastewater systems for Water Framework Directive compliance" (CD4WC) aimed at the assessment of urban drainage measures considering both the integrated urban drainage system and the new requirements introduced by the WFD. It was a research project supported by the European Commission under the Fifth Framework Programme and contributing to the implementation of the Key Action "Sustainable Management and Quality of Water" within the Energy, Environment and Sustainable Development Programme. CD4WC is part of the CITYNET cluster. It was coordinated by TU Dresden (Germany) and included the following partners: RUG / BIOMATH (Belgium), IUT Innsbruck (Austria), Panteion University of Athens (Greece), Ruhrverband (Germany), Tiroler Wasserkraft AG (Austria), AQUAFIN NV (Belgium) and PVK Prague (Czech Republic). The project started in February 2003 and ended in September 2006.

CD4WC aimed at the identification of cost-effective measures in the urban wastewater system. For the evaluation, the integrated urban wastewater system was considered, including the receiving water. The project consisted of nine work packages. Some work packages were dealing with the basic system parts: WP1 System Analysis, WP2 Sewer, WP3 Wastewater Treatment Plant, WP4 Receiving Water and WP5 Source Control. The work packages WP6 Operation, WP7 Integration and WP9 Synthesis aimed at integrating the findings of the other work packages. WP8 Economics deals with economics as an instrument in urban wastewater management.

The Unit of Environmental Engineering at the University of Innsbruck was leader of the work packages WP2 Sewer, WP4 Receiving Water and WP5 Source Control and was involved in the work of WP1 System Analysis and WP7 Integration.

1.5 STRUCTURE OF THE DISSERTATION

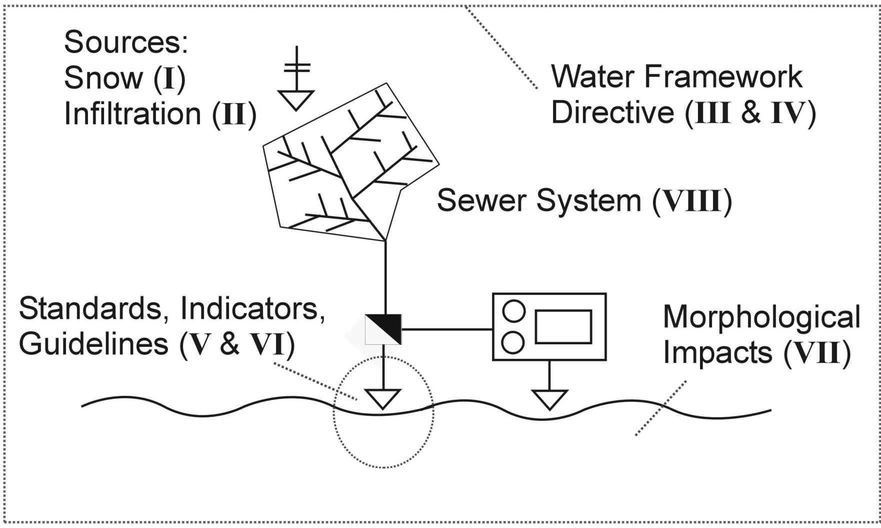

Figure 1.2: Overview on the structure of the dissertation and the allocation of the papers to the different parts of the integrated urban drainage system (number of paper indicated in brackets).

In chapter 2 two measurement campaigns and their results are described. In the first measurement campaign the pollution of urban roadside snow had been determined, because polluted snow can have significant impact on urban drainage and rivers in regions with cold climate and in alpine regions. The measurement campaign on urban snow pollution is described in **paper I** (see annex I):

Engelhard C., Toffol S. D., Lek I., Rauch W. and Dallinger R. (2007). Environmental impacts of urban snow management - The alpine case study of Innsbruck. *The Science of the Total Environment*, **382**, pp. 286–294.

The second measurement campaign assessed the impact of the pollution in parking place runoff on the soil of infiltration swales. Infiltration of runoff from impervious areas is increasingly applied to reduce the amount of stormwater entering the sewer system (or being discharged directly to the receiving water in case of separate sewer systems), the assessment of the impact on soil and groundwater is however still subject of research. The measurement of the infiltration swale pollution is shown in **paper II** (see annex II):

Achleitner S., Engelhard C., Stegner U. and Rauch W. (2007). Local infiltration devices at parking sites - experimental assessment of temporal changes in hydraulic and contaminant removal capacity. *Water Science & Technology*, **55** (4), pp. 193-200.

In chapter 3, the Water Framework Directive and its implementation is illustrated, because the WFD determines the current water policy in the European member states. This chapter is based on the work of the papers:

Achleitner S., DeToffol S., Engelhard C. and Rauch W. (2005). The European Water Framework Directive: Water Quality Classification and Implications to Engineering Planning. *Environmental Management*, **35** (1), 1-9. (not included in the dissertation)

and **paper III** (see annex III):

De Toffol S., Achleitner S., Engelhard C. and Rauch W. (2005). Challenges in the implementation of the Water Framework Directive: case study of the alpine River Drau, Austria. *Water Science & Technology*, **52** (9), pp. 243-250.

and paper **IV** (see annex IV):

Engelhard C. and Rauch W. (2007). Risk analysis and impact assessment of urban stormwater- with emphasis on the EU - WFD. **In**: D. R. Thévenot (Ed.). DayWater: an Adaptive Decision Support System for Urban Stormwater Management, IWA publishing, London. ISBN: 1843391600.

In chapter 4 first the case study of the river Drau is lined out which aimed at the description of the good ecological status and the identification of the most important water quality parameters for the alpine region. The case study is the second part of **paper III** (included in annex III):

De Toffol S., Achleitner S., Engelhard C. and Rauch W. (2005). Challenges in the implementation of the Water Framework Directive: case study of the alpine River Drau, Austria. *Water Science & Technology*, **52** (9), pp. 243-250.

Modelling of river ecology is not possible yet because the cause-effect relationships between water quality and biology are still not identified. Therefore ambient water quality limits were collected instead to identify reliable limits for important wastewater pollutants. Various regulations regarding long-term impacts were compared, for some pollutants the limits of the different regulations are listed in annex IX. These findings are included in **paper IV** (see annex IV):

Engelhard C. and Rauch W. (2007). Risk analysis and impact assessment of urban stormwater- with emphasis on the EU - WFD. **In**: D. R. Thévenot (Ed.). DayWater: an Adaptive Decision Support System for Urban Stormwater Management, IWA publishing, London.

Further a review is given on different approaches used to assess the impacts of urban wet-weather discharges. The impacts of urban wet-weather discharges on the ecology of the receiving water are difficult to assess and the paper reviews the current status of research. The review is included in **paper V** (see annex V):

Engelhard C. and Rauch W. (submitted). Intermittent impacts to receiving water bodies-review on approaches, indicators and limits. Water Research.

In chapter 5 the receiving water indicators chosen for the evaluation of urban drainage measures are explained. They are derived from the main impacts of urban drainage on rivers. These receiving water indicators were used for the evaluation of different urban drainage measures in the project CD4WC (two of these evaluations are shown in chapter 6). Further different traditional emission-based CSO indicators are presented, which are consecutively evaluated on the basis of the receiving water quality indicators. This evaluation is described in **paper VI** (see annex VI):

Engelhard C., De Toffol S. and Rauch W. (2008). Indicators for the CSO performance for compliance with ambient water quality targets. *Urban Water Journal*, **5** (1), 43-49.

Chapter 6 finally shows two examples of application of the receiving water indicators, specified in the previous chapter, to assess the potential of different urban drainage measures. In **paper VII** a study is explained which evaluated the effectiveness of in-stream measures to mitigate the hydraulic impact of peak discharges from combined sewer overflows (see annex VII):

Engelhard C., Achleitner S., Lek I. and Rauch W. (2006). Mitigation measures towards morphological alterations of rivers: The receiving water as part of the integrated wastewater system. *Water Practice & Technology*. **1** (1). doi10.2166/wpt.2006.013

Paper VIII gives the details of a comparison of the impacts on the receiving water from combined and separate sewer systems, and the cost-effectiveness of the two systems (see annex VIII):

De Toffol S., Engelhard C. and Rauch W. (2007). Combined sewer system versus separate system – a comparison of ecological and economical performance indicators. *Water Science & Technology*, **55** (4), pp. 255–264.

1.6 SCIENTIFIC CONTEXT OF THE PAPERS OF THIS DISSERTATION

1.6.1 PAPER I: Environmental impacts of urban snow management - The alpine case study Innsbruck

The accumulation of pollutants in roadside snow has been investigated by various authors, especially in Sweden and Canada. Snow has much higher capacity to accumulate pollutants than stormwater (Glenn and Sansalone, 2002). Therefore several studies measured the pollutant concentrations in snow at urban roads with different traffic densities (Viklander, 1996, 1998, 1999) and in snow banks at highways (Glenn and Sansalone, 2002;Reinosdotter *et al.*, 2005;Sansalone and Glenn, 2002). Especially the partitioning of heavy metals in dissolved and particulate fractions was investigated

(Glenn and Sansalone, 2002;Reinosdotter *et al.*, 2005;Sansalone and Glenn, 2002;Viklander, 1996). Reinosdotter *et al*. (2005) investigated the pollutant concentration in different distances from the road. Viklander (1998) found that the pollutant concentrations in roadside sbow are difficult to predict. Road salt is also an important topic: the roadside snow can contain large concentrations of chloride (Sansalone and Glenn, 2002) which can impact the rivers (Ruth, 2003) or the performance of infiltration devices (Marsalek, 2003). The accumulation of the pollutants can produce highly contaminated runoff in the melt period (Westerlund and Viklander, 2006;Westerlund *et al.*, 2003;Westerlund *et al.*, 2005). Snow disposal into rivers can lead to a decrease of water temperature (Rossi and Hari, 2004).

The case study presented in **paper I** consists on the one hand of a measurement campaign similar to some work done in Sweden or Canada but for the alpine region. Both the impact of different traffic densities and distance from the road were investigated. Further the impacts of the common alpine snow management practice to dispose snow into a river were estimated. This practice is quite common in Tyrol and often leads to problems between fishermen and authorities.

1.6.2 PAPER II: Local infiltration devices at parking sites - experimental assessment of temporal changes in hydraulic and contaminant removal capacity

Stormwater infiltration is a common measure in urban drainage and there exist numerous different types of storm water infiltration devices (for a description see for example Butler and Davies (2004)). It is realised that contamination of stormwater differs according to the type of surface from where the runoff originates and that accordingly also its treatment should be different (Boller, 2004;Land Tirol, 2005;OEWAV R35, 2003). To assess the risk associated with stormwater infiltration, concentration of pollutants in different types of runoff have been measured, for example in roof runoff (Förster, 1996; 1999), or highway runoff (Barbosa and Hvitved-Jacobsen, 1999;Crabtree *et al.*, 2004;Sansalone and Buchberger, 1997); the mechanisms of pollutant transport in the soil of infiltration devices have been investigated (Marcos *et al*., 2002). Different authors investigated pollutant concentrations and distributions in the soil of infiltration devices (Barraud *et al.*, 1999;Datry *et al.*, 2003), (Dechesne *et al.*, 2004;Mikkelsen *et al.*, 1996;Mikkelsen *et al.*, 1997;Mikkelsen *et al.*, 1994) and in roadside green areas of urban regions (Lind and Karro, 1995) and highways (Dierkes and Geiger, 1999).

The study of local infiltration devices at parking sites, described in **paper II** is different from the work presented before. First the soil in infiltration devices receiving runoff from parking lots was measured, second it was searched for correlations of soil pollution with infiltration device characteristics (e.g. age of device, hydraulic conductivity, pH of soil) and last an estimation was made of pollutant loads received by the infiltration devices.

1.6.3 PAPER III: Challenges in the implementation of the Water Framework Directive: case study of the alpine river Drau, Austria

During the implementation of the European Water Framework Directive (2000/60/EC) several important aspects were defined, e.g. water bodies, ambient water quality parameters or reference stretches. To increase the understanding of the implementation process **paper III** gives an overview of that process (more details can be found in Achleitner *et al*. (2005)). The Water Framework Directive (WFD) declares the aim for water bodies as the good ecological status, which is defined based on biological indicators as minor deviation from the natural aquatic biocoenosis. As it is not possible to design urban drainage measures on this basis, **paper III** describes a case study where for the alpine region ambient water quality parameters were searched. Data from the reference stretch for the lower boundary of the good status was used for an attempt to describe the good ecological status in terms of water quality parameters (e.g. nutrients). This approach is similar to what has been done later in (Deutsch and Kreuzinger, 2005) where for each ecoregion general water quality standards were derived. Further the water quality parameters were evaluated in **paper III** regarding to their impact on the biological water quality of the reference stretch and to identify parameters with less relevance for the biological water quality in the alpine region. Such a weighting of indicators has been made by several authors, e.g. Schilling *et al*. (1997), Borchardt and Sperling (1997), Rossi *et al*. (2004a), or ATV (1993).

1.6.4 PAPER IV: Risk analysis and impact assessment of urban stormwater- with emphasis on the EU – WFD

Although the European Water Framework Directive (WFD) influences the work of numerous researches, many of them only know little about this directive and its actual implementation. Therefore **paper IV** outlines the most important features of the Water Framework Directive (2000/60/EC) for urban drainage, e.g. the combined approach or the list of priority substances (2455/2001/EC;Klein *et al.*, 1999;Lepper, 2002), and describes exemplarily the implementation of the WFD in Austria (analysis of current status of Austrian water bodies (BMLFUW, 2005), proposal for environmental quality standards (Wimmer *et al*., 2003), monitoring (Wimmer *et al*., 2002), definition of eco-regions (Chovanec *et al.*, 2000;Koller-Kreimel and Nürnberger, 2002a;Kreuzinger and Deutsch, 2003;Moog *et al.*, 2001;Muhar *et al.*, 2003;Wimmer and Chovanec, 2000), reference stretches (Koller-Kreimel and Nürnberger, 2002b), and groundwater (Philippitsch, 2002)). Following a short description of pollutants in stormwater and wastewater is given (Boller and Steiner, 2002;Brombach *et al.*, 2005;Dierkes, 2000;Eriksson *et al.*, 2002;Welker and Dittmer, 2005) and their impacts on receiving waters (ATV, 1993;House *et al.*, 1993;Lijklema *et al.*, 1993;Schilling *et al.*, 1997). The classification system for receiving waters described in the paper is based on receiving water classifications done by House *et al*. (1993), ATV (1993), Schilling *et al*. (1997), BWK (2001), Rossi *et al.* (2004a), and Burton and Pitt (2002), Schwoerbel (1999), Welch and Jacoby (2004), Wetzel (2001), Greig *et al*. (2005), Orth *et al*. (2003), Borchardt (1992) and Gammeter and Frutiger (1989). The review of different ambient water quality standards included different national and international legislation and guidelines: the European directive on the quality of fresh waters needing protection or improvement in order to support fish life (78/659/EEC), the draft for an Austrian regulation regarding the general restriction of immission in running waters (AImVF, Draft 1995), the Austrian environmental quality standards proposed in the course of the WFD implementation (Wimmer *et al*., 2003), the US water quality standards handbook (US EPA, 2002) and nutrient criteria (US EPA, 2004b), the urban pollution management manual (FWR, 1998), the German BWK guideline 3 (BWK, 2001) and the draft for the new Austrian guideline 19 (OEWAV R19, Draft 2003).

1.6.5 PAPER V: Intermittent impacts to receiving water bodies-review on approaches, indicators and limits

House *et al*. (1993) and Lijklema *et al*. (1993) reviewed the status of research related to assessment of urban drainage impacts on receiving water quality. Ellis (2000) reviewed different approaches to assess the risk of intermittent impacts on receiving water and Preston (2002) the importance of indirect effects for ecological risk assessment. In the project STORM different literature on ambient water quality approaches was reviewed, e.g. (Krejci and Kreikenbaum, 2004;Rossi *et al.*, 2004a). Zabel *et al.* (2001) and Fenz (2002) list the requirements for CSO design in different European states. Karr and Chu (2000) compare the approaches of Index of Biotic Integrity and RIVPACS. Novotny *et al.* (2005) describe different Indices of Biotic Integrity and their relation to anthropogenic stressors.

The aim of **paper V** was to provide an overview of the approaches to protect receiving water quality at wet-weather conditions. Today for intermittent impacts exist such large variety of approaches, differing widely in indicators and underlying principles, that it is difficult to understand their advantages and disadvantages. It was the aim of **paper V** to close this gap and give a critical overview of them, whereat special attention was given to their applicability for wet-weather control planning and to their potential to protect ecological quality. The results show that the benefit of both emission and ambient water quality based approaches is not stringent and scientifically sound evaluated and therefore a combination of the approaches seems favourable.

1.6.6 PAPER VI: Suitability of CSO performance indicators for compliance with ambient water quality targets

The emission-based combined sewer overflow indicators used in **paper VI** are chosen from national and international regulations (ATV-A 128, 1992;Krejci and Kreikenbaum, 2004;OEWAV R19, 1987, Draft 2003;Zabel *et al.*, 2001). The selection of the receiving water indicators is based on different international research (Borchardt, 1992;Harremoës, 1982;House *et al.*, 1993;Hvitved-Jacobsen, 1982;Lijklema *et al.*, 1993;Rossi *et al.*, 2004a;Schilling *et al.*, 1997) and guidelines (BWK, 2001;Danish Engineering Union Wastewater Committee, 1985;FWR, 1998;OEWAV R19, Draft 2003). Such receiving water indicators have been used in different studies, e.g. (Krejci, 2004;Orth *et al.*, 2003;Weilandt *et al.*, 2005). Rauch and Harremoës (1998) showed that total CSO overflow volume is a bad indicator for the oxygen concentration in the receiving water. Lau *et al*. (2002) found that CSO spill frequency / volume can be used as indicator but has significant limitations. The investigation made in this dissertation included more CSO performance indicators and receiving water indicators additionally to total overflow volume and oxygen in the river.

1.6.7 PAPER VII: Mitigation measures towards morphological alterations of rivers: The receiving water as part of the integrated wastewater system

Combined sewer overflows can cause hydraulic impacts especially if the receiving water is relatively small, e.g. Podraza (1999) and Weyand and Schitthelm (2005) found that the main impacts on the investigated rivers resulted from hydraulic impacts. Borchardt (1992) concluded that measures in the receiving water could be an efficient tool for mitigation of hydraulic impacts, if they reduce the shear stress or create hydraulic refugees. Gammeter and Frutiger (1989) proposed limits for hydraulic impacts in the form of erosion frequency limits. This approach has been accepted by the Swiss project STORM (Rossi *et al.*, 2004a;Rossi *et al.*, 2004b) and is therefore included in the software REBEKA (Rauch *et al.*, 2002). The study described in **paper VII** evaluated the cost-effectiveness of different in-stream measures to reduce the erosion frequency. It was found that also restoration measures seem to have potential to effectively mitigate hydraulic impacts from urban wet-weather discharges. This is especially interesting as in Central Europe the main problems are morphological deficits (BMLFUW, 2005;Buffagni *et al.*, 2001;Interwies *et al.*, 2004;Muhar *et al.*, 2000). The efficiency of restoration measures for mitigation of hydraulic impacts is not confirmed yet because so far nobody implemented such measures for mitigation of urban wet-weather discharges. However, it is possible that this will be made in the near future as several guidelines for urban wet-weather discharges stress the importance of the receiving water's morphological quality (e.g. in STORM (Rossi *et al.*, 2004a) or BWK M3 (BWK, 2001)). The study also found that there is insufficient knowledge about the length of the receiving water stretch which is affected by a hydraulic impact. The Austrian Draft for the new guideline 19 (OEWAV R19, Draft 2003) proposes to include a stretch of 1000 times the width of the receiving water surface with minimum 500m and maximum 2500m length. Both, ATV (1997) and BWK (2001) do not specify the length of the hydraulically impacted stretch because in natural watercourses it is not possible to make generally valid statements. In the paper included in the dissertation also simulations were performed to get at least a rough idea of the length of the impacted stretch, in Lek *et al*. (2006) these investigation were pursued further.

1.6.8 PAPER VIII: Combined sewer system versus separate system - a comparison of ecological and economical performance indicators

There has been a long discussion whether combined or separate sewer systems are better and it was generally assumed that separate sewer system produce smaller impacts on the receiving water (Brombach *et al*., 2004). Stormwater from separate sewer systems can contain high concentrations of pollutants, especially heavy metals (Brombach *et al*., 2005) which are discharged to receiving waters and can accumulate in the sediments there (Boller, 1997). Sieker (2003) estimated the pollution loads from combined sewer overflows and from separate system discharges for Germany (with 60% combined systems) and found that significant amounts of pollutants are discharged via separate sewer system outlets. Brombach *et al*. (2004) and Paoletti and Sanfilippo (2004) compared the pollutant emissions from catchments having combined or separate systems. Both found that separate systems are not always preferable. Although the work presented in **paper VIII** is based on a similar approach as in Brombach *et al.* (2004), this paper additionally considered real rain data of different meteorological regimes as well as and different pollutant concentrations. Therefore this paper identified the great importance of rain type and pollutant concentrations on the performance of the two different sewer system types. Sewer separation is the most common applied CSO control measure in the USA (US EPA, 2004a). It is commonly applied to reduce to amount of discharged wastewater, but for example for Atlanta it was calculated that sewer separation would increase the pollutant loads discharged to the receiving waters (US EPA, 1999).

2 TRAFFIC POLLUTION - CONSEQUENCES FOR SNOW MANAGEMENT AND RUNOFF INFILTRATION

Two measurement campaigns were performed to assess the accumulation of pollutants from traffic. The first one, presented in **paper I**, measured concentration of pollutants in urban snow. In the alpine region, melt water from snow can impact the performance of the urban drainage system hydraulically, by decreasing the wastewater temperature and by introducing large amounts of pollutants. Thus for evaluating the performance of the urban drainage system, it is necessary to have knowledge on the pollutant concentrations in the snow. The case study in Innsbruck, presented in **paper I**, was performed because it was found that, despite lots of research in Canada and Scandinavia, there is little information about pollution of urban snow in the alpine region.

Paper II presents the outcomes of a survey on infiltration swales which was accomplished because, although infiltration of surface runoff is common practise in Austria, there is little scientific knowledge regarding the impact on the soil and the risk for groundwater resulting from the infiltration of different types of surface runoff.

2.1 ENVIRONMENTAL IMPACTS OF URBAN SNOW MANAGEMENT

Paper I evaluated the environmental impacts of urban snow management in the alpine case study Innsbruck. It is known that snow in urban areas and at main traffic routes can accumulate significant amounts of pollution, especially heavy metals, suspended solids and chloride. Therefore a measurement campaign was carried out at main streets in the city of Innsbruck. The aim was to quantify pollutants in roadside snow (copper, zinc, lead, cadmium, suspended solids and chloride) and further to assess the potential impact of snow disposal in rivers. For sampling sites, an urban and a rural reference, a site with low traffic density, several sites with high traffic density and a highway site were chosen. Additionally sampling was performed at both 2 m and 4 m distance from the highway. The mean concentration of copper and suspended solids were higher at the high traffic sites and the highway, zinc and lead however were at the low traffic site in a similar range as at the sites with higher traffic. Cadmium was found also in the urban reference in elevated concentrations, indicating that there are other important sources for cadmium beside traffic. The mean chloride concentration was at the urban roads (i.e. low traffic and high traffic sites) similar and at the highway significantly higher. With increasing distance from the highway, the pollutant concentrations decreased: at 2 m distance the mean concentrations of most pollutants were below the mean concentration at low traffic site and at 4 m distance the pollutant concentrations were at the same level as at the reference sites. With the results of the measurement campaign, the pollutant loads discharged into the river Inn due to the snow handling practices in Innsbruck were estimated. The large variations of pollutant concentrations found in the roadside snow samples caused also large uncertainties in the load estimations. Therefore further measurements are necessary to assess the impact of this snow handling practice on the river’s ecology.

2.2 LOCAL INFILTRATION DEVICES AT PARKING SITES

The measurement campaign presented in **paper II** aimed at the investigation of pollutants (hydrocarbons, copper, zinc, lead and cadmium) accumulated in the soil of grassed swales which receive runoff from supermarket parking lots. Infiltration of not or little polluted runoff is a common measure to reduce the stormwater runoff in urban areas. Various guidelines deal with the design and implementation of infiltration devices (e.g. ATV-DVWK-A 138 (2002), ATV-DVWK-M 153 (2000), OENORM B 2506 (2000), OEWAV R35 (2003)) as well as different research projects (among others the project DayWater (DayWater D5.1, 2003;Förster *et al.*, 2004) or CD4WC (CD4WC Deliverable 5.1, 2006)). The pollutants contained in the runoff can accumulate in the soil of the infiltration device. Therefore in the measurement campaign the total pollutant concentrations were determined at various depth (in 0-5 cm, 5-15 cm and 15-30 cm) and compared to a reference sample from outside the swale. Because it is theoretically possible that pollutants from the infiltrated water reach the groundwater, the concentrations in the eluate were determined additionally. The functional capability of infiltration devices can be reduced during their operation, thus this was controlled by measurement of the infiltration capacity, the pH value and the grain size. For neither the pollutant concentrations in the soil nor in the eluate the limits of the Austrian landfill regulation (BGBl. Nr. 146/1996, 2004) were exceeded, except for one sampling site. Based on the results of the measurement campaign and literature data on runoff pollution, an extrapolation was made to estimate the total pollutant load which can be expected to accumulate during the lifetime of the infiltration device. Even after 15 years only for some infiltration swales it is expected that the soil will contain pollutant concentrations above the limits of the Austrian landfill regulation for not contaminated soil (BGBl. Nr. 146/1996, 2004). The measurement campaign showed that infiltration is an adequate for this type of runoff and seems to pose little risk to the environment.

3 WATER FRAMEWORK DIRECTIVE

To be able to assess urban drainage measures in compliance with the aims of the Water Framework Directive, the first step was to understand the Water Framework Directive (WFD). Many points are only defined normatively in the WFD, therefore it was also important to investigate how the directive is implemented. As example the implementation of the WFD in Austria is described.

The work done is included in **paper III** and **IV**:

IV: Engelhard C. and Rauch W. (2007). Risk analysis and impact assessment of urban stormwater- with emphasis on the EU - WFD. **In**: D. R. Thévenot (Ed.). DayWater: an Adaptive Decision Support System for Urban Stormwater Management, IWA publishing, London. ISBN: 1843391600.

III: De Toffol S., Achleitner S., Engelhard C. and Rauch W. (2005). Challenges in the implementation of the Water Framework Directive: case study of the alpine River Drau, Austria. *Water Science & Technology*, **52** (9), pp. 243-250.

3.1 INTRODUCTION TO THE WATER FRAMEWORK DIRECTIVE

The European Water Framework Directive (2000/60/EC) aims at integrated water policy (Barth and Fawell, 2001). It came into force in the year 2000 and is supposed to ensure consistent and clear water policy in the European member states by combining various earlier directives in the field of water (like for example the directive concerning the quality required of surface water intended for the abstraction of drinking water in the member states (75/440/EEC) or the directive on the quality of fresh waters needing protection or improvement in order to support fish life (78/659/EEC), etc.).

The Water Framework Directive aims at a good status for all European water bodies. Further important traits are cross national administration of river basins on the basis of management plans, a reduction of the emission of priority substances, the no-deterioration clause for water bodies, a widespread monitoring of the waters and recovery of costs approach for all water services (Blöch, 1999).

3.1.1 The Combined Approach

The Water Framework Directive features the combined approach, meaning that the more stringent of emission limits and environmental quality standards applies (Achleitner *et al.*, 2005). Therefore it demands the setting of both emission limit values and of environmental quality standards. The combined approach in the Water Framework Directive requires the member states to reduce pollution from point and diffuse sources by applying emission limits on the basis of best available techniques or best available controls. Here old national regulations are still valid, e.g. regulations regarding wastewater treatment plant performance. If the pollution reduction achieved by the emission limits is not sufficient to protect specific receiving waters (that is if the quality objective or quality standard set for the specific receiving water is exceeded) more stringent emission controls have to be defined. The combined approach of the

WFD does not permit to reduce the requirements for wastewater treatment below the emission limits even if the receiving water could cope with the pollution. However, it has to be noted that the EU Water Framework Directive is not European law but instead a guideline to be adapted by the member states. Therefore national water law may occasionally divert from the above (Achleitner *et al.*, 2005).

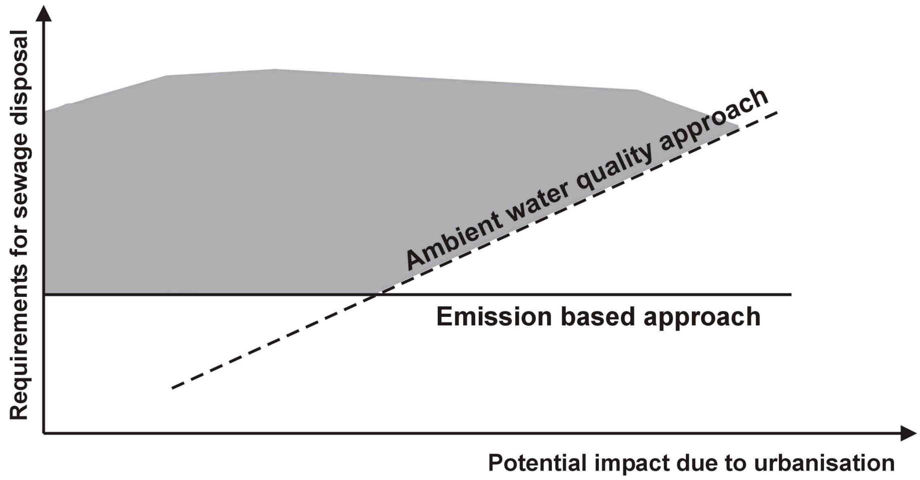

Figure 3.1: Illustration of the combined approach (after Achleitner *et al*. (2005)).

3.1.2 Classification of Surface Waters

Inland surface waters are to be divided in water bodies (CIS, 2003). Each of the water bodies has to have a good status. Further there are two special water body classes: artificial and heavily modified water bodies (CIS-WG2.2, 2003a, 2003b). A water body can be classified as heavily modified water body, if it is substantially changed in character as a result of physical alterations by human activity. An artificial water body is created by human activity. For these two special types of surface water bodies not the good ecological status is to be achieved but reduced requirements denoted “good ecological potential”.

The Water Framework Directive specifies in Annex II that the member states have to characterise their surface waters. For this purpose they have to determine the boundaries of the water bodies and describe them according to the Water Framework Directive’s methodology. As methodology for description either the fixed typology (system A) or the alternative description (system B) has to be applied. The water bodies shall be differentiated by relevant ecoregions, like those proposed for system A (see Figure 3.2). Further type-specific reference conditions for the surface water body types have to be established (CIS-WG2.3, 2003). For the high status, hydromorphological and physicochemical reference conditions are to be established, as well as biological conditions for the good ecological status, respectively for the maximum ecological potential (Deutsch and Kreuzinger, 2005). Further the member states have to carry out an assessment of the likelihood that surface water bodies within the river basin district will fail to meet the environmental quality objectives set in the Water Framework Directive (CIS-WG2.1, 2002).

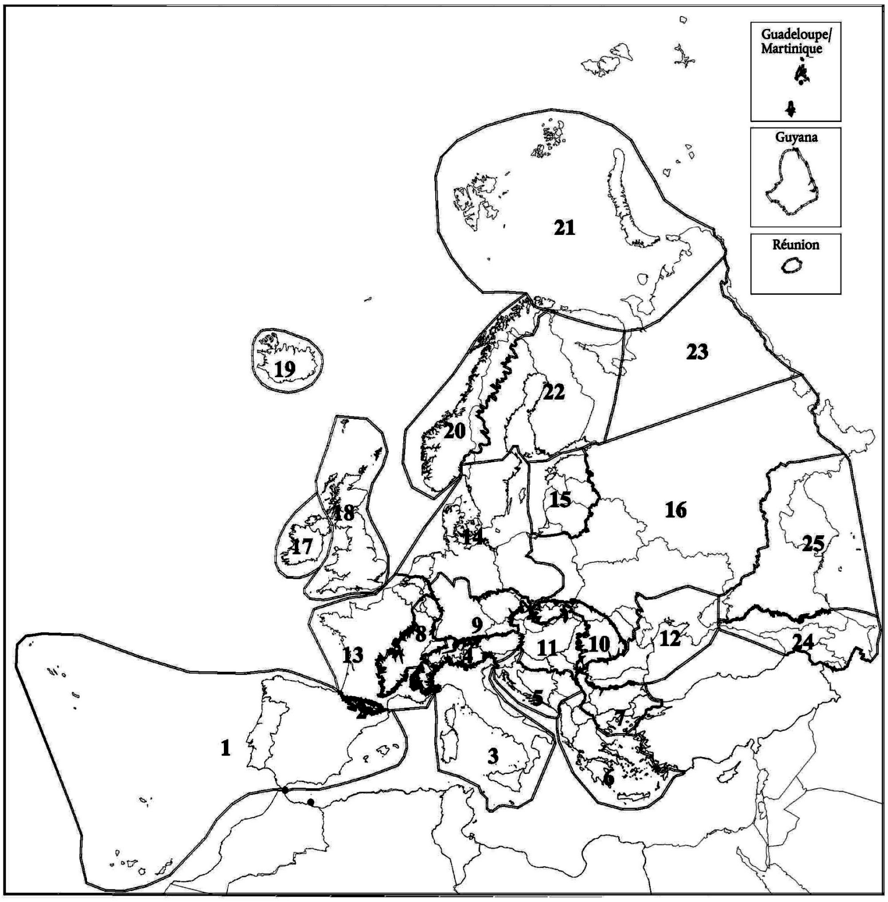

Figure 3.2: System A- Ecoregions for rivers and lakes (Water Framework Directive (2000/60/EC) Annex XI)

1. Iberic-Macaronesian region
2. Pyrenees
3. Italy, Corsica and Malta
4. Alps
5. Dinaric western Balkan
6. Hellenic western Balkan
7. Eastern Balkan
8. Western highlands
9. Central highlands
10. The Carpathians
11. Hungarian lowlands
12. Pontic province
13. Western plains
14. Central plains
15. Baltic province
16. Eastern plains
17. Ireland and Northern Ireland
18. Great Britain
19. Iceland
20. Borealic uplands
21. Tundra
22. Fenno-Scandian shield
23. Taiga
24. The Caucasus
25. Caspic depression

3.1.3 Surface Water Status

"Surface water status" is determined by the poorer of its ecological status and its chemical status. Good surface water chemical status means that pollutant concentrations in the surface water do not exceed limits set in the Water Framework Directive or in other relevant Community legislation. The good ecological status for surface waters is in the WFD only described normatively on the basis of biological elements. The biological elements defined in the WFD are phytoplankton, macrophytes and phytobenthos, benthic invertebrate fauna, and fish fauna. These quality elements should show only small deviations in composition and abundance from the natural status. The implementation of this normative description is to be made by the member states. Supporting for the biological elements, hydromorphological elements as well as chemical and physicochemical elements are to be applied. Hydromorphological elements are hydrology, river continuity and morphology. Physicochemical and chemical elements are on the one hand general parameters, like nutrients or temperature. On the other hand they include specific pollutants, e.g. the priority substances. All surface water bodies have to achieve a good ecological status. Only waters classified as heavily modified or artificial water bodies do not have to comply with the good ecological status but with the reduced requirements of the good ecological potential and a good surface water chemical status.

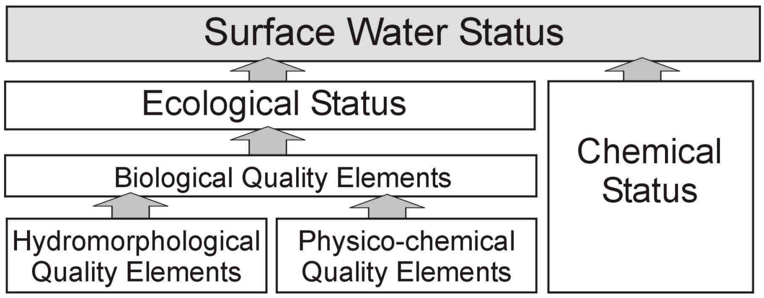

Figure 3.3: Components of the status of surface water bodies.

Environmental objectives are to be set to ensure that good status is achieved (which is to be realised until 15 years after the date of entry into force of the Water Framework Directive) and deterioration in the status of waters is prevented. It is however not defined where the standards apply, directly at each discharge or if there will be a definition of a mixing zone. This is a considerable problem for wastewater discharges (Bleninger *et al.*, 2004). Each member state has to develop a system of five classes for the ecological quality, ranging from bad to high ecological status. The boundary between high and good status as well as the boundary between good and moderate will be established through an intercalibration process to make sure that these boundaries are consistent with the descriptions in the Water Framework Directive (Annex V) and that they are comparable between member states (CIS-WG2.5, 2002). The member states have to present management plans in which they describe the programs of measures to protect and enhance the status of their surface water bodies.

3.1.4 Priority Substances

A list of priority substances is to be developed and regularly updated as prescribed in the Water Framework Directive. The list of priority substances (see 2000/60/EC, annex X) is the basis to establish community-wide harmonised quality standards and emission controls for substances which pose a significant risk to or via the aquatic environment. The priority substances have been identified by using the COMMPS (combined monitoring-based and modelling based priority setting) procedure (Klein *et al.*, 1999). As hazardous those substances are classified which are toxic, persistent and liable to bioaccumulate or other substances which seem to be of similar concern. From the priority substances so called priority hazardous substances have been identified by using information from different hazard assessments, risk assessments and other relevant information like international treaties. According to their “level of concern” (derived by toxicological characteristics and distribution) the substances from the list were classified as priority substances (currently eight substances), priority hazardous substances under revision (14 substances) and priority hazardous substances (11 substances), see Table 3.1. The Water Framework Directive aims during the next 20 years at the progressive reduction of the emission of priority substance, and for priority hazardous substances at the cessation or phasing- out of discharges, emissions and losses.

Table 3.1: List of priority substances of the WFD, identified in (2455/2001/EC).

	PHS	**Name of priority substance**		**PHS**	**Name of priority substance**
1		Alachlor	21	X	Mercury and its compounds
2	(X)	Anthracene	22	(X)	Naphthalene
3	(X)	Atrazine	23		Nickel and its compounds
4		Benzene	24	X	Nonylphenols
5	X (2)	Brominated diphenylethers (1)			(4-(para)-nonylphenol)
6	X	Cadmium and its compounds	25	(X)	Octylphenols
7	X	C_{10-13}-chloroalkanes (1)			(para-tert-octylphenol)
8		Chlorfenvinphos	26	X	Pentachlorobenzene
9	(X)	Chlorpyrifos	27	(X)	Pentachlorophenol
10		1,2-Dichloroethane	28	X	Polyaromatic hydrocarbons
11		Dichloromethane			(Benzo(a)pyrene),
12	(X)	Di(2-ethylhexyl)phthalate (DEHP)			(Benzo(b)fluoranthene),
13	(X)	Diuron			(Benzo(g,h,i)perylene),
14	(X)	Endosulfan			(Benzo(k)fluoranthene),
		(alpha-endosulfan)			(Indeno(1,2,3-cd)pyrene)
15		Fluoranthene (3)	29	(X)	Simazine
16	X	Hexachlorobenzene	30	X	Tributyltin compounds
17	X	Hexachlorobutadiene			(Tributyltin-cation)
18	X	Hexachlorocyclohexane	31	(X)	Trichlorobenzenes
		(gamma-isomer, Lindane)			(1,2,4-Trichlorobenzene)
19	(X)	Isoproturon	32		Trichloromethane (Chloroform)
20	(X)	Lead and its compounds	33	(X)	Trifluralin

PHS Priority Hazardous Substance.
X Identified as priority hazardous substance.
(X) This priority substance is subject to a review for identification as possible “priority hazardous substance”. The Commission will make a proposal to the European Parliament and Council for its final classification not later than 12 months after adoption of this list. The timetable laid down in Article 16 of Directive 2000/60/EC for the Commission's proposals of controls is not affected by this review.
(1) These groups of substances normally include a considerable number of individual compounds. At present, appropriate indicative parameters cannot be given.
(2) Only Pentabromobiphenylether.
(3) Fluoranthene is on the list as an indicator of other, more dangerous Polyaromatic Hydrocarbons.

The European Water Framework Directive contains in annex VIII a list of main pollutants. But the decision of the substances which should be monitored and for which thus environmental quality standards are needed has to be made by the member states. For the priority substances Europe-wide environmental quality standards will be defined after an intercalibration process, but for the other specific pollutants, defined by each member state, the limits will only be valid on a national basis.

3.2 IMPLEMENTATION OF THE WATER FRAMEWORK DIRECTIVE IN AUSTRIA

In Austria it was decided to describe the water bodies according to system B of the Water Framework Directive. For the classification of the types Austria was divided into regions, the so called running water bio-regions (Fließgewässer- Bioregionen), similar to the ecoregions approach of the Water Framework Directive (Moog *et al.*, 2001). The minimal size of the water bodies ranges from 1 km for small water courses to 10 km for large watercourses. The classification is done in two steps, first a basic classification was done which divided water bodies at significant boundaries (e.g. at boundaries of ecoregions or if there were significant hydrological differences). Afterwards a detailed classification was made where stretches with significant anthropogenic impacts (e.g. pollution point sources, migration obstacles etc.) were separated from unimpaired ones (see Figure 3.4). This classification resulted for Austria in about 1000 water bodies with a catchment of more than 100 km^2. This number is not fixed but it is planned to repeat the detailed classification regularly and to adapt the number of water bodies according to the changes of impacts (BMLFUW, 2004).

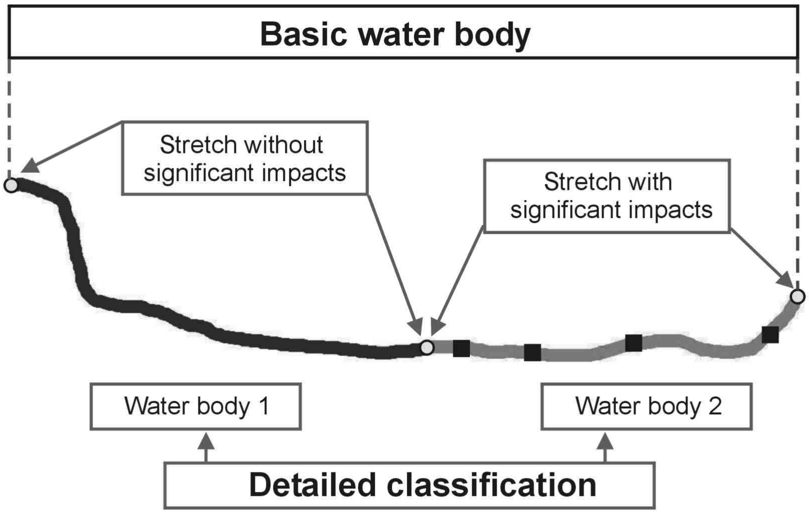

Figure 3.4: Austrian detailed water body classification (after BMLFUW (2004)).

In Austria for the good and the high status, biological and physicochemical quality elements are to be considered, and for the high status also hydromorphological elements. Therefore limits for general physicochemical parameters were proposed, i.e. biochemical oxygen demand (BOD_5), dissolved organic carbon (DOC), nitrate, ortho-phosphate, filtrated total phosphor (Deutsch and Kreuzinger, 2005). The limits were chosen on type-specific reference conditions for each of the 15 running water bioregions. The moderate, poor and bad ecological status shall defined by only biological elements. For the specific pollutants, substances were chosen from relevant European regulations as well as substances which are discharged in significant amounts to the Austrian waterways. From this list of 320 substances those were chosen as relevant for which either local pressures or water quality impacts were identified. For the 33 priority substances identified on a European level and for 43 other relevant substances, identified for the Austria situation, environmental quality standards were determined (Bursch, 2002;Wimmer *et al.*, 2003). In Austria the procedure for the determination of the environmental quality standards is based on a study for the European commission (Lepper, 2002). The Water Framework Directive requires environmental quality standards for water, sediment and biota. But only for water there is currently a good scientific basis and harmonised instructions for sampling and analysis. Therefore in Austria currently only the definition of environmental quality standards for water has been made (Wimmer *et al.*, 2003).

Table 3.2: Austrian proposals for Environmental Quality Standards (EQS) for the priority substances defined in the Water Framework Directive (Wimmer *et al.*, 2003).

Name of priority substance	EQS (µg/l)		Name of priority substance	EQS (µg/l)
Alachlor	3		Mercury and its compounds	-
Anthracene	0.2		Naphthalene	1
Atrazine	1		Nickel and its compounds	-
Benzene	80		Nonylphenols	0.3
Brominated diphenylethers	0.5		(4-(para)-nonylphenol)	
Cadmium and its compounds	-		Octylphenols	1
C10-13-chloroalkanes	0.5		(para-tert-octylphenol)	
Chlorfenvinphos	0.01		Pentachlorobenzene	1
Chlorpyrifos	0.0005		Pentachlorophenol	-
1,2-Dichloroethane	-		Polyaromatic hydrocarbons	-
Dichloromethane	10		(Benzo(a)pyrene),	
Di(2-ethylhexyl)phthalate (DEHP)	8		(Benzo(b)fluoranthene),	
Diuron	0.2		(Benzo(g,h,i)perylene),	
Endosulfan	0.001		(Benzo(k)fluoranthene),	
(alpha-endosulfan)			(Indeno(1,2,3-cd)pyrene)	
Fluoranthene	-		Simazine	1
Hexachlorobenzene	-		Tributyltin compounds	0.001
Hexachlorobutadiene	-		(Tributyltin-cation)	-
Hexachlorocyclohexane	-		Trichlorobenzenes	
(gamma-isomer, Lindane)			(1,2,4-Trichlorobenzene)	
Isoproturon	0.2		Trichloromethane (Chloroform)	-
Lead and its compounds	11		Trifluralin	0.1

For non-metallic pollutants the environmental quality standards are predicted no-effect concentrations (PNEC) which are calculated from toxicological data like lethal concentrations (LC) or no-observed-effect-concentration (NOEC) together with a corresponding safety factor according to the Water Framework Directive (2000/60/EC), annex V. For metals the situation is more complicated. The bioavailability of metals depends on the water chemistry (like on pH, alkalinity, water hardness, presence of complexing agents, etc.). Further, metals have geogenic sources and thus, depending on geochemical and hydrological conditions, the natural concentrations (also called background concentrations) in running waters are highly variable (Wimmer *et al.*, 2003). Therefore the environmental quality standards were defined on the basis of an added risk approach, whereat the environmental quality standard (EQS) is composed of the natural background concentration in the river ($C_{natural\ background}$) and the maximal tolerable additional concentration (C_{max}):

$$EQS = C_{natural\ background} + C_{\max}$$

The added risk approach is based on the assumption that the organisms are adapted to the natural metal concentration and that the same additional exposure leads to the same effect, independent of the background concentration. The maximal tolerable additional concentration was derived by a statistical interpolation method as predicted no-effect concentration from the 5-percentil of a logarithmic transformed distribution of no effect values from different species. The added safety factors vary between one and five. The environmental quality standards for metals in Austria include currently no dependency on water hardness.

Every member state had to assess which of its water bodies might fail to achieve the good status and report the outcomes to the European commission. The results of this assessment were published in the report "Österreichischer Bericht der IST - Bestandsaufnahme" (BMLFUW, 2005). For surface water bodies Austria defined a set of criteria to assess significant pollutant and hydromorphological impacts. For specific pollutants the environmental quality standards were applied. The general chemical status was investigated using the limits defined in a draft for an ambient water quality directive (AImVF, Draft 1995) and the saprobic index (a biological index which covers the impact of organic pollution) (OENORM M 6232, 1997). As hydromorphological impacts the following instances were defined: decreased flow by hydropower plant withdrawal, high flow variations by hydropower plant operation, backwater, migration obstacles and changes in watercourse structure. The assessment (BMLFUW, 2005) found that only about 20 % of the Austrian running surface water bodies will achieve the good status. About 40% will surely fail to achieve the good status and for the rest, the data (either the information on the water body or the data used to define the criteria for the assessment) is not sufficient to clearly define their status. Surface water bodies fail to achieve the good status mostly because of hydromorphological problems (BMLFUW, 2005; Muhar *et al*., 2000). Therefore Austria identified 44% of its water bodies as candidates for heavily modified water bodies (BMLFUW, 2005).

3.3 PRESSURES ARISING FROM THE WFD TO URBAN DRAINAGE SYSTEM DESIGN AND OPERATION

The Water Framework Directive is designed to be the frame of the European water policy and to assure consistent and integrated water management in Europe. The management of the rivers is to be done on the basis of river basin management plans. In the Water Framework Directive the combined approach is specified, that means that for wastewater treatment primarily the old emission standards apply (e. g. the urban waste water treatment directive (91/271/EEC)). Regarding the environmental quality standards, there are still many uncertainties, e.g. regarding monitoring, mixing zones, standards etc. Currently no new requirements from the WFD to wastewater treatment plants are observed. But it is possible that the situation will change with the further assessment of the watercourses in the monitoring programs of the WFD (Weyand, 2006).

In Central Europe wastewater treatment is highly developed and most of the population is connected to wastewater treatment plants (e.g. 92% in Germany or 96% in the UK are connected to main sewers (Butler and Davies, 2004)). In these countries, the pressure on the good ecological quality derives mainly from morphological degradation (Grünebaum *et al.*, 2002;Muhar *et al.*, 2000). Significant costs will arise for the improvement of the situation (Weyand *et al.*, 2005). Usually the amount of pollutants from diffuse sources is in these regions larger than from point sources (Fenz, 2002;

Grünebaum *et al*., 2002). This picture may be different in Southern Europe (e. g. in Greece only 58% and in Portugal 57% of the population is connected to main sewers (Butler and Davies, 2004)).

It is however to be expected that the requirement to reduce the emissions of the priority substances will influence urban drainage but also here it is not clear in which way (Weyand, 2006; Butler and Davies, 2004). Most probably the combined approach of the WFD will influence stormwater treatment (Weyand, 2006; Butler and Davies, 2004). To achieve a good ecological status in heavily urbanised rivers, for stormwater treatment cooperation of experts in the fields of biology, urban drainage and hydraulics is necessary (Weyand and Schitthelm, 2005).

4 DESCRIPTION OF ECOLOGICAL STATUS USING WATER QUALITY PARAMETERS

The aim of optimising the wastewater system with regard to the Water Framework Directive is to achieve a good ecological status. As described above, the good ecological status is composed of a good status of the biological and chemical quality elements. Unfortunately, it was impossible to find cause-effect relations between the running water biocoenosis and chemical parameters which would allow modelling of the Water Framework Directive's biological quality elements. River water quality models do not include aquatic species, and no Europe-wide approaches for predicting aquatic biocoenosis are available.

As modelling the biological quality elements on European scale is currently impossible, the good ecological status had to be described on the basis of physicochemical quality elements. In the case study of the alpine river Drau, the relevant physicochemical parameters to describe the water quality status of an alpine river were identified (the case study is described in more detail in paper **III**).

The next step was a literature review on ambient water quality standards to identify reliable ambient water quality limits. Two different types of regulations were investigated, on the one hand regulations concerning intermittent impacts from stormwater, on the other hand long-term ambient water quality standards. **Paper V** describes the results of a critical literature review of different ambient water quality based approaches to assess the impacts from combined sewer overflows.

In the Water Framework Directive the good ecological status is not clearly defined. It is only stated that these biological quality elements (i.e. phytoplankton, macrophytes and phytobenthos, benthic invertebrate fauna, and fish fauna) should show only small deviations in composition and abundance from the natural status (see chapter 3.1.3). The evaluation of the performance of different urban drainage measures and their comparison in an integrated urban wastewater system can only be done on the basis of computer simulations. Therefore it was searched for models which are able to represent the ecological status of a river which could be included in such simulations.

Most river water quality models are based on a similar approach as the models for wastewater treatment plants, i.e. they describe groups of organisms which convert relevant wastewater components. These river water quality models aim at the simulation of receiving water pollution processes, e.g. eutrophication, acute and chronic toxicity (Rauch *et al.*, 1998), but not at modelling composition and abundance of single aquatic species. For example the river water quality model no.1 (RWQM1) includes the most important processes for C, O, N, and P cycling in a river under aerobic or anoxic conditions (Reichert *et al.*, 2001;Shanahan *et al.*, 2001;Vanrolleghem *et al.*, 2001). The organisms which are responsible for the conversion are grouped together, e.g. X_H contains all facultative anaerobic heterotrophic organisms, X_{ALG} stands for algae and macrophytes, nitrifying bacteria are summarised as X_{N1} (ammonia to nitrite) and X_{N2} (nitrite to nitrate), and all types of consumers are described with the parameter X_{CON}.

There are also approaches to model lotic ecosystem biocoenosis, e.g. based on neural networks. Examples for such neural network prediction systems are RIVPACS, which predicts the natural macroinvertebrate composition of British running waters (Clarke *et* *al.*, 2003), a German study by Schleiter *et al* (1999) or the European project PAEQANN, which aimed at the prediction of the biocoenosis in different European regions (Lek *et al.*, 2003). The prediction of macroinvertebrate community with neural networks performs quite reasonable, but only local and if high resolution data of water quality and quantity as well as of the macroinvertebrate community is available.

4.1 THE GOOD ECOLOGICAL STATUS – THE CASE STUDY DRAU

The first approach to assess the good ecological status was made in the case study of the alpine river Drau in Eastern Tyrol. This river stretch had been chosen because at that time it was proposed as calibration stretch for the lower boundary of the good status for Tyrol (although this has been changed later). The stretch of the Drau, which was proposed as calibration stretch, is in its upper part a residual water stretch (most of the water is bypassed to a hydropower station) and in its lower part it is highly influenced by fast alternations of peak discharges and low discharges due to hydropower generation. There are three wastewater treatment plants whose discharges influence the stretch. The aim of the case study was to identify the relevant physicochemical parameters to describe the water quality status of an alpine river. The results are summarised in the following table.

Table 4.1: Impact parameters importance for an alpine river (x: important; xx: very important; xxx: dominant; (): dependent on local conditions).

	Hydraulic disturbance	Morphology	Dissolved oxygen	Toxic effects	Nutrients	Chloride
Alpine rivers	xx	xxx	/	(x)	/	(x)

The case study showed that the Water Framework Directive will have only little influence on urban drainage in the alpine region. The main impacts on the rivers' ecology result from morphological changes (e.g. urbanisation, flood protection) and hydropower. For alpine rivers emission-based design guidelines for wastewater treatment plants are usually sufficient to secure good water quality. Negative impacts from acute toxic effects could not be excluded in the case study because only data from long-term water quality monitoring with low sampling frequency were available. Therefore it is possible that the WFD will influence alpine stormwater management. Further details of the case study can be found in **paper III**.

4.2 COMPARISON OF DIFFERENT AMBIENT WATER QUALITY STANDARDS FOR LONG-TERM IMPACTS

As modelling of the Water Framework Directive's biological quality elements is impossible, it was necessary to find another approach of dealing with the good ecological status. Therefore a study was conducted which aimed at comparison of various existing limits regarding ambient water quality to identify reliable limit values. It was assumed that compliance with such limits should secure the good ecological status from the water quality side. The findings are also shown in **paper IV**.

Ambient water quality limits can be specified for the water phase, the sediment or biota. Most common are limits for the water phase (Wimmer *et al.*, 2003). These limits are based on toxicological data like lethal concentration (LC) and include safety factors, e.g. the safety factors to be applied for the standards for the priority substances are defined in annex V of the WFD (2000/60/EC). The characteristics of the pollutants and their environmental fate have to be considered additionally. The environmental fate is different for each pollutant. On the one hand the pollutants can be changed by abiotic physical or chemical reactions, e. g. photolysis, volatilization, chemical speciation, sorption (Burton and Pitt, 2002). On the other hand, after uptake, they are subject to reactions in the organisms' metabolism (biotransformation), or can be accumulated in the tissue (bioaccumulation) (Burton and Pitt, 2002). Both, the abiotic as well as the metabolic reactions can lead to changes of the pollutant's toxicity: some substances are mineralised, some changed to less toxic products, but some are also converted to higher toxic substances (Burton and Pitt, 2002;Timbrell, 1993). Heavy metals are usually bound to particles (Burton and Pitt, 2002). Those substances which pose the highest risk to the environment, due to their high persistence and large emission, are identified as priority substances in the Water Framework Directive (see chapter 3.1.4). Ambient water quality limits are derived by the identification of the concentration with the lowest reliable and relevant adverse effect from the toxicological data that is available. From this concentration the limits are extrapolated using safety factors (Whitehouse, 2001), e.g. specified in the WFD in annex V.

Different ambient water quality regulations were investigated, e.g. the European directive on the quality of fresh waters needing protection or improvement in order to support fish life (78/659/EEC), the British Urban Pollution Management Manual (FWR, 1998), the US National Recommended Water Quality Criteria (US EPA, 2002) and the Water Quality Criteria – Nutrients (US EPA, 2004b), the environmental quality standards for Austria proposed during the implementation of the WFD (Wimmer *et al.*, 2003), the Draft of an general ambient water quality regulation for Austria (AImVF, Draft 1995), the environmental quality standards for Germany proposed during the implementation of the WFD (LAWA, 2003), the BWK technical fact sheet M3 from North-Rhine Westphalia (BWK, 2001), the proposal of the ATV working group on nitrite in running waters (ATV, 1994), a study done in North-Rhine Westphalia (Orth *et al.*, 2003), the European directive concerning the quality of bathing water (76/160/EEC), the European directive concerning the quality required of surface water intended for the abstraction of drinking water in the Member States (75/440/EEC) and figures from the book Introduction to Limnology (Schwoerbel, 1999). The standards for nitrite, un-ionised ammonia (NH_3), dissolved oxygen, total phosphorus, copper, zinc and cadmium found in these regulations are listed in annex IX of this dissertation.

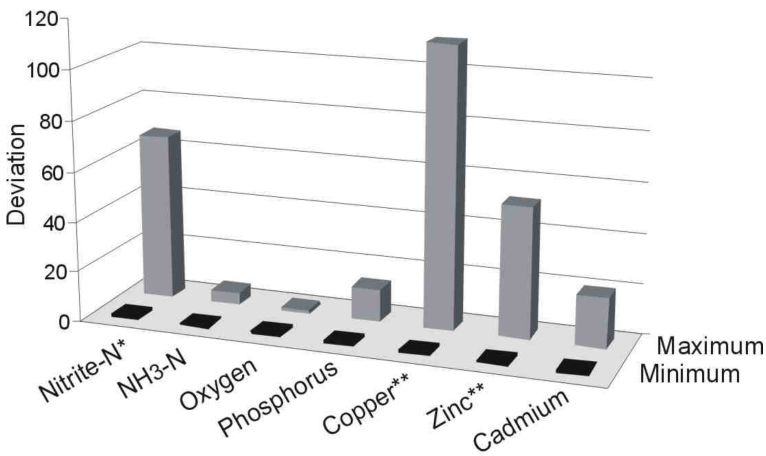

Figure 4.1: Comparison for different substances of the minimum and maximum values of salmonid waters found in ambient water regulations. Deviation= value/ minimum value. * Limits depending on chloride concentration. ** Limits depending on carbonate concentration in the water.

Figure 4.1 shows the relative differences of the limits for salmonid waters found in the above listed regulations. The largest variations were found for those substances which are specified in dependence of another substance, here nitrite, copper and zinc. The differences result partly from the fact that the limits for the different classes of the same regulation already vary widely from each other, e.g. from 5 µg Cu/l at 10 mg $CaCO_3$/l to 112 µg Cu/l at 500 mg $CaCO_3$/l in 78/659/EEC (see Table IX-5 in annex IX).

Also for cadmium relatively high variations were found. Cadmium is identified as priority hazardous substance in the WFD. Therefore it aims at the phasing out of its emissions and intercalibrated limits will be set for cadmium during the implementation (for Austria, see Wimmer *et al*. (2004)).

General parameters as oxygen concentration, phosphorus, nitrogen, salinity, temperature or pH are type-specific parameters, i.e. the natural concentrations of these parameters are quite specific and even varying for different sections of the same river. To include this diversity, in the USA the standards for nutrients (i.e. phosphorus, nitrogen, chlorophyll and turbidity) are defined specifically for each ecoregion (US EPA, 2004a). Also in the Austrian implementation of the Water Framework Directive limits for general parameters have been defined for each Austrian running water type region (Deutsch and Kreuzinger, 2005).

Although for un-ionised ammonia and minimum oxygen concentration there are also type-specific differences, the variations between different regulations are smaller than for e.g. heavy metals or phosphor, probably because the acute toxic properties of these substances are the driving factor for setting limits.

4.3 INTERMITTENT IMPACTS TO RECEIVING WATER BODIES

Paper V gives a critical review of different approaches to assess the impact on rivers from the sewer system at wet-weather conditions. It gives a short introduction into biological assessment methods and presents emission-based and ambient water quality approaches to assess intermittent impacts from the urban drainage system.

The protection of the rivers' ecology got increasingly of importance (Lijklema, 1995). The Water Framework Directive is only one of the most recently enacted regulations, e.g. already the US Clean Water Act (CWA, 1972) required to maintain the "biological integrity" of the waters. Respectively in urban drainage new methods were needed. The traditional approaches used by civil engineers to design the urban drainage system are emission standards. These standards are often defined using Best Available Technology approaches (Ragas *et al.*, 2005), which means the standards are defined such that they can be fulfilled with currently established technologies. However, there is not even relation of emission limits and water quality of the receiving water, and respectively even less with ecological quality. The proceeding development and application of computer models in urban drainage initiated the formulation of ambient water quality based approaches in civil engineering for the assessment of stormwater impacts (Rauch *et al.*, 1998). These approaches aim at the description of the water quality status of the receiving water in terms of physicochemical parameters. The capability of these approaches to protect the ecological quality though has not yet been scientifically sound proofed. Only few case studies have been performed (e.g. (Gammeter, 1996), (Fuchs, 1998), (Podraza and Widera, 1998) or (Orth *et al.*, 2003)). The validation of the ambient water quality based approaches would require investigation of the integrated urban drainage system and thus huge monitoring efforts (Vanrolleghem *et al.*, 1999). Both assessment of water quality with biological indicators and toxicity testing are currently not sufficiently developed for application in urban wet-weather flow planning.

In the Water Framework Directive the combined approach is specified. Thus in future the emission guidelines for the design of wet-weather flow control will still be valid and be combined with ambient water quality based approaches. However, the validation of the appropriateness of emission and ambient water quality approaches to protect the ecological quality is necessary.

5 INDICATORS FOR THE ASSESSMENT OF URBAN DRAINAGE MEASURES

Indicators for stormwater management can be direct or indirect measurements of conditions or elements that indicate trends or responses of watershed conditions to stormwater management activities and can be selected to measure stress or the activities that lead to impacts on receiving waters, to assess the resource itself, and to measure the regulatory compliance (Burton and Pitt, 2002).

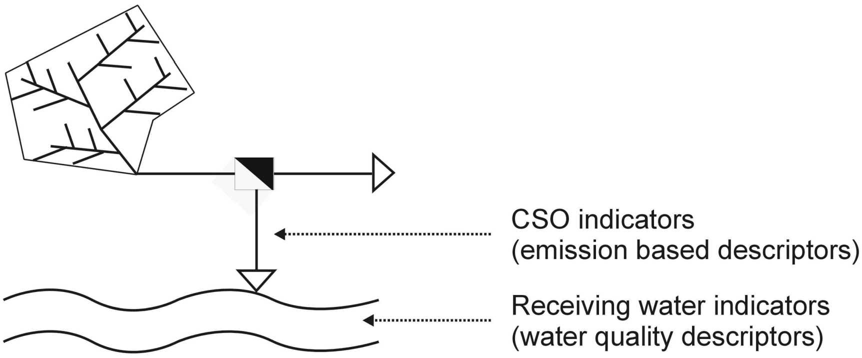

Figure 5.1: Combined sewer overflow and receiving water indicators.

In the following indicators for the assessment of the impact from urban drainage measures on the receiving water are listed. The indicators are selected for assessment of the impacts from wet-weather discharges, caused either by combined sewer overflows or storm sewer outlets. In chapter 5.1 receiving water indicators are introduced, they describe physical and chemical impacts on the water quality. They were defined because ecological quality in terms of biological quality elements could not be described in the computer model.

In chapter 5.2 CSO performance indicators are described. These are emission-based indicators which are used traditionally in urban drainage, for example in CSO design guidelines. In chapter 5.3, the CSO performance indicators are tested if they are correlated with the receiving water indicators, that means it was investigated if the CSO performance indicators describe the impact of the CSO on the water quality of the receiving water.

5.1 RECEIVING WATER INDICATORS FOR URBAN WET-WEATHER DISCHARGES

5.1.1 Receiving Water Indicators

The following receiving water indicators shall describe the status of the receiving water in terms of environmental criteria, more precisely its hydraulic, biochemical and chemical status. One indicator was defined for each of the impact types: hydraulic impacts, oxygen depletion by input of organic substances, eutrophication, acute toxic effects, and accumulation of persistent substances. It is assumed that these indicators are able to describe urban drainage impacts on the ecological status of the receiving water because these impact types have been identified by various other workers (ATV, 1993;Borchardt and Sperling, 1997;BWK, 2001;FWR, 1998;House *et al.*, 1993;Lijklema *et al.*, 1993;Rossi *et al.*, 2004a;Schilling *et al.*, 1997).

The following receiving water indicators have been chosen. As indicator for

- hydraulic impacts: the erosion frequency (Gammeter and Frutiger, 1989;Rauch *et al.*, 2002),
- depletion by input of organic substances: the critical oxygen deficit (BWK, 2001),
- eutrophication: the total nitrogen load per year,
- acute toxic effects: the un-ionised ammonia concentration exceeded continuously for a period of one hour per year, and
- accumulation of persistent substances: the copper load discharged per year.

The receiving water indicators have been defined to evaluate the results of numerical simulations of the integrated urban drainage system. Here they can be used for relative comparison (to compare scenarios and determine from which the impact is lowest, as it has been done in the studies described in chapter 6). For the indicators hydraulic impact (erosion frequency), oxygen depletion by input of organic substances (critical oxygen deficit), and acute toxic effects (un-ionised ammonia concentration), the results can also be compared with limits given in ambient water quality based regulations (erosion frequency: (Gammeter and Frutiger, 1989;Rossi *et al.*, 2004a); acute oxygen deficits: (BWK, 2001;Danish Engineering Union Wastewater Committee, 1985;FWR, 1998;OEWAV R19, Draft 2003;Rossi *et al.*, 2004a); acute toxic effects of un-ionised ammonia: (BWK, 2001;FWR, 1998;OEWAV R19, Draft 2003;Rossi *et al.*, 2004a)).

Examples for both applications are described in chapter 6: in **paper VII** the effect of the measures was compared relative to each other, but also the magnitude of the measures necessary to comply with the erosion frequency limits defined by Gammeter and Frutiger (1989) was calculated.

For the computed simulation results of **paper VIII** only a relative comparison of the indicators was possible. As the rivers used in this study as receiving water were artificial and were assumed to be initially clean, no exceedance of limits occurred- and thus the different simulation scenarios could not be compared with each other on the basis of environmental quality standards.

5.1.2 Hydraulic Impact

In urban areas large parts of the ground are impervious so that only a small proportion of the stormwater can infiltrate. The runoff coefficient (that is the proportion of rainfall that contributes to runoff from the surface) is in residential areas between 0.3 and 0.7, in the city centre even between 0.7 and 0.95 (Butler and Davies, 2004). Urbanisation results in an increase of flow in the rivers, especially the peaks can be magnified from 2 to 50 times to flows of predevelopment. Especially the frequency of flood events with low return periods (e.g. 2-years flood) will be multiplied (Roesner and Bledsoe, 2003).

Thus in urban areas, large parts of the rain water enters the sewer system with only short delay, often resulting in an increase of flow in the receiving water usually before the flow increase from the natural catchment arrives at the point of the urban discharge. The arrival of the discharge before the natural peak is problematic because natural floods give an advance warning due to a slow increase of flow. This allows the organisms to search for refugees on time. Artificial peaks from combined sewer overflows or storm sewer outlets give no such warning. The pollution of the discharge can influence the drift of the aquatic biocoenosis, e.g. oxygen deficits can lead to deliberate exposure to the flow of benthic macroinvertebrates to improve oxygen uptake (Borchardt, 1992;Gammeter and Frutiger, 1989). The increase of flow can also multiply the number of erosion events in the river. The losses of the populations by drift have to be compensated. The potential of a river to compensate such losses is called in the German-speaking world “Wiederbesiedlungs-potential” (recolonisation potential) (BWK, 2001). The recolonisation potential is determined by the morphological characteristics of the river. It is reduced by migration obstacles, if habitat diversity is lost (e.g. by channelisation or channel stabilisation measures), and if there is no connection to confluents with a natural status. A low recolonisation potential makes a river more susceptible to all kinds of disturbances (BWK, 2001;Frutiger and Gammeter, 1996).

Standards

There are different indicators for intermittent hydraulic impacts, most important of CSOs. One approach is to restrict the additional discharge on the basis of the natural flow of the river, as done in the BWK M3 (BWK, 2001).

Objectives of BWK M3 for hydraulic impacts are fulfilled if:

$$Q_{E1,zul.} < 1.0 * Hq_{1,pnat} * \frac{A_{red}}{100} + x * Hq_{1,pnat} * A_{Eo} [l/s] \qquad [1]$$

$Hq_{1,\,pnat}$	... potential natural annual flood
A_{red}	... paved area of attached urban area
A_{Eo}	... surface catchment of the watercourse
$Q_{e1,\,zul}$	... acceptable critical yearly influent discharge
x	... multiplication factor for the acceptable increase of discharge by anthropogenic influences, normally 0.1

The draft for the new ÖWAV technical fact sheet 19 (Draft 2003) includes as limit for hydraulic impact 0.1 to 0.5 times the 1-year flood whereat the percentage has to be chosen considering the receiving water’s morphology (e.g. sediment type, channel geometry and recolonisation potential). In Switzerland not the flow is restricted but the number of erosion events. Here it is assumed that the impact is caused by disturbance of the river bed. The number of additional erosion events caused by urban drainage that is acceptable for a river depends on its recolonisation potential and the availability of refugees. It varies between 0.5 (low recolonisation potential and little refugees) and 10 events per year (high recolonisation potential and many refugees available) (Frutiger and Gammeter, 1996).

Modelling of the hydraulic impact:

The determination of the erosion frequency in the computer simulations presented in the papers of this dissertation (**paper VI**, **paper VII** and **paper VIII**) have been made with the software REBEKA which uses the approach by Meyer-Peter (see below). Lek *et al.* (2006) compared the results of different sediment transport formulas on the computed number of erosion events and length of the erosion stretch. Large variations between the results were found, showing how important the choice of the right sediment transport formula is. Lek *et al.* (2006) also investigated the effect of multiple discharge points at a river stretch and how the waves interfere with each other, an effect which can be of great importance.

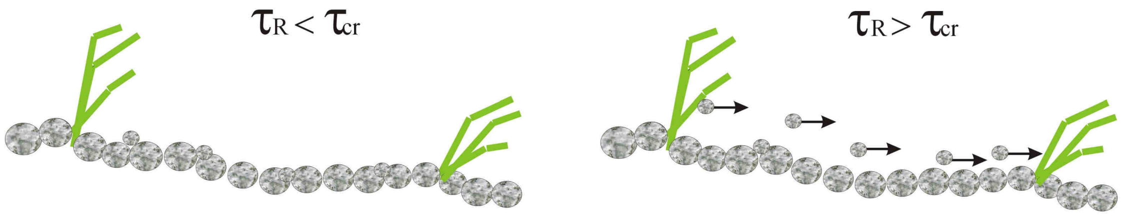

Figure 5.2: Illustration of the approach by Meyer- Peter.

The critical discharge was determined based on the model underlying the Swiss software REBEKA (Rauch *et al.*, 2000).

Actual shear stress: $\tau_R = g \cdot \rho \cdot R \cdot I \left(\frac{k \cdot d_{90}^{1/6}}{26} \right)^{3/2}$ $[N / m^2]$ [2]

τ_R	... actual bottom shear stress [N/m²]
ρ	... density of the water [kg/m³]
g	... acceleration of gravity [m/s²]
k	... Strickler coefficient of side friction [$m^{1/3}$ /s]
R	... hydraulic radius [m]
I	... river bed slope [-]

Critical shear stress: $\tau_{cr} = 0.047 \cdot g \cdot (\rho_S - \rho) \cdot d_m$ $[N / m^2]$ [3]

τ_{cr}	... actual bottom shear stress [N/m²]
ρ	... density of the water [kg/m³]
ρ_s	... density of the bed material [kg/m³]

Receiving Water Indicator

As indicator the erosion frequency in the receiving water was chosen, based on the Swiss approach by Frutiger and Gammeter (1996). The erosion frequency was determined using the approach of Meyer- Peter, whereat an erosion event takes place if the actual shear stress in the river exceeds its critical shear stress (see also **paper VII**).

5.1.3 Oxygen Depletion

Acute oxygen depletion is mainly caused by the wastewater in the CSO discharge because surface runoff contains significantly less readily degradable organic matter. Wastewater discharges can cause acute toxic effects due to immediate oxygen depletion by easily degradable organic material, but also delayed effects of oxygen depletion due to the slower degradation of particulate organic matter. Standards for acute oxygen impacts are defined in various guidelines and regulations (e.g. Danish guideline (Danish Engineering Union Wastewater Committee, 1985), US Ambient Water Quality Criteria for Dissolved Oxygen (US EPA, 1986), Urban Pollution Management Manual (FWR, 1998), the draft of the new ÖWAV R19 (Draft 2003), BWK M3 (BWK, 2001), STORM (Rossi *et al.*, 2004a)). These are described in detail in **paper V**.

Standards

Both the BWK M3 (BWK, 2001) and the ÖWAV R19 (Draft 2003) define 5 mg/l as standard for dissolved oxygen in the water phase. Low oxygen concentrations cause always stress, even if they are not acute lethal. Therefore the Urban Pollution Management Manual (FWR, 1998) is based on a concentration/duration threshold approach, which defines for different oxygen concentrations the minimum time for recovery (expressed in terms of frequency, that is return period). The standards for (a) ecosystem suitable for sustainable salmonid fishery, (b) ecosystem suitable for sustainable cyprinid fishery and (c) marginal cyprinid fishery ecosystem are shown in the following table. The limits include an interdependency of the standards for dissolved oxygen and un-ionised ammonia. At high un-ionised ammonia concentrations, higher standards for dissolved oxygen apply.

Table 5.1: Fundamental Intermittent standards for dissolved oxygen (mg/l) – concentration/duration threshold not to be breached more frequently than shown. Standards for ecosystem suitable for (a) sustainable salmonid fishery, (b) sustainable cyprinid fishery and (c) marginal cyprinid fishery ecosystem (FWR, 1998).

		1 hour				6 hours				24 hours		
		a	**b**	**c**		**a**	**b**	**c**		**a**	**b**	**c**
1 month		5	4	3		5.5	5	3.5		6	5.5	4
3 months		4.5	3.5	2.5		5	4.5	3		5.5	5	3.5
1 year		4	3	2		4.5	4	2.5		5	4.5	3

Notes
1. These limits apply when the concurrent un-ionised ammonia (NH_3-N) concentration is below 0.02 mg/l. The following correction factors apply at higher concurrent un-ionised ammonia concentrations:
0.02 - 0.15 mg NH_3-N/l: correction factor = + (0.97 x log_e(mg NH_3-N/l) + 3.8) mg O/l
>0.15 mg NH_3-N/l: correction factor = +2 mg O/l.
2. A correction factor of 3 mg O/l is added for salmonid spawning grounds.

Modelling of the critical oxygen deficit:

The Streeter-Phelps model, developed in 1925, is one of the earliest models for modelling oxygen depletion (Tchobanoglous, 1987). It calculates the oxygen deficit which is caused by a wastewater discharge, taking into account deoxygenation by various stream organisms due to oxidation of BOD, and reaeration. It is one of the most frequently used river models (Ray, 1995). More advanced models also include the effect of photosynthesis (Simonsen and Harremoës, 1978), differentiate between immediate and delayed oxygen depletion (Harremoës, 1982;Hvitved-Jacobsen, 1982), or represent daily oxygen fluctuations (Harremoës *et al.*, 1996;Jacobsen *et al.*, 1996). State of the art river water quality models include as processes influencing the oxygen concentration: reaeration, biodegradation, sediment oxygen demand, photosynthesis, respiration and nitrification (Rauch *et al.*, 1998).

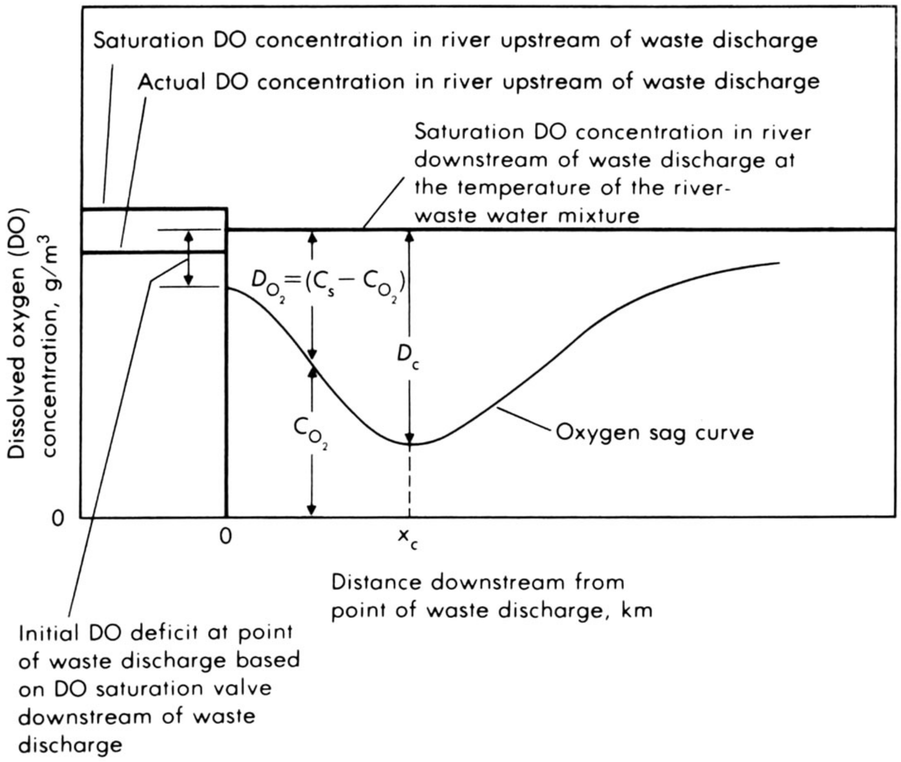

Figure 5.3: Characteristic oxygen-sag curve obtained using the Streeter-Phelps equation (Tchobanoglous, 1987).

For the calculation of the critical oxygen deficit in the studies presented later in this dissertation the Streeter-Phelps model was used as described in the BWK M3 (BWK, 2001) (see formulas formulas [4]-[9]).

Calculation of the critical oxygen deficit based on the approach of Streeter and Phelps (BWK, 2001):

Streeter-Phelps formula: $D(t) = D_o \cdot e^{(-k_2 \cdot t)} + \frac{k_1 \cdot c_G}{k_2 - k_1} \cdot \left[e^{(-k_1 \cdot t)} - e^{(-k_2 \cdot t)} \right]$ [4]

[mg/l]

D_0	initial oxygen deficit [mg/l]
k_1	degradation rate [1/h] (default BWK M3: 0.03/h)
k_2	reaeration rate [1/h] (see formula [6])
c_G	BOD concentration [mg/l]

The critical oxygen deficit (Dc) is calculated from that as:

Critical (maximal) oxygen deficit: $Dc = \frac{k_1}{k_2} \cdot c_G \cdot e^{(-k_1 \cdot t_c)}$ [mg/l] [5]

k_1	degradation rate [1/h];(default BWK M3: 0.03/h)
k_2	reaeration rate [1/h]; (see formula [6])
c_G	BOD concentration [mg/l]
t_c	critical flow time [h]

Reaeration rate: $k_2 = \frac{\left[\left(3 + \frac{40}{k_{st}} \right) \cdot \frac{v_m}{h_m} + 0.5 \right]}{h_m \cdot 24}$ [1/h] [6]

k_{st}	Manning-Strickler coefficient [$m^{1/3}$/s]; (default values for different river bed types proposed in BWK M3, usually 20-60 $m^{1/3}$/s)
h_m	mean water level at mean low water flow (MNQ) [m]
v_m	mean velocity at MNQ [m/s]

The mean water level at MNQ (h_m) is derived with the Manning Strickler formula.

$v = k_{st} \cdot \sqrt{I} \cdot R_h^{2/3}$ [m/s] [7]

v	flow velocity [m/s]
k_{st}	Manning-Strickler coefficient [$m^{1/3}$/s]
h_m	mean water level at mean low water flow (MNQ) [m]
R_h	hydraulic radius [m]

Critical flow time: $t_c = \frac{1}{k_2 - k_1} \cdot \ln \left\{ \frac{k2}{k1} \left[1 - \frac{Do(k_2 - k_1)}{k_1 \cdot c_G} \right] \right\}$ [h] [8]

With the critical oxygen deficit, the minimum oxygen concentration can be calculated (the saturation of oxygen in water depends on the water temperature):

Minimum oxygen concentration: $c_{DO,\,min} = c_{DO,\,saturation} - D_c$ [mg/l] [9]

$c_{DO,\,saturation}$ ... oxygen concentration at saturation

Receiving Water Indicator

As indicator for acute oxygen depletion the critical oxygen deficit was chosen. The critical oxygen deficit is the oxygen that is consumed by degradation of the BOD in the river. It has been calculated on the basis of the approach of the BWK M3 (see formulas [4]-[9]). Originally it was intended to use the limits for acute oxygen deficits as receiving water indicator, as they are specified in the Urban Pollution Management manual (FWR, 1998). After the first simulations, however, it was found that the investigated scenarios (unpolluted rivers, runoff pollution according to literature etc) resulted in only small variations of the oxygen dynamics, so that a comparison of different simulation scenarios was not possible. Therefore the hour with the highest BOD load was determined instead. From this load the mean concentration of BOD and the mean discharge were derived (for the period of one hour) and with the formulas [4]-[8], the critical oxygen deficit was calculated.

5.1.4 Eutrophication

As already pointed out, nutrient standards have to be defined river type-specific. The substances that cause eutrophication are nitrogen and phosphorus, the two main plant nutrients. The loads discharged by combined sewer overflows in Austria have been estimated to amount to approximately 1-2 % for nitrogen and 2% for phosphorus of the total emissions from the urban drainage system (Fenz, 2002).

Receiving Water Indicator

The total nitrogen load discharged per year into the river has been chosen as indicator for the input of plant nutrients from the urban drainage system.

Nitrogen load discharged per year: $N_{load} = \frac{\sum N_{load\,i}}{n}$ [t/a] [10]

n ... number of years of the simulation

5.1.5 Acute Toxic Effects

There are several substances in wastewater which could cause acute toxic effect but ammonia is usually present in high concentrations and thus can lead to problems in the receiving water. Ammonia and un-ionised ammonia are in equilibrium whereat the amount of un-ionised ammonia depends on the water temperature and the pH. It can be calculated according to the following formula (BWK, 2001):

Un-ionised ammonia: $NH_3 - N = \frac{1}{\left[10^{(p_{Ks} - pH)} + 1\right]} \cdot NH_4 - N$ [mg/l] [11]

with $pKs = 0.09018 + \frac{2729.92}{273.2 + T}$ [12]

T ... temperature in °C

In the simulations, the temperature variation in the river was calculated according to the following formula (Rauch *et al.*, 2000), which is based on an investigation of the temperature dynamics in an alpine river.

$$Temperature\,(m) = \frac{T_{\min} + T_{\max}}{2} + \frac{T_{\min} - T_{\max}}{2} \cdot \cos\left[\frac{(2\pi \cdot (m+4))}{12}\right] \quad [°C] \qquad [\ 13\ ]$$

T ... temperature [°C]
T_{min} ... minimal temperature in February
T_{max} ... maximal temperature in August
m ... value of the month (January = 1; December = 12)

Standards

Acute toxic impacts by un-ionised ammonia are an important topic regarding combined sewer overflows. The German BWK M3 specifies that the concentration has to be below 0.1 mg NH_3-N/l. The ÖWAV R19 (Draft 2003) defines standards for ammonia of 1 hour duration and distinguishes between salmonid waters, where 2.5 mg NH_4-N/l are not to be exceeded, and cyprinid waters, where concentration above 5 mg NH_4-N/l are seen as critical (it is assumed that with these limits it is made sure that the un-ionised ammonia concentration does not exceed 0.1 and 0.2 mg NH_3-N/l, respectively). The UPM sets the standards for un-ionised ammonia based on concentration/duration thresholds with specified return periods (see Table 5.2), on the same approach as for dissolved oxygen. Here again, there is an interdependency of the standards for dissolved oxygen and un-ionised ammonia: at low oxygen concentrations, lower standards for un-ionised ammonia apply.

Table 5.2: Fundamental Intermittent standards for un-ionised ammonia (mg NH_3-N/l) – concentration/duration threshold not to be breached more frequently than shown. Standards for ecosystem suitable for (a) sustainable salmonid fishery, (b) sustainable cyprinid fishery and (c) marginal cyprinid fishery ecosystem (FWR, 1998).

	1 hour			**6 hours**			**24 hours**		
	a	**b**	**c**	**a**	**b**	**c**	**a**	**b**	**c**
1 month	0.065	0.150	0.175	0.025	0.075	0.100	0.018	0.030	0.050
3 months	0.095	0.225	0.250	0.035	0.125	0.150	0.025	0.050	0.080
1 year	0.105	0.250	0.300	0.040	0.150	0.200	0.030	0.065	0.140

Notes
1. These limits apply when the concurrent dissolved oxygen concentration is above 5 mg/l. At lower concurrent dissolved oxygen concentrations the following correction factor applies:
<5 mg/l DO, multiplicative correction factor = 0.0126 (mg DO/l)$^{2.72}$
2. The standards also assume that the concurrent pH is greater than 7 and temperature is greater than 5°C. For lower pH and temperatures the following correction factors apply:
pH <7, multiplicative correction factor = 0.0003(pH)$^{4.17}$
Temperature <5°C, multiplicative correction factor = 0.5

Receiving Water Indicator

As receiving water indicator the concentration of un-ionised ammonia (mg NH_3-N/l) was chosen. As for dissolved oxygen, also for un-ionised ammonia originally an evaluation of the computed results with the Urban Pollution Management Manual (FWR, 1998) was planned for the investigation. However, as for dissolved oxygen, for un-ionised ammonia also no significant exceedances of the limit were found.

Thus for comparison of the different simulation scenarios the un-ionised ammonia indicator was defined as the un-ionised ammonia concentration that is reached or exceeded for one hour continuously in the receiving water:

$$C_{NH_3-N,\max} = \max\left(\sum^{1\,\text{hour}} C_{NH_3} \geq C_{NH_3-N,\max}\right) \quad [\text{mg NH}_3\text{-N/l}] \qquad [\,14\,]$$

5.1.6 Accumulation of Persistent Substances

There are various persistent toxic substances in wastewater but for most of them there is only little known about their pathways. Heavy metals are measured relatively often by environmental studies, e.g. in wastewater, surface runoff or snow. Cadmium, lead, zinc or copper concentrations is found around 10 times more often recorded than PAH concentrations. PAH concentrations have been only measured around 15 times by reliable sources (“ATV DVWK Datenpool 2001”, see (Brombach and Fuchs, 2003)) Compared to other persistent substances, there is also relatively much known about the behaviour of heavy metals, e.g. the percentage adsorbed to particles has been investigated in various studies, in snow (Glenn and Sansalone, 2002;Reinosdotter, 2003;Viklander, 1999) snow melt runoff (Westerlund *et al*., 2003), in surface runoff and in the sewer (e.g. by Chebbo and Gromaire (2004)), or in the wastewater treatment plant (Karvelas *et al.*, 2003).

Receiving Water Indicator

Copper has been found to be present in similar concentrations in surface runoff and in wastewater. Therefore it was chosen as an indicator for the behaviour of persistent substances. The indicator defined to assess the accumulation of persistent substances was the copper load discharged per year.

Copper load discharged per year: $Cu_{load} = \frac{\sum Cu_{load\,i}}{n}$ [kg/a] [15]

n ... number of years of the simulation

5.2 CSO PERFORMANCE INDICATORS

In the following emission-based CSO performance indicators are presented which are taken from different combined sewer overflow guidelines. These indicators are used to assess the performance of combined sewer overflows.

5.2.1 Number of Overflows per Year

The number of CSO overflows per year (NO) is an often applied indicator, e.g. in Belgium (Flanders) or the Netherlands (Zabel *et al.*, 2001). The computed number of overflows depends largely on how an overflow event is determined. It can either be defined by breaks in the rain or by breaks in the discharge. Depending on how and how long the break is defined, completely different numbers of overflow events are obtained. Here two CSO events are seen as distinct if there is a break of one hour between the discharges.

5.2.2 Mean Annual Overflow Volume

The mean annual overflow volume (VQO) is calculated as the total overflow volume divided by the number of years of the simulation.

Mean annual overflow volume $VQO = \frac{\sum VQO_i}{n}$ [mm/a] [16]

n ... number of years of the simulation

5.2.3 Maximum Overflow Event Once per Year

The maximum overflow event once per year (Q_{max}) was chosen as indicator for hydraulic impacts of CSO discharges. This indicator is used in the new ÖWAV R19 (Draft 2003), see also receiving water indicator for hydraulic impact (chapter 5.1.2). It was calculated with the Weibull plotting formula.

Return period $T = \frac{i}{k+1}$ [17]

i ... rank of the values arranged in descending order
k ... length of simulation [years]

The maximum overflow event once per year is the discharge during one simulation step (in the simulations described in the following chapters always 5min) [m^3/s].

5.2.4 CSO Efficiency

The CSO efficiency (η) is used as indicator in ÖWAV R19 (Draft 2003), and the acceptable overflow rate (equals 1/ η) in ATV A 128 (1992). The CSO efficiency has been calculated according to the following formula.

CSO efficiency $\eta = \left[1 - \frac{VQO}{VQR}\right] \cdot 100$ [18]

VQO ... mean annual overflow volume
VQR ... rain runoff

5.3 CORRELATION ANALYSIS OF INDICATORS FOR THE CSO PERFORMANCE WITH RECEIVING WATER INDICATORS

Combined sewer overflows have often been designed according to emission-based guidelines. Only recently also the effect on the receiving water became important. The different emission and ambient water quality based approaches are discussed in **paper V**. Emission-based CSO performance indicators are widely applied, but for example Rauch and Harremoës (1998) showed that there is little correlation between CSO volume reduction and resulting oxygen concentration in the river. **Paper VI** investigated if there is a correlation between impacts on rivers and CSO performance indicators. For this purpose numerical simulations were made with the software CityDrain (Achleitner, 2006;Achleitner *et al.*, in press) and the results evaluated the receiving water indicators and the CSO performance indicators described above.

The results show that the mean annual overflow volume (VQO) is ther best of the CSO performance indicators to describe the receiving water indicators, second best is the CSO performance (η). But none of the CSO performance indicators, also not mean annual overflow volume or the CSO performance, showed any correlation with the indicators for acute impacts, i.e. critical oxygen deficit and un-ionised ammonia concentration. These findings suggest that regulations based on CSO performance indicators as number of overflows should be at least critically reviewed and if necessary revised.

6 EVALUATION OF URBAN DRAINAGE MEASURES

The following chapter describes examples of the application of the indicators described in chapter 5. The first study, presented in **paper VII**, evaluated the cost-effectiveness of in-stream measures to reduce the hydraulic impact from combined sewer overflows. The study presented in **paper VIII** aimed at assessing the environmental impact from combined sewer systems compared to separate sewer systems. The studies are based on computer simulations with software representing the integrated urban drainage system.

6.1 MITIGATION MEASURES TOWARDS MORPHOLOGICAL ALTERATIONS OF RIVERS

The integrated system comprises not only of the technical system like the sewer or the wastewater treatment plant, but also of the river. Therefore improvement can also be achieved by measures in the river. In **paper VII** a study was performed to assess the cost-effectiveness of in-stream morphological measures. The simulations were done with the program REBEKA (Rauch *et al.*, 2000;Rauch *et al.*, 2002). The in-stream measures were simulated by changes of the parameters which influence erosion in REBEKA: the slope and the width of the river, and the grain size of the sediment. These measures can be “translated” to restoration measures: a decrease of slope could be realised by an increase of meandering which means an increase of the river’s length in a certain stretch. Widening of the river bed is also a common restoration measure although in reality it results similar as an increase of meandering. The increase of grain size is more critical as the introduced sediment is likely to be transported downstream at the next higher flood event, therefore the stability of this measure is uncertain. The implementation of the changes to reduce the erosion frequency could of course also be made by technical measures, as weirs or paving of the river bed, but in the light of the WFD this seems counterproductive.

As indicator for hydraulic impacts, the erosion frequency was applied (see chapter 5.1.2). The effects of the in-stream measures were compared to the effect of an increase of CSO basin volume, a common technical measure which yet proved to be ineffective for reducing hydraulic impacts. The results of the study showed that restoration measures might be a cost-effective solution to mitigate hydraulic impacts from the urban drainage system which additionally would improve the morphological status of the watercourse. The pollutants in the wastewater however could cause problems if the flow in the river is changed by restoration measures, e.g. it is possible that increased settling of sediments or decreased reaeration rate creates oxygen deficit problems.

6.2 COMPARISON OF THE PERFORMANCE OF COMBINED AND SEPARATE SEWER SYSTEMS

Paper VIII shows the results of a study in which the receiving water indicators were applied to compare the performance of combined and separate sewer systems. The comparison was made with computer simulations with the program City Drain (Achleitner, 2006;Achleitner *et al.*, in press). Combined sewer systems, where sewage and stormwater are conducted in the same pipe, are the traditional type of sewer systems. This is due to the development of urban drainage, where initially the main goal was to improve the hygienic situation by removing waste and stormwater from the urban development. At high storm events the hydraulic capacity of a combined sewer system can be exceeded and it is necessary to discharge part of the wastewater directly (via combined sewer overflows) instead of transporting it to the wastewater treatment plant. Due the problems of the combined system, in many countries separate sewer systems are preferred today. Often even combined systems are changed to separate systems. It is assumed that the stormwater is relatively clean and can be discharged without risk to surface waters. But looking at the concentrations measured in surface runoff and the storm sewer of separate systems, it has to be expected that a considerable amount of pollutants is discharged with the stormwater from separate systems.

The study aimed at comparing the pollution discharged due to stormwater from the two systems. Therefore catchments of the same size either drained by a combined or a separate system were simulated and the impacts compared. The impacts on the receiving water were assessed using the receiving water indicators described in chapter 5.1. The influence of different boundary conditions was considered: the pollution of stormwater and sewage, the rain type and the population density. Beside the ecological performance, also the costs of the systems were included.

It was found that generally from separate systems larger amounts of copper (used here as an indicator for heavy metals which are contained in significant amounts in surface runoff) were discharged into the receiving water, whereat combined systems generally discharged higher BOD loads, and caused higher un-ionised ammonia concentrations and larger oxygen deficits. The magnitude of the impact on the receiving water depended largely on the rain characteristics. Both sewer system types had a similar performance if the pollutant concentration in both wastewater and stormwater was low. If no stormwater treatment is applied, also separate sewer systems discharge considerable pollutant loads.

7 CONCLUSIONS

The results of the measurement campaign presented in **paper I** found considerable amounts of heavy metals, chloride and suspended solids in the roadside snow. At the low traffic site the heavy metal concentrations were in the same range as at the high traffic and the highway site. Copper was an exception, here the highest concentration was measured at the high traffic sites, likely because brakes are a source for copper (BUWAL, 1996). Cadmium concentrations on the other hand were at the urban reference site in the same range as the mean concentrations of the roadside snow samples, showing that beside traffic there are probably other important sources for cadmium. This agrees with the results of a study by Dallinger *et al.* (1998). The measurement campaign also confirmed the findings of other authors (Dierkes and Geiger, 1999; Reinosdotter *et al*., 2005) that pollution decreases rapidly with distance from the street. High chloride and suspended solid concentrations were found at the urban traffic sites and even higher chloride concentrations at the highway. This corresponds with findings that during the melt period the concentrations of suspended solids are significantly higher than in stormwater (Westerlund and Viklander, 2006; Westerlund *et al*., 2003). The concentrations of the heavy metals measured in the case study in Innsbruck are similar to what has been found by other authors (Reinosdotter, 2003; Viklander, 1998), only Glenn and Sansalone (2002) report significantly higher concentrations for highways. Compared to the concentrations in runoff the mean concentrations measured in the snow roadside samples were mostly at the upper end or above the range reported in literature (Brombach *et al*., 2005; Welker and Dittmer, 2005) which confirms that pollution accumulates in the snow. It was not possible to identify trends in the accumulation of the pollutants in the snow but also other author found that, due to the numerous influencing factors, it is very difficult to predict pollutant concentrations of roadside snow (Viklander, 1998). Based on the concentrations measured for the urban traffic sites, an estimation of the pollutant loads, which are dumped due to the practice of snow disposal into the river Inn, was made. The estimation showed that with this practice, an impact on the river, especially in the case of smaller rivers, cannot be excluded. There are however large uncertainties in this estimation which would need to be reduced by further measurements in both snow and river to optimise future snow management.

The measurement campaign of different Tyrolean infiltration swales, presented in **paper II**, investigated the pollutant contents of the devices' soil, the pH and parameters of the hydrological functionality, as grain size and hydraulic permeability. The investigated pollutants were the heavy metals zinc, copper, lead and cadmium, and the hydrocarbon index. Unfortunately, the references taken from nearby soils proved not to be reliable, because repeatedly by subtracting the pollutant contents of the reference site from the measured mean pollutant load of the swale' soil, negative pollutant contents were obtained. The heavy metal contents were compared with the age of the devices, their hydraulic permeabilities and their pH, but for none of the three parameters a correlation could be found with any of the four heavy metals. The soil samples were taken in three layers of different depth, because it was expected that the highest concentration would be found in the top five centimetres. The expected depth profile of the pollutants could not be found, although such profiles have been reported by various authors (Barraud *et al.*, 1999; Dierkes and Geiger, 1999; Mikkelsen *et al.*, 1996; Mikkelsen *et al.*, 1997). It is possible that such profiles did not exist in the investigated swales but it is also possible that, when removing the vegetation from the soil sample, the highest polluted layer was accidentally removed, too. Based on data found in literature, an estimation of the pollution introduced to the infiltration swales was made. It showed that the load of heavy metals that is to be expected to accumulate during 15 years lifetime due to surface runoff infiltration would exceed the standards of the landfill regulation only at very few sites, and only if highly polluted surface runoff would be discharged. The results of the study show no significant risk by infiltration of parking place runoff for soil and groundwater. Also Mikkelsen *et al.* (1996; 1997) concluded that neither surface nor sub-surface infiltration poses significant risk for groundwater. For final conclusions, a long-term study starting with the newly constructed infiltration device and also including the concentrations in the runoff would be desirable.

The aim of the case study at the river Drau (presented in **paper III**) was to identify the relevant physicochemical parameters to describe the water quality status of an alpine river. Such a reduction of parameters to the most important ones has been done by various authors (House *et al.*, 1993; ATV, 1993; Schilling *et al.*, 1997; Borchardt and Sperling, 1997; BWK, 2001; Rossi *et al.*, 2004). Based on the results of the case study, the importance of the parameters describing impacts on the surface water status of alpine rivers was judged. It was concluded that hydraulic disturbance and morphological deficiencies are very important, while toxic effects and elevated chloride concentrations can be of importance. On the other hand, nutrients and dissolved oxygen depletion are not relevant parameters because discharges from facilities designed on emission-based criteria usually are sufficient to protect alpine running water. When applying such a parameter reduction, it has always to be considered that, although the conclusion may generally be true, there can be exceptions (Podraza, 1999). The findings of the case study correspond with the results of other work done during the Austrian implementation of WFD which found that high percentages of the Austrian water bodies have hydromorphological problems (Muhar, 2000; BMLFUW, 2005). Nevertheless, this picture could be different depending on the indicators developed during the Water Framework Directive’s implementation. If type-specific biological indicators would be developed, these could be able to reflect changes in the nutrient status which cannot be assessed with the current saprobic index.

The assessment of urban drainage measures with regard to the WFD can best be done by computer simulations. The current models are able to represent all parts of the urban drainage system: catchment, wastewater treatment plant and receiving water. The testing of different measures with the computer allows the analysis of the system behaviour (Butler and Davies, 2004). Several models for rivers have been developed to be applied in simulations of the urban drainage system, starting from simple oxygen consumption (Harremoës, 1982) to complex models which can represent additionally also processes as nitrification or photosynthesis (Rauch *et al.*, 1998). These models represent important processes which convert pollutants, but not different aquatic species. There are some models for species composition and abundance, e.g. RIVPACS (Clarke *et al*., 2003), PAEQANN (Lek *et al*., 2003), or a German study by Schleiter *et al* (1999), yet there applicability is strictly restricted to the regions they have been developed for. The dynamics and relations of aquatic ecosystems are not yet fully understood (Wetzel, 2001), therefore it is currently not possible to develop general models for river biocoenosis.

Due to the current lack of appropriate models, the evaluation and comparison of measures in different parts of the urban drainage system on a European scale can only be made by using ambient water quality standards as representation for ecological quality. Therefore a literature review collected various standards for both intermittent and long-term impacts (see **papers IV** and **V**). It was found that the limits for long-term impacts given in regulations show great differences. Especially the limits for heavy metals show high discrepancies because of the various applied dependencies of standards on water hardness. The large variation of water quality standards have also been shown by Ragas *et al*. (2005). During the implementation of the WFD some new environmental quality standards will be developed. On the one hand there will be limits set for the priority substances (Europe-wide) and other relevant substances (member state specific), for Austria see Wimmer *et al*. (2003). The problem of water quality standards is that they often do not consider pollutant interactions (Ellis, 2000) and indirect effects (Preston, 2002). In the frame of the WFD it is not sufficient to protect the watercourse from serious toxic impacts, but additionally its natural character has to be preserved. Therefore for each region, type-specific nutrient criteria are needed. If there will be limits for the general chemical and physico-chemical elements specified in the WFD is not yet become apparent. In Austria however quality standards for BOD, DOC, nitrate, phosphate and phosphorus have been proposed for each ecoregion (Deutsch and Kreuzinger, 2005).

Paper V presents a critical review of different approaches to assess intermittent impacts. There are different types of approaches, for example emission-based design guidelines or ambient water quality limits based approaches. Conventionally stormwater treatment facilities are designed based on emission limits. These limits have advanced from simple sewage dilution limits to percentage wastewater treated (e.g. CSO performance of ÖWAV R19 (Draft, 2003)). Zabel *et al*. (2001) and Fenz (2002) reviewed the current emission-based CSO guidelines in Europe. Various guidelines have been developed in the last decades based on ambient water quality limits (e.g. the Danish approach (Danish Engineering Union Wastewater Committee, 1985), UPM (FWR, 1998), BWK M3 (BWK, 2001), or STORM (Rossi *et al*., 2004a)) and are still being developed (e.g. ÖWAV R19 (Draft 2003)). These guidelines contain limits for the most important impacts and often also include the synergistic effects of high un-ionised ammonia and low oxygen concentrations. The problem of these standards is that they are complex and therefore relatively difficult to apply. Further they are limited to a restricted number of pollutants because it is not feasible to measure all possibly present substances. Different ambient water quality standards have been reviewed by the project STORM (Rossi *et al*., 2004a). Ellis (2000) reviewed various approaches (chemical limits, biological assessment, direct toxicity assessment and biomarker techniques) for the risk assessment of intermittent pollution events, concluding that each approach has its limitations. Neither the benefits of emission nor of ambient water quality based approaches for the reduction of intermittent impacts on receiving waters are scientifically proofed. Due to the WFD they will probably used in combination in the future. However, to provide cost-efficiency in urban drainage, the validation of the approaches has to be accomplished.

For the assessment of urban drainage measures receiving water indicators have been defined (applied in **paper VI, VII** and **VIII)**. These receiving water indicators are based on the main impacts caused by wastewater discharges, as identified by various workers (House *et al.*, 1993; ATV, 1993; Schilling *et al.*, 1997; BWK, 2001; Rossi *et al.*, 2004). Some of these indicators can be evaluated using ambient water quality standards for acute impacts. For example the critical oxygen deficit, the un-ionised ammonia or the discharge can be compared to the standards given in ambient water quality guidelines for intermittent impacts, e.g. Danish Engineering Union Wastewater Committee (1985), the UPM (FWR, 1998), BWK M3 (BWK, 2001), or Rossi *et al*. (2004).

In **paper VI** the correlation between the defined receiving water indicators and different CSO performance indicators was investigated. It was found that emission-based CSO performance indicators, like the number of overflows or the maximum discharge event, are not able to reflect the impact on receiving water quality. Only the total overflow volume, and to a lower extent also the CSO efficiency, have some correlation with the receiving water indicators erosion frequency, discharged copper load and discharged nitrogen load. For the indicators for acute impacts, none of the investigated CSO performance indicators was able to represent the impact. From the correlation analysis, it can be concluded that regulations based on the number of overflows should be revised. Further it was found that for assessment of acute impacts, simulations or measurements are inevitable. Also Rauch and Harremoës (1998) found that total CSO volume is not a good indicator for oxygen status of the river, and Lau *et al*. (2002) showed that the applicability of CSO spill frequency / volume as indicator for oxygen and ammonia is restricted.

Paper VII aimed at the assessment of the cost-effectiveness of in-stream measures for mitigation of hydraulic impacts. It could be demonstrated that in-stream measures, even restoration measures, have potential to reduce the frequency of erosion events. The problem of proposing restoration measures in urban environment is that they are usually not applicable, because in urbanised areas there is usually not enough space to allow the river to follow a natural course. The increase of CSO basin volume, a conventional measure if there are problems due to combined sewer overflows, was found to be an inefficient measure for the reduction of hydraulic impacts. It was also found that in-stream measures are potentially even cheaper than an increase of CSO basin volume. Still, there are large uncertainties regarding the length of the stretch which is impacted by erosion and therefore also regarding the costs (as the costs of the in-stream measure depend on the length of the stretch). A following study showed that it is difficult to predict the length of the impacted stretch, especially if there are several discharge structures whose waves interfere with each other, and that the predicted length varies widely depending on the sediment transport equation applied in the simulation (Lek *et al*., 2006). In **paper VII** the in-stream measures were only investigated regarding hydraulic impacts. Toxic effects by un-ionised ammonia or oxygen depletion were not included. Nevertheless, restoration of the river enhances its recolonisation potential and thus also its resilience to toxic impacts. On the other hand it has never been really investigated if restoration measures could deteriorate the situation in a polluted river. Work done by Frey (2001) indicates that there could be problems with oxygen due to the decreased flow velocity and the increased deposition of material in restored stretches of polluted rivers.

Paper VIII describes a comparison of the performance of combined and separate sewer systems using ecological and economical indicators. In many countries separate sewer systems are preferred today. In the study however it could be shown that separate sewer systems without stormwater treatment introduce quite significant amounts of pollutants, especially heavy metals, into the receiving water. Also Boller (1997) stated that with separate systems heavy metals are discharged into the receiving water, which in combined systems are retained in the wastewater treatment plant sludge. It was shown in **paper VIII** that also separate sewer systems can induce hydraulic impacts in the receiving water. Further it was demonstrated that the magnitude of the impact on the receiving water depends on the rain characteristics. The composition and amount of pollution from the storm sewer of a separate system depends on the type of surfaces connected to it, e.g. percentage of metallic installations like copper roofs, the traffic density, or industrial areas (ATV-DVWK-M 153, 2000), respectively the pollution of the stormwater varies over large ranges (Brombach *et al*., 2005). In **paper VIII** it was shown that the pollution concentration of wastewater and stormwater influences the differences in the performance of the two systems, e. g. at low pollutant concentrations the two systems have a similar performance. Generally from the separate sewer system smaller loads of organic substances but without stormwater treatment high amounts heavy metals are discharged. This has also been found by Brombach *et al*. (2004).

The assessment of urban drainage measures with regard to the Water Framework Directive on the basis of computer simulations requires appropriate indicators to assess the impact on the receiving water. Currently the good ecological status is not defined in a way that is applicable in urban drainage modelling, therefore ambient water quality standards should be applied. Different standards have been reviewed, for both long-term and intermittent impacts. The general comparison of measures in different scenarios proofed difficult. Thus instead of limit exceedances, loads and concentrations were defined as receiving water indicators. The application of the receiving water indicators in **paper VII** and **VIII** shows that these indicators can identify the type of impact on the receiving water which is influenced by urban drainage measures. Although the indicators do not describe the exact situation in the receiving water, they can yet help to identify interesting options. This can narrow the selection of options in a specific planning procedure, although before implementation it is still necessary to evaluate the final options case-specific, e.g. by simulations of the integrated urban wastewater system including at least a simple model of the specific receiving water.

The aim of Water Framework Directive is a good ecological status of the European water bodies. Urban drainage is only one of the impacts on the European waters. Especially in Central Europe the main pressures on the rivers derive from morphological degradation. In future due to the European Water Framework Directive ambient water quality approaches and the prioirity substances have to be considered in urban drainage. However, it has not been scientifically sound proofed that the ambient water quality limits existing today in Europe are suitable to describe impacts on the ecological quality. For priority substances currently new limits are being specified. For urban drainage however also adequate limits for general parameters (as nutrients or BOD) would be required, both long-term standards and for acute impacts. These standards are needed for planning purposes to evaluate the results of simulations of the integrated urban drainage system. However, such limits are not required in the Water Framework Directive and thus it is uncertain whether they will be defined. The biological indicators to measure the ecological quality according to the Water Framework Directive are currently still under development. Thus it is unclear if they will be applicable for urban drainage, where biological indicators would be needed which are able to identify the causes of impacts. Presently the implementation of the Water Framework Directive is still in progress. Neither standards nor tools for its application in water management have yet been developed. Therefore it is currently difficult in urban drainage to take the Water Framework Directive into account. The consequences of the Water Framework Directive for urban drainage can currently not be evaluated with certainty. However, this will probably become clear in the further implementation process.

8 OUTLOOK

The Water Framework Directive demands that all European water bodies achieve at least a good ecological status. Yet the definition of the good ecological status is still ongoing. Currently it is defined with biological elements, but today it is not possible to model lotic ecosystems, although there are approaches. It is also not realistic that there will be easily applicable models in the next future because lotic ecosystems are very complex and the relationships between different species are only insufficiently understood. Also the impact of most pollutants and especially the impact of mixtures of pollutants on different species and the effect on the biocoenosis resulting from the impact on a species are not known. Therefore in the near future it will be inevitable to use ambient water quality limits in urban drainage, although the definition of good ecological quality is not directly translatable to physicochemical water quality parameters. Aquatic ecosystems are highly complex and there are many different factors which influence the aquatic biocoenosis so that most probably in the near future the cause-effect relations abiotic and biotic elements of the aquatic ecosystems will not be understood. Due to the implementation of the Water Framework Directive numerous research projects are ongoing. Therfore the understanding of aquatic ecosystems will increase. However, enormous research efforts are still required to reduce the uncertainties connected with the application of ambient water quality approaches in urban drainage. In this dissertation various ambient water quality regulations and guidelines have been presented. Based on these, receiving water indicators were chosen to assess the effect of urban drainage measures on running waters. During the further implementation of the Water Framework Directive, the good ecological status will be defined. On this basis the ambient water quality based approaches should be verified and improved.

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ANNEX I

ENVIRONMENTAL IMPACTS OF URBAN SNOW MANAGEMENT - THE ALPINE CASE STUDY INNSBRUCK

Engelhard C., Toffol S. D., Lek I., Rauch W. and Dallinger R. (2007). Environmental impacts of urban snow management - The alpine case study of Innsbruck. *The Science of the Total Environment*, **382**, pp. 286–294.

ANNEX II

LOCAL INFILTRATION DEVICES AT PARKING SITES - EXPERIMENTAL ASSESSMENT OF TEMPORAL CHANGES IN HYDRAULIC AND CONTAMINANT REMOVAL CAPACITY

Achleitner S., Engelhard C., Stegner U. and Rauch W. (2007). Local infiltration devices at parking sites - experimental assessment of temporal changes in hydraulic and contaminant removal capacity. *Water Science & Technology*, **55** (4), pp. 193-200.

ANNEX III

CHALLENGES IN THE IMPLEMENTATION OF THE WATER FRAMEWORK DIRECTIVE: CASE STUDY OF THE ALPINE RIVER DRAU, AUSTRIA

De Toffol S., Achleitner S., Engelhard C. and Rauch W. (2005). Challenges in the implementation of the Water Framework Directive: case study of the alpine River Drau, Austria. *Water Science & Technology*, **52** (9), pp. 243-250.

ANNEX IV

RISK ANALYSIS AND IMPACT ASSESSMENT OF URBAN STORMWATER- WITH EMPHASIS ON THE EU - WFD

Engelhard C. and Rauch W. (2007). Risk analysis and impact assessment of urban stormwater- with emphasis on the EU - WFD. **In**: D. R. Thévenot (Ed.). DayWater: an Adaptive Decision Support System for Urban Stormwater Management, IWA publishing, London. ISBN: 1843391600.

ANNEX V

INTERMITTENT IMPACTS TO RECEIVING WATER BODIES-REVIEW ON APPROACHES, INDICATORS AND LIMITS

Engelhard C. and Rauch W. (submitted) Intermittent impacts to receiving water bodies-review on approaches, indicators and limits. *Water Research*.

Intermittent impacts to receiving water bodies

Review on approaches, indicators and limits

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ABSTRACT

Pollution from urban stormwater discharges can contribute considerable pollution to the watercourses. In this paper a literature review is presented which gives an overview on the past and current methods to prevent impacts from urban wet-weather flows. The potential of the different approaches to enhance the ecological quality of the receiving waters is critically evaluated. The results of the review show that the aim of water management is nowadays generally to secure good ecological quality of watercourses. The conventional approach to design wet-weather controls on emission based guidelines is well established and easy to apply, but cannot guarantee the ecological quality of the receiving water. Toxicity testing methods detect the integrated toxic effects of complex mixtures of substances, but the methods are still premature and currently not applicable for urban drainage planning. Ambient water quality approaches based on physicochemical limits are significantly more complex to apply than emission based approaches, but they are able to take the water quality into account in the planning process. However, the relations of water quality and ecological quality are still insufficiently understood. Currently there are no biological measurement methods available which provide indicators to assess intermittent impacts. Also only very few case studies have been performed to validate the ambient water quality based approaches. Altogether, presently a combined approach seems to be best feasible for control of intermittent impacts from urban wet-weather flows, whereat controls are designed according to well established emission guidelines, and ambient water quality approaches are applied only if problems with the quality of the receiving water are detected.

Key words: combined sewer overflow; emission standards; ambient water quality

INTRODUCTION

The reduction of impacts on the receiving water quality from urban areas at wet-weather conditions is a demanding task in urban drainage. Intermittent impacts can be caused by stormsewer outfalls but especially by combined sewer overflows (CSO). Combined sewer overflow discharges consists of a mixture of stormwater and wastewater, and particularly the organic pollutants and the ammonia contained in the wastewater are responsible for intermittent impacts on the receiving water quality. In the middle of the 20th century wet-weather discharges were increasingly recognised as source for receiving water pollution (e.g. Bode and Weyand, 2002; Borchardt and Statzner, 1990; Burian et al., 1999; Butler and Davies, 2004). Accordingly the management of wet-weather flows became an important topic in research and new concepts for stormwater treatment were developed (Burian et al., 1999).

The methods for the design of stormwater treatment facilities have been continuously improved. From the primary dilution methods, engineers created more sophisticated design rules as for example the critical rain (Bode and Weyand, 2002) or the ‘formula A’ in the UK (Butler and Davies, 2004). Computer programs were developed but focusing primarily only on the discharge in the sewer. Since the 1980s additionally water quality models for rivers were developed by cooperation between engineers and natural scientists (Rauch et al., 1998). First the research mainly focused on acute effects of oxygen depletion (Harremoës, 1982; Hvitved-Jacobsen, 1982), resulting in the definition of the Danish approach (Danish Engineering Union Wastewater Committee, 1985). Today the ecological risks from combined sewer overflows are recognised widely (Ellis, 2000) and various guidelines for wet-weather pollution from urban drainage based on ambient water quality have been developed, e.g. the UK Urban Pollution Management (UPM) (FWR, 1998), the Swiss project STORM (Krejci et al., 2004b; Rossi et al., 2004a) or the German guideline BWK M3 (BWK, 2001).

The aim of this paper is to give an overview of the approaches to protect receiving water quality at wet-weather conditions. For intermittent impacts a large variety of approaches, which differ widely in the indicators and underlying principles, exist today. Therefore it is difficult to understand the advantages and disadvantages of the numerous approaches. It is the aim of this paper to close this gap and give a critical overview of them. Special attention is given to their applicability in wet-weather control planning and to their potential to protect ecological quality. Therefore first the different types of impacts caused by intermittent discharges on receiving waters are presented. The pollutants contained in wet-weather discharges cause due to their different nature impacts on various time and spatial scale. Then the general typology of the methods to define limits for intermittent impacts is explained. One major difference between the approaches is that ambient water quality approaches consider the status of the receiving water, whereas emission limits do not. Following various emission based design guidelines for combined sewer overflows and CSO basins are shown. Nowadays the protection of the ecological quality of the waters is an important task. Therefore the current methods to measure ecological quality in terms of biological water quality indicators are explained. The limitations of the actual biological indicators are shown. Especially the impossibility to link the biological indicators to physicochemical water quality parameters is a problem for their application for wet-weather flow management. To protect the receiving water quality, various ambient water quality guidelines have been developed, which evaluate the impact from intermittent discharges, especially combined sewer overflows, on the basis of limits for physical and chemical parameters. Different approaches and the limits used therein are presented. As it is impossible to monitor wet-weather discharges with reasonable effort, these approaches are mainly applied to evaluate the results of computer simulations of the integrated urban drainage system (integrated means including catchment, wastewater treatment plant and receiving water) (Rauch et al., 2005). Some authors measured the effect of wet-weather discharges on the basis of changes of the aquatic biocoenosis, and case studies are presented here. Beside measurement of biological indicators and physicochemical parameters, toxicity tests are used to assess the impact of wet-weather discharges on the receiving water quality. The advantages and disadvantages of this approach are discussed.

The comparison of the different approaches to regulate intermittent impacts from wet-weather pollution shows that each has its advantages and its drawbacks. Emission limits are relatively easy to apply and there is little uncertainty about the costs (Krejci and Kreikenbaum, 2004). However there is little relation between an emission limit value and the concentration of the respective chemical in the receiving water (Whitehouse, 2001) and even less with the ecological quality. Ambient water quality based approaches consider the varying characteristics of the receiving water. The wet-weather standards used in these approaches are still associated with considerable uncertainties and therefore need to be developed further (Ellis, 2000). Additionally, there are several problems associated with ambient water quality standards, e.g. they are difficult to apply, usually not site-specific (because defined on relatively large spatial scales), and there are difficulties due to the complexity of the cause effect relationships to define targets (Lijklema, 1995). Toxicity testing for wet-weather discharges is still on pilot study level and currently not applicable for wet-weather flow management. The biological measurement methods are being improved, especially in Europe this is an important topic nowadays (e.g. Hering et al. (2004)). Still, currently no biological measurement methods are available which can assess impairment from intermittent discharges, and even if impairment is detected, it is very difficult to determine its cause.

GENERAL TYPOLOGY OF INTERMITTENT IMPACTS

Although wet-weather discharges usually have a short duration, they cause several ecologically relevant impacts on the receiving water in dependence of the pollutants contained in the discharge. Stormwater outfall discharges contain pollutants of the surface from which the runoff originates. These are in urban areas especially traffic related pollutants (e.g. heavy metals or polycyclic aromatic hydrocarbons) and in smaller amounts also organic pollution. Combined sewer overflow discharges contain beside the stormwater pollution also wastewater. Therefore CSO discharges have significantly higher concentrations of ammonia and organic pollutants (for concentrations see for example Brombach et al. (2005)). The timescale of the impacts of pollutants from urban discharges range from acute over delayed to accumulating effects (House et al., 1993).These effects can be further classified as hydraulic, chemical, physical, and bio-chemical impacts (Borchardt and Sperling, 1997; Novotny and Witte, 1997; Schilling et al., 1997), see for example Table 1.

Table 1: Impacts of urban discharges on receiving waters (Schilling *et al.*, 1997).

time scale	**characterization**	**indicator variable**
acute	hydraulic	flow, shear stress, bed erosion
(hours)	chemical	toxic substances (NH_3)
	physical	suspended solids
	bio-chemical	oxygen depletion in the water body
	hygienic	bacteria, virus
	aesthetic	floating material, odour
delayed	hydraulic	sediment carrying capacity
(days)	chemical	toxic substances (NH_3, NO_2)
	bio-chemical	oxygen depletion in the sediments
	hygienic	bacteria, virus
	aesthetic	floatables, debris, oil
accumulating	hydrologic	flow regime, morphology
(weeks, years)	chemical	heavy metals
		persistent organic substances
		inorganic and organic sediments
	bio-chemical	oxygen depletion (eutrophication)

Beside the time scale also the spatial scale is important (House et al., 1993). Acute toxic effects, as they can be caused by un-ionised ammonia, occur only local and for short time. Accumulation of persistent pollutants (numerous such substances can be contained in both sewage and stormwater) can occur during decades and can impact a whole river system (see Figure 1).

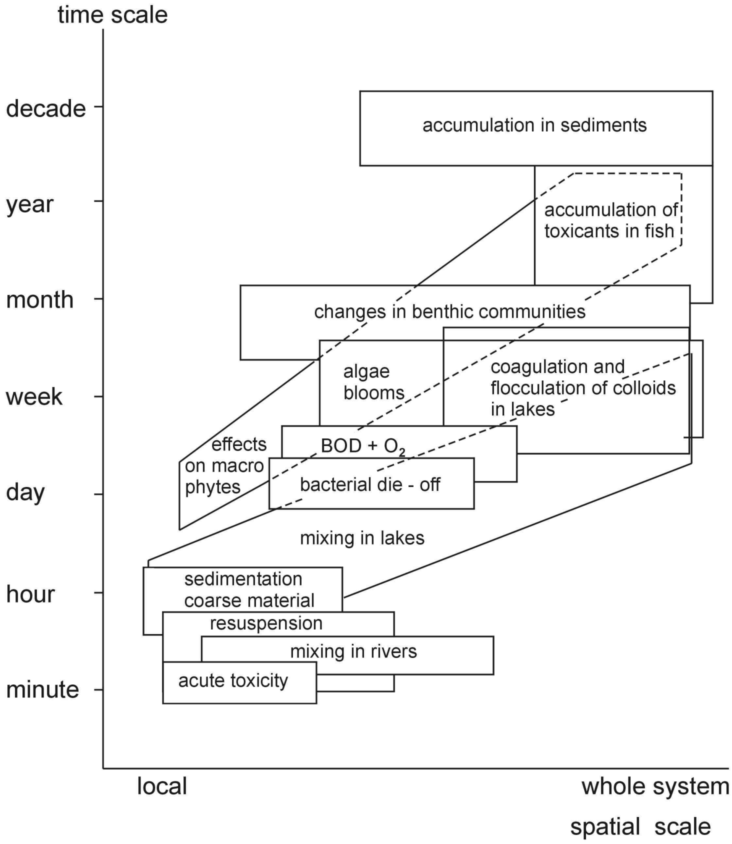

Figure 1: Time and spatial scales for receiving water impacts (redrawn from Lijklema *et al*. (1989))

Most countries have significant percentages of combined sewer systems (see for example Table 3) and respectively numerous combined sewer overflow structures. Monitoring of combined sewer overflow impacts is complicated and expensive. Therefore for cost-efficient reduction of their environmental impacts, preliminary identification of problematic structures is advantageous. Wolf and Borchardt (1990) made a proposal for an approach to assess the risk potential of combined sewer overflows for the receiving watercourse on the basis of an "ecological tolerable pollution load per inhabitant" in dependence of flow velocity, water depth and toxic un-ionised ammonia concentration. This proposal has led to a risk estimation approach included in the advanced requirements for CSO discharges of the German association for water, wastewater and waste ((ATV, 1993, 1997), English description can be found in Borchardt and Sperling (1997)). With this approach it can be estimated whether there is a risk of ecological damage for a receiving water due to chemical impacts (risk factor defined as inhabitant per base flow of the receiving water), or hydraulic impacts (risk factor defined as percentage impervious area to total catchment area).

The most common approach to predefine the risk potential of urban discharges is the qualitative description of risk in relation to the type of the receiving water. Based on typical characteristics (as flow velocity or size), the risk potential of important pollutants for the specific type of receiving water is stated. The receiving water types used in these schemes differ in fundamental parameters which determine their reaction to urban wet-weather discharges. Various authors developed such schemes (for example House et al. (1993), ATV (1993), Schilling et al. (1997), in the German BWK M3 (BWK, 2001) and in the project STORM (Rossi et al., 2004a), but the different schemes are generally consistent with each other. An example for such a classification scheme can be seen in Table 2.

Table 2: Overview on selected watercourse protection problems related to sewer system discharges at wet-weather conditions, their relevance in Switzerland and proposed emission and ambient water quality requirements in the frame of the project STORM (Rossi *et al.*, 2004a).

Type	Aesth-etics	Hygiene (mikroorg.)	Temp-erature	Mech.-hydraul. stress	NH_3/ NH_4-N	TSS sedim.	TSS turbidity	Nutrients
Spring region	+ / E	o / I	o / I	+ / I	+ / I	+ / I	+ / I	x
Small midland creek	+ / E	o / I	o / I	+ / I	+ / I	+ / I	+ / I	x
Small pre-alpine creek	+ / E	o / I	x / I	+ / I	+ / I	o / I	+ / I	x
Large midland creek	+ / E	+ / I	x	o / I	o / I	+ / I	+ / I	x
Large pre-alpine creek	+ / E	+ / I	x	o / I	o / I	x	+ / I	x
Larger river	+ / E	+ / I	x	x	x / I	x	x	o / I
Small lake	+ / E	+ / I	x	x	x	+ / I	x	+ / I
Large lake	+ / E	+ / I	x	x	x	+ / I	x	+ / I

Frequency of occurrence: frequent (+), occasionally (o), not observed so far (x)
Relevance of problem: important (dark grey), possibly important (light grey), unimportant (white)
Type of requirement: emission (E), ambient water quality (I)

In the schemes, receiving waters are classified as stagnant and running waters. Running waters are on the one hand distinguished according to their size (e.g. river and creeks), on the other hand according to their slope (e.g. in lowland and mountain watercourses). With this classification, each group possesses several parameters which determine the magnitude of impacts. For example stagnant waters (e.g. lakes but also impounded streams) are sensitive to nutrients, as the pollutants accumulate and can easily lead to eutrophication. Large water bodies as streams or big lakes are generally not sensitive to acute effects because they have sufficient dilution capacity. Creeks however are due to their size sensitive to both acute toxic and hydraulic impacts. Creeks with low flow velocity (as lowland and midland creeks) are susceptible to oxygen depletion, because they have low sediment transport capacity and low reaeration rate.

Naturally, such classification schemes give only a general idea. Dependent on local conditions (as for example initial level of pollution), the schemes may not be appropriate. For example Podraza (1999) found that in the investigated receiving water there occurred sometimes oxygen deficits near the river bed, although the receiving water was a small mountainous river for which oxygen concentration and solids should have no or little relevance according to ATV (1993). Therefore predictions of impact relevance in a general way have always to be verified site-specific.

GENERAL TYPOLOGY OF LIMITS AND STANDARDS

Emission limits

Emission limits are often referred to as “end of pipe” limits to express that they are derived irrespective of the receiving water (Whitehouse, 2001). The underlying principle of emission limits is the ‘precautionary principle approach’. It is assumed that all pollution is potentially harmful and thus all that can be eliminated with the current technology should be removed. With the Best Available Technology (BAT) approach, standards are defined so that they can be achieved with the currently available technology. Usually also economic consideration play a role, which is expressed in the name of the approach: BATNEEC: Best Available Technology Not Entailing Excessive Costs or BPM: Best Practicable Means (Ragas et al., 2005). Emission limits derived by a BAT approach have to be updated regularly to include technological progress. It is important to recognise that no definite link exists between an emission limit value and the concentration of the controlled chemical in the receiving water (Whitehouse, 2001). Even less it is possible to find stringent relation between emission limits and a quantitative description of the ecological quality of a river system.

Environmental quality standards and approaches

Environmental quality standards are usually based on a certain environmental quality objective, like “the protection of human health and ecosystems” or “protection of the drinking water supply”. Based on the environmental quality objective, standards are defined to achieve the objective (Ragas et al., 2005).

Increasingly the environmental quality objectives include the protection of ecosystems. Biological assessment methods, which measure indicators like species composition and abundance, or trophic composition, are used to check the compliance with this objective. The first biological water quality measures were saprobic indices which described organic pollution. Today increasingly methods are used which are able to reflect various impacts and thus describe the biological integrity of a water.

Chemical specific limits are usually defined to achieve the environmental quality objective. Such limits are defined for very different types of substances, like nutrients, oxygen, acute toxic substances or persistent toxic substances. Precondition for the derivation of a chemical specific limit is that sufficient high quality toxicological data exists for the pollutant. Toxicological data is usually restricted to a limited number of test organism species. Therefore safety factors are applied for the extrapolation of the chemical specific limits from the toxicological data (Whitehouse, 2001).

Whole effluent toxicity based approaches have been developed to measure the toxic effect of a mixture of pollutants where the different components of the mixture are not known. Toxicity tests usually measure mortality. Permits based on whole effluent toxicity are widely used for industrial discharges (Power and Boumphrey, 2004), pilot research is also done for wastewater treatment plants and wet-weather discharges (Ellis, 2000).

EMISSION BASED APPROACHES

Approaches based on emission standards are the traditional way of engineers. Emission standards usually do not include relations to the characteristics of the respective receiving water, except for simplistic statements as special protection requirements for sensitive waters. The advantages of emission based guidelines are that they are easy to apply and require the same effort independent of the region (i.e. no competitive disadvantages due to the receiving water’s characteristics).

The legal situation for CSO design is relatively inconsistent, usually no national design guidelines for combined sewer overflow exists. An exception are the United States of America (US EPA, 2004) where CSO control is defined in national law. Sometimes national guidelines exists which are treated as legally binding requirements (e.g. Germany (ATV-A 128, 1992) or Austria (OEWAV R19, 1987)), but in most cases a variety of different guidelines is used (see Table 2). An overview on European CSO design rules is also given for example in Zabel et al. (2001) or Fenz (2002). A comparison of the CSO design rules in Canada and USA is for example given by Zukovs (2005). Also for separate sewer systems emission based guidelines exist, e.g. for reduction of hydraulic impacts, but these are not discussed here.

Combined sewer overflows are mostly designed as side-weirs, which divide the flow in two, one going to the wastewater treatment plant and one discharging to the watercourse. Rules for the design of overflows exist since the end of the 19th century, for example Sickert (1998) gives an historical overview. Until the middle of the 20th century these rules were typically based on the dilution method, where overflow is allowed only when there is a sufficient dilution of the dry weather flow with rain runoff (usually between 2 and 8 times the dry weather flow) (Bode and Weyand, 2002). In this approach the runoff is assumed to be not substantially polluted, however, measurements have shown that runoff can contain significant pollution (compare for example Brombach et al. (2005)). In the second half of the 20th century wet-weather flows were recognised as a major cause for receiving water quality degradation (Bode and Weyand, 2002; Burian et al., 1999; Butler and Davies, 2004) and it was searched for solutions.

Combined sewer overflows consist either only of a weir, over which the excessive water is discharged to the receiving water, or they can also include a detention basin. At CSO structures without basin, only the flow that is retained in the system for treatment can be prescribed. At CSO structures with basin additionally also the size of the basin can be specified. Pollution due to wet-weather conditions from combined sewer systems can originate either from combined sewer overflow discharges, but also from reduced wastewater treatment plant performance (Rauch and Harremoës, 1996). Therefore beside the CSO also the inflow to the wastewater treatment plant is defined in regulations (see Table 3, column QWWTP). This limit has to be considered when designing the last overflow before the wastewater treatment plant because otherwise the hydraulic capacity of the wastewater treatment plant could be exceeded.

At CSO structures the flow that is retained in the system can be prescribed using different methods, e.g. with the dilution method, the number of overflows or the critical rain (see Table 3, column Q_t(CSO)). In the dilution method, an overflow is allowed only if there is a sufficient dilution of the dry weather flow with rain (e.g. the European standard EN 752-4 proposes 5-8 times the dry weather flow (EN 752, 1998)). Another possibility to specify the retained flow is the limitation of the number of overflows per year, sometimes dependent on the receiving water (e.g. Belgium, Denmark or Netherlands) (Zabel et al., 2001). In the critical rain method, the retained flow is defined in direct relation to the runoff (for example in Germany 7.5-15 $l/(s \cdot ha_{impervious\ area})$ (ATV-A 128, 1992)).

For CSO structures including basins, the volume of the basin can be regulated (see Table 3, column 'CSO storage volume'). The CSO volume can be prescribed according to the size of the connected catchment (e.g. 10-40 $m^3/ha_{impervious\ area}$ in Germany (ATV-A 128, 1992)). The volume can also be defined over the detention time for the stormwater (e.g. 2 hours at 3 times dry weather flow in Great Britain (FWR, 1998)), or the return period of the rain that is to be intercepted (e.g. rain of 3-6 months return period in France (CERTU, 2003)).

Table 3: Overview on the requirements for CSO in different countries (De Toffol, 2006). Described are the percentage combined sewer system in the country (%CSS), throttle to the wastewater treatment plant (QWWTP), throttle at CSO (Qt(CSO)), prescribed CSO storage volume, percentage of stormwater to be treated at the wastewater treatment plant (% Treatment WWTP) and if effects on the receiving water are considered in the regulation.

Country	**Guideline**	**% CSS**	**Q_{wwtp}**	**Q_t (CSO)**	**CSO storage volume**	**% Treatment WWTP**	**Effects on RW considered**
Austria	OEWAV R19 (1987)	75-80	2 Q_{DWFp}	<15 $l/(s \cdot ha_{Ared})$	15-25 m^3/ha_{Aimp}	-	no
Austria new	OEWAV R19 (Draft 2003)		2 Q_{DWFp}			50% rain runoff	yes
Belgium (Flanders)		70	3-5 Q_{DWFm}	5-10 Q_{DWFm} NO=7/a	Remaining spilling vol. with T = 1/7 year	-	yes
Denmark		45-50	2 Q_{DWFp}	5 Q_{DWFp} NO=2-10/a		-	yes
Finland		10-15	2 Q_{DWFp}	6-7 Q_{DWFm}		-	
France	(CERTU, 2003)	70-80	2-3 Q_{DWFm}	3 Q_{DWFp}	Interception of rainfall with T = 3-6 months	-	sometimes
Germany	(ATV-A 128, 1992)	67	2 Q_{DWFp}	7.5-15 $l/(s \cdot ha_{Aimp})$	10-40 m^3/ha_{Aimp}	90% of COD load	
	(BWK, 2001)		-	-	-	-	yes
Greece		20	2 Q_{DWFm}	3-6 Q_{DWFm}		-	sometimes
Ireland		60-80	3 Q_{DWFm}	6-9 Q_{DWFm}		-	sometimes
Italy	Local e.g. (Toscana, 2006)	60-70	2 Q_{DWFm}	3-5 Q_{DWFm}		-	sometimes
Luxem-bourg	(ATV-A 128, 1992)	80-90	2-3 Q_{DWFm}	7.5-15 $l/(s \cdot ha_{Aimp})$	10-40 m^3/ha_{Aimp}	-	
Nether-lands		74	3 Q_{DWFp}	5 Q_{DWFm} NO = 3-10/a	ca. 70 m^3/ha_{Aimp}	-	sometimes
Portugal		40-50	2 Q_{DWFm}	6 Q_{DWFm}		yes	sometimes
Spain		70	2 Q_{DWFm}	5 Q_{DWFm}		-	no
Sweden		25-40	3-4 Q_{DWFm}	5-20 Q_{DWFm}		-	
Switzer-land	(AfU, 1977); (GSchG, 1991); (GSchV, 1998)		2 Q_{DWFp}			-	yes
UK	(FWR, 1998)	70	3 Q_{DWFm}	6-9 Q_{DWFm}	t_D= 2h at 3 Q_{DWF}	-	yes
USA	(CWA, 1972);(US EPA, 1995)			NO=4-6 /a		85% combined wastewater	yes

Q_{DWFp}: dry weather flow peak discharge; Q_{DWFm}: dry weather flow mean discharge; A_{imp}: connected impervious area; NO: number of overflows; t_D: detention time; T: return period

ENVIRONMENTAL QUALITY STANDARDS AND APPROACHES

Biological water quality indicators

Biological monitoring is a powerful tool for assessing river degradation because biota form a complex web which integrates the condition of the river (Karr and Chu, 2000).

The first biological indicators for water quality have already been developed at the beginning of the 20th century, e.g. the saprobic system by Kolkwitz and Marsson in 1908 (Mauch, 1998; Schwoerbel, 1999). During the last decades, ecological quality has been increasingly acknowledged as valuable (Norris and Hawkins, 2000). Accordingly the legislation changed (examples are the US Clean Water Act (CWA, 1972) or the European Water Framework Directive (2000/60/EC)). For this purpose new biological indicators have been developed which are able to reflect more conditions but only the saprobic status.

Generally, biological quality is quantified by features which describe the aquatic biocoenosis, e.g. biocoenosis composition, species abundance or allocation of functional feeding groups. Numerous biological assessment methods have been developed for all kinds of aquatic taxonomic groups (confer for example PAEQANN deliverable 3 (2005)). Biological assessment methods have usually been developed site specific and purpose specific (Simon, 2000) and are based on comparison of the actual state and reference conditions (Angermeier and Karr, 1994). Establishing good reference conditions is crucial, but often no natural watercourses are available and therefore historic data or minimally impaired sites have to be used instead (Nijboer et al., 2004).

Evaluation of biological water quality is generally made in form of indices (e.g. saprobic indices, diversity indices or biotic indices) (PAEQANN, 2005). Usually not single indicators are used in ecology, but several indices are integrated for evaluation, e.g. in a multimetric as the index of biotic integrity (Boulton, 1999).

A problem of biological water quality assessment is that the measurement can often only be made by experts, and strictly standardised assessment procedures (e.g. for sampling) are necessary because otherwise the results are not comparable (Simon, 2000). The information of biological water quality depends on the method with which it has been measured, e.g. life time of the measured species or the purpose of the method. Life time decides the time horizon, e.g. fish integrate various impacts over their relative long lifetime and therefore indices based on fish allow long-term conclusions. In contrast, macroinvertebrates have significantly shorter lifespan and accordingly shorter is the time horizon of methods based on them (PAEQANN, 2005).

The purpose of biological water quality assessment methods is different. For example, if as indicator some saprobic index is chosen, the biological quality relates only to impacts by organic substances (Schwoerbel, 1999); other methods aim at the assessment of ecological quality or biological integrity. River health indicators however are composed of numerous indicators including saprobic quality (Angermeier and Karr, 1994), species diversity and even human uses (Simon, 2000). Indicators can be early warning indicators, diagnostic indicators or compliance indicators (Boulton, 1999).

Following first biological measurement methods for saprobic quality are described, these indices evaluate the organic pollution status of a watercourse. Then some assessment methods for ecological integrity are explained, here two main types are to be distinguished: multimetric and predictive modelling approaches. However, currently none of the biological assessment methods is able to give guidance for wet-weather control design and planning, as the indicators reflect the integrated reaction to all impacts in the river. Thus acute impacts and their cause cannot be identified.

Table 4: Examples for biological assessment methods and countries in which they are applied.

Approach	Impact type	Method	Name	Indicator	Country	References
Saprobic indices	Organic pollution	Statistical analysis of measurement data	Saprobic index	Macro-invertebrates	Germany	(DIN 38 410, 1990)
					Austria	(OENORM M 6232, 1997)
Indices of Biotic Integrity	Impact on biological integrity / ecological quality	Multimetric; statistical analysis of measurement data	Index of Biotic Integrity	Fish, macro-invertebrates, plant	USA	(Karr and Chu, 2000)
					Australia	(Harris and Silveira, 1999)
			AQEM	Macro-invertebrates	Europe	(Hering *et al.*, 2004)
			European Fish Index	Fish	Europe	(Pont *et al.*, 2006)
RIVPACS	Deviation from natural status	Predictive modelling	RIVPACS	Macro-invertebrates	UK	(Clarke *et al.*, 2003)
			AUSRIVAS	Macro-invertebrates	Australia	(Smith *et al.*, 1999)
			BEAST	Macro-invertebrates	Canada	(Bailey *et al.*, 1998)
PAEQANN	Deviation from natural status	Predictive modelling	PAEQANN	Diatoms, macro-invertebrates, fish	Europe	(Lek *et al.*, 2003)

Assessment of organic pollution

Most methods to assess the biological quality of a watercourse are based on macroinvertebrates. In the German speaking region, the biological quality of the rivers is measured with the saprobic index (e.g. Germany DIN 38 410 (1990), Austria ÖNorm 6232 (1997)). The saprobic index is a quite old system, its development already started in the beginning of the 20th century (Schwoerbel, 1999). Unfortunately it can only describe impacts by organic pollution but not other impacts, as for example toxic, hydraulic impacts or a poor morphological status of a river (Böhmer et al., 2004).

The saprobic index is based on the fact that some animals have very specific requirements regarding nutrient and oxygen concentration. Species whose abundances are significantly correlated with saprobic condition are identified and used as indicators. For example organisms with high oxygen requirements are indicators for clean waters, whereas the occurrence of organisms which can tolerate low oxygen concentrations indicates polluted waters. The natural saprobic status of running waters depends on their type, i.e. lowland watercourses will naturally have a slightly increased saprobic index (GDCh, 2004).

Assessment of biological integrity

Saprobic indices include only species which are indicators for increased nutrient concentrations but they are not able to reflect toxic or morphological impacts. Therefore they are not sufficient to reflect the biological integrity or ecological quality of the watercourse. (The term “biological integrity” is used in the US Clean Water Act and refers to conditions under little or no impacts from anthropogenic actions (Angermeier and Karr, 1994). “Good ecological status” comes from the European Water Framework Directive and is described as only small deviation from the natural status (Achleitner et al., 2005)). The increasing recognition of good ecological quality as a value and its definition as goal in legislation has led to various research to develop assessment protocols and new indices which can represent various stressors that can impact the ecological quality.

There are two main types of approaches for the assessment of biological integrity: the multimetric approach (e.g. index of biotic integrity (IBI)), and the predictive modelling approach (e.g. RIVPACS). Both the multimetric approach and the predictive modelling approach need to establish reference conditions from measurements at unimpaired or minimally impaired sites. In the predictive model approach an empirical model is developed which makes site-specific predictions of expected taxonomic composition. Measurements at a specific site can then be compared against this prediction. Multimetrics are based on the identification of spatially discrete areas within which the range of values is small for each indicator and which are used for the definition of the expected conditions (Norris and Hawkins, 2000).

On which approach is the better, there are different opinions: while Karr and Chu (2000) think multimetrics are superior to the predictive model approach, Norris and Hawkins (2000) believe the predictive model approach to be better.

Indices of biotic integrity

In the United States of America indices of biotic integrity are widely used to assess the ecological quality of watercourses. The index of biotic integrity (IBI) was originally developed by Karr in the 1980ties but several new IBIs have been developed (Novotny et al., 2005). The original IBI was created to assess lotic fish communities. It consisted of numerical criteria on species composition, and diversity, trophic composition, population density, tolerance to anthropogenic impacts, and health. These indicators were integrated into a single numerical index. The watercourse was assessed by comparing the measured condition against the region-specific expectation for an undegraded watercourse (Angermeier and Karr, 1994). Based on this original concept new indices of biotic integrity were developed, on the one hand for other regions, also outside the USA. IBIs have been developed for example in Australia (e.g. by Harris and Silveira (1999)). On the other hand indices have been created which also include other aquatic groups, e.g. using macroinvertebrates or periphyton, to improve the applicability of the IBI for specific waters like lakes or large rivers.

IBIs provide an integrated assessment of water shed and water pollution stressors. However, linking the IBIs to stressors is still a problem (Novotny et al., 2005). Therefore currently the IBIs do not allow identification of impact sources.

In Europe efforts to establish Europe-wide multimetrics have only started recently, e.g. by the European projects AQEM (www.aqem.de), FAME (fame.boku.ac.at) or STAR (www.eu-star.at). The main driving force for their development is the European Water Framework Directive. For example the European project AQEM ("The Development and Testing of an Integrated Assessment System for the Ecological Quality of Streams and Rivers throughout Europe using Benthic Macroinvertebrates") developed a multimetric classification based on benthic macroinvertebrates (Hering et al., 2004) determining quality by comparison with reference conditions (Buffagni et al., 2001; Nijboer et al., 2004). The sampling protocol developed in AQEM was further improved, e.g. in the European project STAR (Standardisation of River Classifications) (Haase et al., 2004), or by Böhmer et al. (2004). Also fish based indices are developed, e.g. the project FAME developed the European Fish Index (EFI) which is based on the concept of the IBI (FAME, 2005; Pont et al., 2006).

Predictive modelling approaches

Predictive modelling approaches predict the community which should be present at a given site based on relationships between environmental parameters and biological indicators (Norris and Hawkins, 2000). Examples for the assessment of biological quality based on predictive modelling are the British RIVPACS (River Invertebrate Prediction and Classification System) or the European approach PAEQANN (Predicting Aquatic Ecosystems Quality using Artificial Neural Networks). Both approaches predict communities which should be present in a specific stream type under minimally anthropogenic impacted conditions. But while RIVPACS is based on statistics (Clarke et al., 2003), in PAEQANN the relations are established by an artificial neural network (Gevrey et al., 2003). RIVPACS has been developed in Great Britain and predicts macroinvertebrate communities on the basis of statistical relationships between the fauna and environmental characteristics (e.g. altitude or distance from source) (Clarke et al., 2003). RIVPACS based approaches have been developed also in various other countries like AUSRIVAS in Australia (Schofield and Davies, 1996; Smith et al., 1999) or BEAST in Canada (Reynoldson et al. (1995) cited in Bailey et al., (1998)) but there have been also developments in the USA, New Zealand and Europe (www.ceh.ac.uk/sections/re/ RIVPACS.html). PAEQANN is based on an artificial neural network (ANN) approach (Gevrey et al., 2003) and predicts diatom (Gevrey et al., 2004; Tison et al., 2004), macroinvertebrate (Park et al., 2003a; Park et al., 2003b; Park et al., 2004) and fish communities (Aguilar Ibarra et al., 2003).

Assessment of intermittent impacts on the basis of physical and chemical parameters

The most common approach is the assessment of intermittent impacts on the basis of physical and chemical parameters because these are easy to measure and can be included in computer simulation tools. This is important because numerical modelling has become a major tool in urban drainage planning. Various standards to assess acute pollution impacts have been developed during the last decades. Although they are mainly designed for combined sewer overflows, they are generally also applicable for stormsewer outfalls. Among the first are standards for acute oxygen deficits (Danish Engineering Union Wastewater Committee (1985) or US EPA (1986)). The identification of synergistic effects of oxygen deficits and un-ionised ammonia led to the development of interrelated standards for these two substances (FWR, 1998). Today most approaches also specify standards for hydraulic impacts, either on discharge volume (e.g. BWK M3 (BWK, 2001) or OEWAV R19 (Draft 2003)) or on the frequency of critical (erosion producing) events (Rossi et al., 2004a). Some of the ambient water quality based approaches for the assessment of combined sewer overflows are described in the following pages.

Water quality standards are usually based on toxicological data obtained under laboratory conditions with single substance and constant concentration. From these experiments toxicity data for specific impact magnitude and exposure duration are derived, like ‘lethal concentration’ (for example LC_{50}(96hr) is the concentration at which 50% of the test organism population die during 96 hours) or ‘no observed effect concentration’ (NOEC). The usefulness of such data to describe the effects of combined sewer overflows is limited. Therefore experiments have been conducted applying for example fluctuating concentrations (Brooks et al., 1996; House et al., 1993). Others investigated the impact resulting from combined oxygen deficits and un-ionised ammonia impacts (Borchardt, 1992; Gammeter and Frutiger, 1990; Magaud et al., 1997) or compared the effect of ammonia as single substance with the effect of ammonia from wastewater (Borchardt, 1992). Beside acute mortality also post mortality increase (Borchardt, 1992) and behavioural changes have been investigated (Borchardt, 1992; Borchardt and Statzner, 1990; Gammeter and Frutiger, 1990). To improve the transferability of the results to natural conditions, some authors used artificial channels with near-natural conditions (e.g. studies of the effect of increased discharge, ammonia and low oxygen concentration on macroinvertebrates (Gammeter and Frutiger, 1990) or macroinvertebrates and salmonids (Borchardt, 1992; Borchardt and Statzner, 1990).

Novotny and Witte (1997) prepared risk functions for wet-weather discharges on the basis of toxicity data. Their approach was however only based on laboratory toxicity data and therefore its transferability to natural conditions is limited (Ellis, 2000). Further, most risk approaches only consider direct effects (as acute mortality) but neglect indirect effects (i.e. effects on species' interactions, which can influence significantly the ecosystem's response (Preston, 2002), see Figure 2).

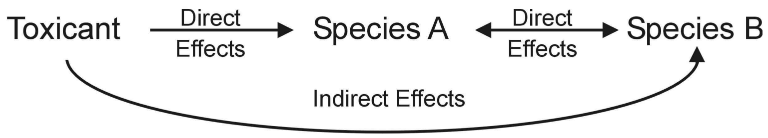

Figure 2: Conceptual model of indirect effects in ecotoxicology. Indirect ecological effects may result from the effects of toxicants on the interactions of interdependent species (redrawn from Preston (2002)).

There is a considerable risk from numerous possibly present pollutants in both wastewater and stormwater, which cannot be monitored due to their extremely high number. Eriksson et al. (2002) identified 900 different xenobiotic organic compounds which are possibly present in Swedish greywater (household wastewater from kitchen, bath and laundry). Therefore they developed a methodology for stormwater management to identify the most critical and representative chemical pollutants. With this methodology, the site-specific priority pollutants can be identified which can subsequently be included in planning and monitoring (Eriksson et al., 2005). The identification of specifically problematic substances has also been done in the European Water Framework Directive, which specifies priority and priority hazardous substances (annex X, (2000/60/EC)). Future wet-weather flow management in Europe will have to include these substances.

It has also not been tested if a scaling down of toxicity data (i.e. to calculate for example an LC_{50}(8hr) from a LC_{50}(96hr)) produces correct results (Ellis, 2000). Often a differentiation in salmonid waters and cyprinid waters (e.g. 78/659/EEC) or coldwater and warmwater (e.g. US EPA, 1986) is made, because salmonid/coldwater fishes are adapted to clean highland streams with high oxygen concentration (US EPA, 1986). Sometimes also limits for salmonid spawning grounds are included because their eggs and fry (living in the interstitial) are endangered by oxygen depletion in the sediment (Argent and Flebbe, 1999).

It has been found that after a disturbance, (e.g. by low oxygen concentrations or high un-ionised ammonia concentrations) fish are more susceptible to other pollutants (FWR, 1998), and that an increase in the frequency of disturbances leads to an increased recovery time and a reduced capability to cope with disturbance (Burton and Pitt, 2002). Low dissolved oxygen concentration cause invertebrate drift and it takes some time until the population has recovered. Un-ionised ammonia causes even in sublethal concentrations permanent damage to fish (FWR, 1998). Therefore limits for acute impacts of un-ionised ammonia and oxygen deficits often include recovery time. The recovery time is specified as intensity - duration - frequency limits. The shorter the duration of the pollutant exposure and the more infrequent it occurs, the more severe impacts are tolerable. As an example for intensity- duration- frequency dependent limits the standards for un-ionised ammonia of the UPM manual (FWR, 1998) are shown in Figure 3.

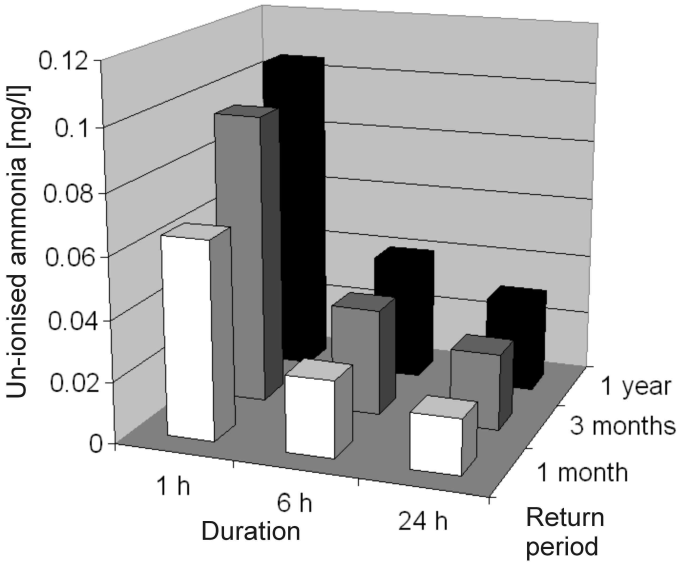

Figure 3: Example for intensity - duration - frequency dependent limits. Fundamental intermittent standards for un-ionised ammonia of the Urban Pollution Management Manual (FWR, 1998).

The Danish approach

The Danish approach to prevent pollution impacts from combined sewer overflows specifies criteria for minimum oxygen concentrations (Danish Engineering Union Wastewater Committee, 1985). The quality criteria for oxygen are based on two endpoints. One endpoint is the oxygen concentration sufficient to support fish lastingly, the return period is defined as 0.1 years. The other endpoint is the absolute minimum oxygen concentration which can be tolerated for only very short periods, here the LC_{50} (1h), respectively LC_{50} (12h), was applied. For the absolute minimum concentration a maximum return period (T_{max}) is defined. Between these two oxygen concentrations and their respective return periods, the return periods for intermediate oxygen concentrations are linear interpolated. This interpolation is done for two impact durations (1 hour and 12 hours) and for three river types (salmonid spawning grounds, salmonid waters and cyprinid waters). An English description of the approach is included in House et al. (1993).

Table 5: Limits of the Danish approach (Danish Engineering Union Wastewater Committee, 1985)

River Type	Maximum return period (T_{max})	Duration: 1h $T \leq 0.1$ year	Duration: 12h $T \leq 0.1$ year	Duration: 1h T_{max}	Duration: 12h T_{max}
Salmonid spawning ground	16 years	8 mg O_2/l	9 mg O_2/l	-	-
Salmonid water	12 years	6 mg O_2/l	7 mg O_2/l	1.0 mg O_2/l	1.5 mg O_2/l
Cyprinid water	8 years	4 mg O_2/l	5 mg O_2/l	1.5 mg O_2/l	2.0 mg O_2/l

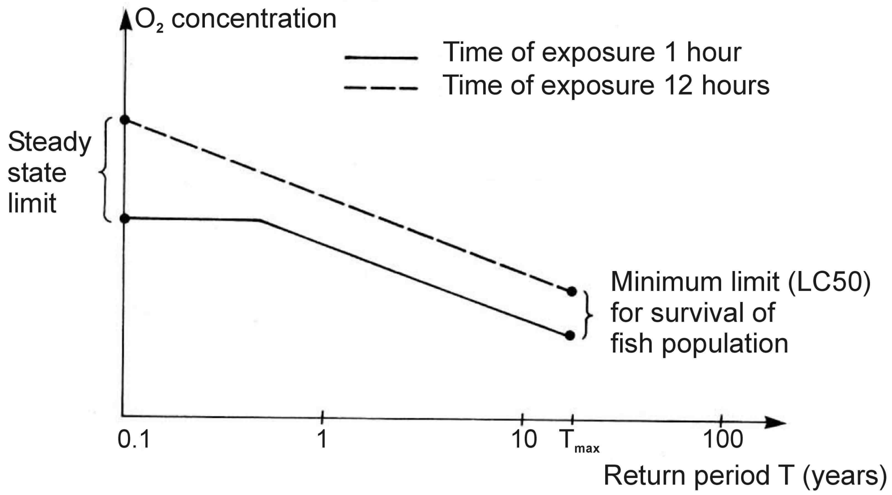

Figure 4: Principle of Danish oxygen quality criteria (Danish Engineering Union Wastewater Committee, 1985).

Urban Pollution Management Manual (UPM Manual)

The Urban Pollution Management (UPM) developed in Great Britain (FWR, 1998) provides standards for acute effects of low dissolved oxygen concentrations and un-ionised ammonia, called Fundamental Intermittent Standards. After exposure to un-ionised ammonia or low dissolved oxygen concentrations, fish will be more susceptible to exposure to any contaminant. The UPM includes this effect by defining recovery periods, the higher the un-ionised ammonia concentration/the lower the dissolved oxygen concentration: the longer the recovery period (see Table 6 and Table 7). However, it is also noted in the UPM manual that these recovery periods could not be defined with any certainty. Low dissolved oxygen and high un-ionised ammonia have synergistic effects, i.e. they aggravate each other. Therefore the UPM defines correction factors for both substances in case the other substance is also present in high concentrations (as indicated below Table 6 and Table 7).

Table 6: Fundamental Intermittent Standards for dissolved oxygen [mg O/l] – concentration/duration threshold not to be breached more frequently than shown.

Return Period	Sustainable Salmonid Fishery			Sustainable Cyprinid Fishery			Marginal Cyprinid Fishery Ecosystem		
	1 h	6 h	24 h	1 h	6 h	24 h	1 h	6 h	24 h
1 month	5.0	5.5	6.0	4.0	5.0	5.5	3.0	3.5	4.0
3 months	4.5	5.0	5.5	3.5	4.5	5.0	2.5	3.0	3.5
1 year	4.0	4.5	5.0	3.0	4.0	4.5	2.0	2.5	3.0

NH_3 concentration	*Correction factor to be applied to DO thresholds*
less than 0.02 mg NH_3-N/l	No correction
0.02 – 0.15 mg NH_3-N/l	+ (0.97 x $\log_e$ (mg NH_3-N/l) + 3.8 mg O/l
more than 0.15 mg NH_3-N/l	+ 2 mg O/l
	Salmonid spawning grounds: + 3 mg O/l

Table 7: Fundamental Intermittent standards for un-ionised ammonia [mg NH_3-N/l] – concentration/ duration threshold not to be breached more frequently than shown.

Return Period	Sustainable Salmonid Fishery 1 h	6 h	24 h	Sustainable Cyprinid Fishery 1 h	6 h	24 h	Marginal Cyprinid Fishery Ecosystem 1 h	6 h	24 h
1 month	0.065	0.025	0.018	0.150	0.075	0.030	0.175	0.100	0.050
3 months	0.095	0.035	0.025	0.225	0.125	0.050	0.250	0.150	0.080
1 year	0.105	0.040	0.030	0.250	0.150	0.065	0.300	0.200	0.140
DO concentration	*Correction factor to be applied to un-ionised ammonia thresholds*								
5 mg O/l or higher	No correction								
< 5 mg O/l	x 0.0126(mg O/l)$^{2.72}$								

Acute criteria for pollutants in the USA

The National Recommended Water Quality Criteria of the United States of America (US EPA, 2002) include, besides the standards for long-term impacts (called chronic criterion, CCC), also for various substances standards for acute impacts. These standards for acute impacts are called acute criterion (CMC). Acute criteria are defined for example for ammonia, dissolved oxygen or chloride, but also for various heavy metals (e.g. cadmium, copper, lead or zinc).

The freshwater ammonia criterion is pH, temperature and life-stage dependent. The CMC for total ammonia nitrogen is specified as one-hour average concentration which is not to be exceeded more than once every three years on the average. The CMC (acute criterion) is calculated using the following equations (US EPA, 2002):

Where salmonid fish are present:

$$CMC = \frac{0.275}{1+10^{7.204-pH}} + \frac{39.0}{1+10^{pH-7.204}} \qquad \text{[mg N/l]}$$

Or where salmonid fish are not present:

$$CMC = \frac{0.411}{1+10^{7.204-pH}} + \frac{58.4}{1+10^{pH-7.204}} \qquad \text{[mg N/l]}$$

The criteria for dissolved oxygen are defined in “Ambient Water Quality Criteria for Dissolved Oxygen” (US EPA, 1986). For dissolved oxygen, it is distinguished between coldwater criteria and warmwater criteria. These are further distinguished into criteria for early life stages and other life stages (see Table 8).

Table 8: Water quality criteria for ambient dissolved oxygen concentration, i= intergravel oxygen concentrations (US EPA, 1986).

	Coldwater criterion		Warmwater Criterion	
	Early Life Stages	Other Life Stages	Early Life Stages	Other Life Stages
30 Day Mean	not available	6.5	not available	5.5
7 Day Mean	9.5 (5.6 ^{i})	not available	6.0	not available
7 Day Mean Minimum	not available	5.0	not available	4.0
1 Day Minimum	8.0 (5.0 ^{i})	4.0	5.0	3.0

BWK M3

The technical fact sheet M3 of the German Association of Water Resources, Waste Management and Land Reclamation Engineers (BWK) is the first comprehensive ambient water quality approach for the assessment of wet-weather discharges in Germany (BWK, 2001). In this fact sheet, standards for morphological impacts, un-ionised ammonia and dissolved oxygen are specified. The standard for morphological impacts due to an increase of the watercourse's discharge by urban wet-weather discharges is defined as percentage of the near-natural 1-year flood. The sum of all urban wet-weather discharges into a water body (Q_{E1}, zul.), that is not to be exceeded, is defined in the following formula:

$$Q_{E1,zul.} < 1.0 * Hq_{1,pnat} * \frac{A_{red}}{100} + x * Hq_{1,pnat} * A_{Eo} [l/s]$$

$Hq_{1, pnat}$ ... potential near-natural yearly flood
A_{red} ... paved area of attached urban area
A_{Eo} ... surface catchment of the watercourse
$Q_{e1, zul}$ ... acceptable critical yearly influent discharge
x ... multiplication factor for the acceptable increase of discharge by anthropogenic influences, normally 0.1

The standard in BWK M3 for dissolved oxygen is more than 5 mg O_2 /l, and for un-ionised ammonia less than 0.1 mg NH_3-N /l. The standards for impacts by low dissolved oxygen and high un-ionised ammonia concentrations are chosen so that even through synergistic effects sensitive organisms are not apparently damaged. The same should be valid for standards for suspended solids, but it was found that specification of general valid limits for suspended solids was not possible with the present knowledge.

The Swiss project STORM

In Switzerland the project STORM (Krejci et al., 2004b) developed basic knowledge and methods to harmonise requirements for wastewater discharges at wet-weather conditions with current Swiss legislation regarding water protection. The Swiss legislation requires amongst others integrated, site specific and problem oriented planning of urban drainage (Krejci and Kreikenbaum, 2004). Therefore limits for acute impacts were proposed (Rossi and Hari, 2004; Rossi et al., 2004a; Rossi et al., 2004b), and simulation tools for screening and planning were produced (REBEKA (Rauch et al., 2002) and REBEKA II (Fankhauser, 2004)). Further planning procedures were created based on the developed knowledge (Kreikenbaum et al., 2004; Krejci, 2004a) and tested (Krejci, 2004b; Krejci et al., 2004a).

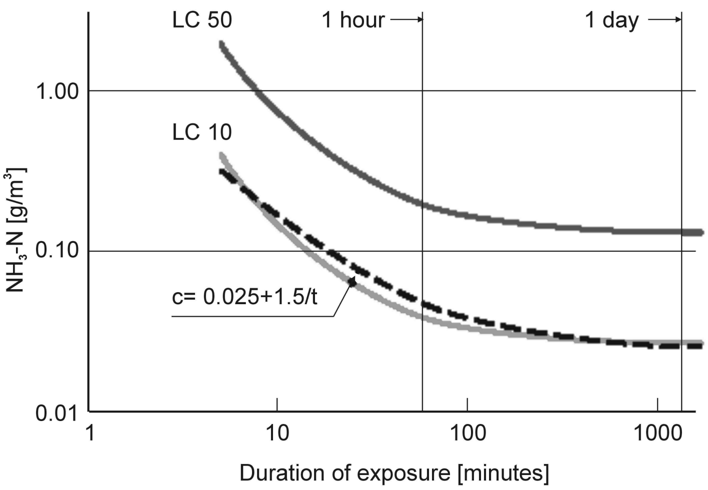

Figure 5: Critical intensity and duration of impact of un-ionised ammonia for trout. The proposed standards are shown as dashed line. These values should not be exceeded more often than once every five years (Rossi *et al.*, 2004a).

The proposed standards for un-ionised ammonia (see Figure 5) were defined such that in case of an exceedance at 100% oxygen saturation on an average 10% of the trout population die, respectively 20% of the population at 40% oxygen saturation. For the standards for solids, STORM distinguishes between effects due to increased turbidity and the effects of sediment accumulation. For turbidity the following standards are proposed: 50 mg/l during 60 minutes, 300 mg/l during 10 minutes and for longer exposure times (up to 24 hours) maximum 25 mg/l. The standards for particle accumulation are divided in standards to prevent the blockage of river bed, accumulation of persistent substances and oxygen depletion (see Table 9). During spawning season (in Switzerland between September and March) no blockage should be tolerated (Rossi et al., 2004a).

Table 9: Proposals for standards for maximal TSS accumulation rates in sediments for a gravel river bed.

Criterion	Standard for TSS accumulation	Maximum time of standard exceedance [% per year]
Blockage of river bed (physical)	625 g_{TSS} m^{-2} a^{-1}	20 %
Accumulation of persistent substances (heavy metals, PAH)	25 g_{TSS} m^{-2} a^{-1}	5 %
Oxygen depletion:		
Combined system (overflows)	5 g_{TSS} m^{-2} d^{-1}	10% (0% between September and March)
Separate system	16 g_{TSS} m^{-2} d^{-1}	10% (0% between September and March)

As standards for oxygen depletion, the Danish approach (see Figure 4) and the oxygen standards of the British UPM for Sustainable Salmonid Fishery (see Table 6) are proposed (Rossi et al., 2004a).

Hydraulic impacts of wet-weather discharges are prevented by restricting of the number of erosion events per year in the receiving water. The number of erosion events, which are acceptable for a river, depends significantly on the status of the receiving water. The better its morphological quality, the higher the tolerable number of erosion events (Frutiger and Gammeter, 1996). The assessment of morphological quality is based on the Swiss methodology (“eco-morphological quality”), see Table 10. A critical event, i.e. an event that causes erosion, is determined using the approach of Meyer-Peter (Rauch et al., 2002).

Table 10: Standards of hydraulic-mechanical impacts, maximum number of critical events per year. class I means “natural/near natural”, class II “slightly degraded”, class III “highly degraded” and class IV “unnatural/ artificial” (Rossi et al., 2004a).

Eco-morphological	Variability of width			
quality	significant	restricted	none	
Class I	10	5	3	critical events/ year
Class II	5	3	1	critical events/ year
Class III & IV	3	1	<1	critical events/ year

The temperature which is tolerated by the aquatic organisms depends in general on numerous factors, e.g. the temperature at which the organisms normally live, the fish species and the development stage, or the duration and magnitude of the temperature change (Rossi and Hari, 2004). The standards proposed for acute temperature changes are shown in Table 5 and Figure 6.

Table 11: Standards for the water temperature in the receiving water (Rossi and Hari, 2004; Rossi et al., 2004a).

Temperature parameter	Standard
Maximum temperature after the discharge from the sewer (summer time)	< 25 °C (or Figure 6)
Maximum temperature after the discharge from the sewer (winter time)	< 12 °C
Maximal tolerable changes of temperature by discharges from the sewer	< 7 °C

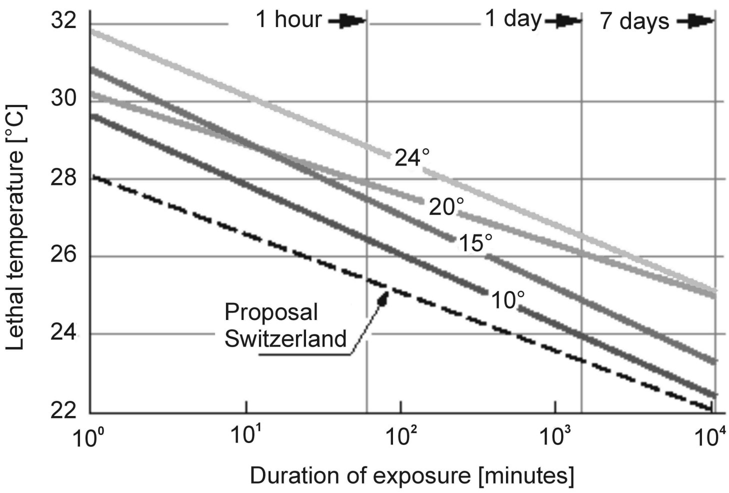

Figure 6: Survival temperature in dependence of the duration of exposure for juvenile salmonids (in the first to third year) at different acclimatisation temperatures (10, 15, 20, 24 °C) (Rossi and Hari, 2004).

The results of the project STORM are the basis for the development of a new Swiss guideline for wet-weather discharges from urban drainage (Krejci et al., 2004b).

Austrian OEWAV guideline 19 (Draft)

In Austria, the fact sheet 19 regarding the design of combined sewer overflows is currently being updated to include an ambient water quality based approach (OEWAV R19, Draft 2003). The new fact sheet 19 consists of an emission based approach for the design of overflow structures, which specifies for the total sewer system a percentage for suspended and for particulate substances, which is to be conveyed to the wastewater treatment plant. Additionally it also includes further requirements for an ambient water quality approach. For the assessment of the impact, all discharges into a stretch with the length of 1000 times the width of the receiving water's water surface, respectively minimum 500 m and maximum 2500 m length, are to be considered together.

The limit for hydraulic impacts is defined in dependence of the 1-year flood of watercourse:

$$Q_{e,1} \geq 0.1\ to\ 0.5 \cdot HQ_1$$

$Q_{e,1}$ ... 1-year stormwater runoff from combined sewer overflows and storm sewer outlets (l/s)

HQ_1 ... 1-year flood of watercourse (l/s)

The smaller value of 10% is valid for watercourses with mainly clayey- sandy watercourse sediment, low variability of width and low recolonisation potential from adjacent watercourses or the upstream stretch. The higher value of 50% is valid for watercourses with rocky sediment, high variability of channel width and high recolonisation potential. Intermediate values are to be determined according to the eco-morphological state of the watercourse.

To avoid acute impacts by un-ionised ammonia, the ammonia concentration should not exceed on short-term (1 hour) in salmonid waters 2.5 mg NH_4-N/l (being equal to 0.1 mg/l NH_3-N at pH 8 and 20°C) and in cyprinid waters 5 mg NH_4-N/l (being equal to 0.2 mg/l NH_3-N at pH 8 and 20°C). The dissolved oxygen concentration should not decrease below 5 mg/l in the water, and it is assumed that this standard prevents anaerobic status in the upper layer of the river bed. For suspended solids, the standard by the German ATV (ATV, 1993) of 50 mg/l is proposed (although other authors think the scientific basis for this standard is insufficient (BWK, 2001)).

Biological assessment of combined sewer overflow impacts

Some work has already been done on the effect of CSO discharges on biocoenosis of the receiving water, but it is only very little compared to the huge amount of work performed with simulations and chemical water quality parameters. Gammeter (1996) investigated in the 'integrated urban drainage case study in Fehraltdorf' (Switzerland) the impact of urban drainage on the benthic invertebrate community of several small mountain running waters. She found that the most severe impacts of wet weather discharges resulted from factors with long-term impacts, like habitat or morphology changes. Fuchs (1998) examined the impact of a CSO on a small German mountain stream in the Black Forest. He tested various indices based on macroinvertebrates for their ability to reflect the CSO impact and concluded that indices based on Ephemeroptera, Plecoptera and Trichoptera are suitable to identify CSO impacts, contrary to the saprobic index. Similar to the findings in Switzerland, he found the main impact of the investigated CSO was a hydraulic impact. Podraza and Widera (1998) investigated protozoan and macroinvertebrate communities upstream and downstream of a CSO in a small urban mountain stream in North Rhine Westphalia in Germany. Their results show that these two groups respond completely different to the CSO discharges. Protozoa feed on bacteria whose amount is increased both by the bacteria contained in the CSO discharges and by better bacterial growth due to the nutrient input by the CSO. Thus the number of protozoan taxa and their abundance, especially of sessile, bacteriovorus species, was found generally to be elevated downstream of the CSO (Widera, 2000). Macroinvertebrates on the other hand are due to their size and habitats more susceptible to the hydraulic impact of the CSO and were thus rather decreased in taxa and abundance downstream of the CSO compared to the reference sampling point upstream. Partly Ephemeroptera and Gammarida were even missing below the CSO (Podraza, 1999). Still, the study showed that the influence of the CSO for both groups could only be identified by long-term regular measurements at the species level (Podraza and Widera, 1998).

Assessment of urban wet-weather flows with whole effluent toxicity testing

The ambient water quality approaches described above are based on measurements of different chemical parameters in the receiving water. Both stormwater runoff and combined sewer overflows contain numerous chemicals, so that the measurement of all possibly present substances is impossible. Further the complex composition of the discharges can result in synergistic or even antagonistic effects. Testing of the impact of the whole discharge allows an overall evaluation. It can be done with direct toxicity testing or with in situ ecotoxicological studies. These approaches test the whole effluent against a limited set of aquatic organisms for its ecotoxicity. Direct toxicity testing methods are called among others ‘direct toxicity assessment’ (DTA) or ‘whole effluent toxicity’ (WET) (Whitehouse, 2001).

Direct toxicity testing is performed under laboratory conditions. The test organisms usually represent different trophic levels (e.g. bacteria, algae, invertebrates, and fish) (Power and Boumphrey, 2004). Generally lethality is measured and therefore toxicity testing cannot assess sublethal stress (Ellis, 2000). Other toxicity tests address cytotoxicity (cellular damage) or genotoxicity (damage to the genetic material). These tests are performed using cells or bioparticles (Marsalek et al., 1999b). Further toxicity assessment methods have been developed which also consider post-exposure effects (e.g. post-exposure lethal exposure time by Brent and Herricks (1999)), to overcome the problem of delayed impacts.

Toxicity testing can be made for water but also for sediment extracts. For CSO it seems important to consider both, as Marsalek et al. (1999b) found that CSO discharges generally seemed less toxic than stormwater. They concluded that this is due to the fact that the bioavailability of pollutants is reduced in the CSO discharges by sorption to organic matter. This however would lead to increased toxicity of the CSO sludge.

Toxicity testing methods have been applied worldwide for discharge permits of industrial effluents. Pilot studies for combined sewer overflows have been made for example in Canada (Marsalek et al., 1999b), the United States and Great Britain (Ellis, 2000). Direct toxicity testing has been applied for stormwater runoff (see for example Marsalek et al. (1999a), or Boxall and Maltby (1995)). It has already been used to authorise stormwater effluents, although the technique is still premature. One problem is the variability in survival rates, which occurs especially at the intermediate toxicity levels that are used as basis for permits (Ellis, 2000). At the current stage toxicity data provide not sufficient information for planning and design of wet-weather flow management (Marsalek et al., 1999b).

The acceptable end-of-pipe toxicity can be derived as (Ellis, 2000):

$$\text{acceptable toxicity} = \left(\frac{\text{end}-\text{of}-\text{pipe toxicity for most sensitive species}}{\text{safety factor (normally } x\,10)} \right) \geq \text{effluent concentration}_{\text{edge of mixing zone}}$$

An assessment of acute toxicity of wet-weather discharges can be made in situ by tests using benthic macroinvertebrates, keeping them in cages upstream and downstream of investigated structure. Beside acute lethal effects, also accumulation of persistent toxic substances can thus be investigated (House et al., 1993; Mulliss et al., 1997; Mulliss et al., 1994; Mulliss et al., 1996).

Behavioural changes, e.g. drift, elevated respiration movements, active avoidance reactions, or perturbed behaviour are more sensitive indicators for stress from wet-weather discharges than mortality. Wet-weather discharges can lead to increased drift rates, up to highly elevated drift ('catastrophic drift') (Borchardt, 1992), because of active avoidance reactions, changes of location (e.g. from bottom side to upper side of a stone) or perturbed behaviour (e.g. uncoordinated movements or rigor) (Borchardt, 1992; Gammeter and Frutiger, 1990). Behavioural changes however are difficult to monitor in situ, although an automatic device for monitoring behavioural changes of fish has been constructed using gill ventilation responses (sub-lethal stress response) (described in House et al., (1993)).

DISCUSSION

In future the status of the receiving water will probably become the benchmark for wet-weather flow management. Under this presumption the different approaches to manage pollution from wet-weather flows are to be evaluated. The conventional management strategy is the definition of emission limits. The emission limits are usually derived with a 'precautionary principle' approach. This means that elimination of pollution on the state-of-the-art level is provided. The rules for the design are clearly set and guarantee planning with relatively certain cost estimates (Krejci and Kreikenbaum, 2004). The requirements are defined such that they are relatively consistent, so that local conditions do not cause excessive economic disadvantages. Emission based strategies, however, cannot assure the quality of the receiving water. There is no relation between the emission limits and the ecological status of the receiving water.

Conventional biological assessment methods measure the integrated effect of all impacts on a receiving water over an elongated period of time (Ellis, 2000). The time span which is covered depends on the mean life time of the indicator organisms, e.g. indices based on macroinvertebrates usually integrate several months, whereas fish based indicators reflect up to several years. Exclusively biological measures permit the assessment of 'biological integrity' (aim of the CWA (1972)), or 'ecological quality' (goal of the European Water Framework Directive (2000/60/EC)). However, measurements with biological indicators allow only post-impact assessment (Ellis, 2000). Further it is difficult to develop robust indices which can be applied over a larger spatial scale, because aquatic biocoenosis differ widely from each other on very small regional scales. The more advanced a biological assessment method is, the higher specialised experts are needed for their application (Simon, 2000). Further biological measurements are time consuming and thus expensive (Ellis, 2000). Aquatic ecosystems are highly complex systems and the interactions between their different components are not completely understood (Wetzel, 2001). Therefore it is usually not possible to identify the cause for impairment measured with biological indices (Ellis, 2000). Efforts are currently made to define biological indices which allow the identification of specific causes (see for example the European project AQEM (Hering et al., 2004), or Novotny et al. (2005)). However, the advantage that biological measures integrate various stressors is also the major drawback, as it hinders the clear identification of impacts. Currently no biological assessment method suitable to evaluate the impacts from intermittent duscharges is available. For the application in wet-weather flow management, biological indicators would be necessary which are clearly correlated to the different types of impact and are able to assess acute impacts.

Toxicity testing allows the integrated assessment of the toxicity of all pollutants contained in an effluent, including interactions of toxic effects (as additive and synergistic effects). Therefore these methods can predict biological impact of complex mixtures of pollutants. From toxicity tests numerical limits can be derived. However, these tests do not describe the response of the ecosystem (e.g. the indirect effects on other species) (Ellis, 2000), nor the fate of the toxics in the environment (like transformation, sorption, or bioaccumulation). Further the results are obtained with a narrow range of test organisms (Whitehouse, 2001), and as the toxicity varies for different species, limits defined from this data may not be sufficient to protect specifically important species (like endangered species). The pollutants causing the toxicity are also not identified with toxicity testing. The high variability of wet-weather discharges can also not be assessed. Toxicity testing is still under development, and currently it is not applicable for wet-weather control design and planning (Marsalek et al., 1999b).

The renunciation from strict emission approaches will introduce considerable uncertainty and complexity to urban drainage planning. The cooperation of engineers with natural scientist will be essential in future (Krejci and Kreikenbaum, 2004). From such cooperation, the ambient water quality based guidelines for intermittent impacts have been created. These guidelines specify environmental quality standards for different important impacts known to be caused by wet-weather flows. These comprise usually acute toxic impacts from un-ionised ammonia and oxygen deficits, and acute hydraulic impacts (here limits are either set for maximal permissible discharge or erosion frequency). Some guidelines also include acute limits for further pollutants (e.g. heavy metals), temperature changes or suspended solids/turbidity. These numerical standards can be used to evaluate the results of computer simulations of the integrated urban drainage system. However, there are several uncertainties connected with environmental quality standards. First, the limits are extrapolated from toxicological data of a restricted number of species and it is unclear if the included safety factors are too tight or too stringent (Whitehouse, 2001). It is also not proofed that scaling of the toxicological data to different exposure times is valid (e.g. from a LC_{50}(96h) value to a LC_{50}(1h)) (Ellis, 2000). This, however, is usually done to calculate limits. The transferability of toxicological data obtained under laboratory conditions to natural conditions introduces additional uncertainties (Borchardt, 1992). Further the methodology used to define the limits does not account for the impacts from delayed uptake, bioaccumulation or sediment related chronic toxic impact (Ellis, 2000). It has been recognised that pollutant interactions and frequency of exposure can significantly influence the impact on the receiving water (Burton and Pitt, 2002;FWR, 1998). However, there is insufficient data and therefore today only for very few pollutants (e.g. un-ionised ammonia and oxygen deficit), limits are defined which consider interactions and recovery time. Thus the nature and level of risk cannot be assessed with the current ambient water quality approaches (Ellis, 2000). The application of these approaches is complex and requires adequate tools as computer programs (Rauch et al., 2005).

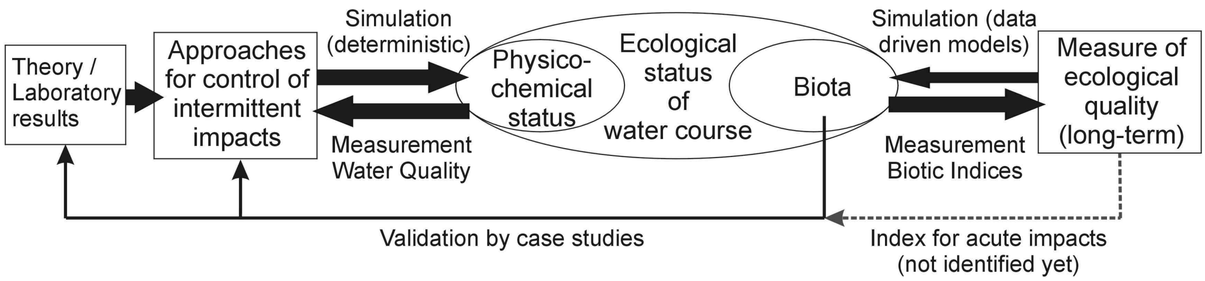

Figure 7: Current status of intermittent impact assessment.

Generally, the benefit of both emission and ambient water quality based approaches is not stringent and scientifically sound evaluated. Both are derived from theory and laboratory results, and only few case studies have been performed to evaluate the approaches. The reasons are the difficulty in assessment of the impact of wet-weather flows and the multitude of influencing factors, whereat stormwater is only one of numerous impacts on receiving waters (Merz and Gujer, 1997). Scientifically sound assessment needs investigation of the integrated urban drainage system and thus requires huge amount of monitoring efforts (Vanrolleghem et al., 1999).

How to proceed:

1. Emission based methods are well established. Although a direct and definite benefit has not been proofed, this approach will be pursued further due to safety considerations (precautionary principle).
2. Ambient water quality based methods aim at a direct cause-effect mechanism, however, because of its high uncertainty (i.e. numerous influencing factors) also this approach is problematic. As for the emission based approaches, the scientifically sound evaluation of the efficiency of the ambient water quality based methods is lacking. Additionally it is unclear if the parameters usually utilised in these methods (e.g. dissolved oxygen, un-ionised ammonia) are suitable for an adequate description of the biocoenosis of running waters. However, it is to be expected that this approach – preferably in combination with emission based methods – will further be applied and improved.
3. Biological indicators and toxicity tests are generally not adequate for urban drainage planning as both methods are based upon monitoring. This means that it is possible to assess the ecological impact only after the implementation of a measure. However, these methods will probably be improved further and become of importance for urban water management in combination with emission and ambient water quality based methods.

CONCLUSIONS

Emission based criteria for urban wet-weather discharges can reduce the impact on the receiving water significantly, but cannot guarantee a good receiving water ecological quality. Biological measurement methods can represent the ecological status of a watercourse, but as the cause-effect relations between changes of the aquatic biocoenosis and physicochemical water parameters are not known today, the results of the biological measurements cannot be used to identify problems caused by urban wet-weather discharges. Collaborations between engineers and limnologists have resulted in different ambient water quality based approaches to assess the impact of urban wet-weather discharges. These approaches are usually based on chemical or physical specific limits. The computer simulation tools, needed for the application of these approaches, exist today. However, pollutant interactions can easily be overlooked, sub-lethal effects are not always described, and scaling of toxicity data to different durations has insufficient scientific basis.

Currently the combination of computer simulations of the integrated urban drainage system and ambient water quality approaches seem to be the best solution to deal with receiving water quality in wet-weather flow management. It allows the identification of certain impacts already in the planning stage, therefore this combination will probably further be established. However, this approach is complex and requires detailed data of the urban drainage system and its receiving waters. Therefore a combined approach seems to be the most feasible, whereat the urban drainage system is generally designed with emission based standards, and ambient water quality based approaches are only applied, if problems with receiving water quality are detected.

Although the development of ambient water quality based approaches to assess the impacts of urban wet-weather discharges is an important step to harmonise urban drainage planning procedures with the aims of surface water quality legislation, there is still a considerable need to identify cause-effect relations between ecological parameters and chemical water quality to establish ecological goal functions for urban drainage planning. For the future, easily applicable biological indicators should be developed which allow the evaluation of the performance of ambient water quality approach derived management strategies. This would facilitate both, compliance verification of existing structures, and improvement of ambient water quality based approaches (especially to adapt the limits to site-specific requirements).

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ANNEX VI

SUITABILITY OF CSO PERFORMANCE INDICATORS FOR COMPLIANCE WITH AMBIENT WATER QUALITY TARGETS

Engelhard C., De Toffol S. and Rauch W. (2008). Indicators for the CSO performance for compliance with ambient water quality targets. *Urban Water Journal*, **5** (1), 43-49.

ANNEX VII

MITIGATION MEASURES TOWARDS MORPHOLOGICAL ALTERATIONS OF RIVERS: THE RECEIVING WATER AS PART OF THE INTEGRATED WASTEWATER SYSTEM

Engelhard C., Achleitner S., Lek I. and Rauch W. (2006). Mitigation measures towards morphological alterations of rivers: The receiving water as part of the integrated wastewater system. *Water Practice & Technology*. **1** (1). doi10.2166/wpt.2006.013

ANNEX VIII

COMBINED SEWER SYSTEM VERSUS SEPARATE SYSTEM - A COMPARISON OF ECOLOGICAL AND ECONOMICAL PERFORMANCE INDICATORS

De Toffol S., Engelhard C. and Rauch W. (2007). Combined sewer system versus separate system – a comparison of ecological and economical performance indicators. *Water Science & Technology*, **55** (4), pp. 255–264.

ANNEX IX

SUMMARY OF DIFFERENT

WATER QUALITY STANDARDS

ANNEX IX: WATER QUALITY STANDARDS

1. Different long-term standards for nitrite

Table IX-1: Different long-term standards for nitrite (NO_2).

	Salmonid waters mg/l	Cyprinid waters mg/l	Source
Nitrites (NO_2)	≤ 0.01 $^{(G)}$ ≈ 0.003 mg NO_2-N/l	≤ 0.03 $^{(G)}$ ≈ 0.009 mg NO_2-N/l	78/659/EEC
Nitrite (as NO_2-N)	0-3 mg Cl^-/l: 3-7.5 mg Cl^-/l: 7.5-15mg Cl^-/l: 15-30 mg Cl^-/l: >30 mg Cl-/l:	0.01 mg/l 0.05 mg/l 0.09 mg/l 0.12 mg/l 0.15 mg/l	(Wimmer *et al*.,2003)
Nitrite	highland watercourse 0.03 ≈ 0.009 mg NO_2-N/l	lowland watercourse 0.06 ≈ 0.018 mg NO_2-N/l	(AImVF, Draft 1995)
Nitrite-N (90percentile) < 10 mg Cl^-/l > 10 mg Cl^-/l	0.03 0.20	0.06 0.40	(ATV, 1994)
sensible water with <10mg Cl^-/l	0.05		(Orth *et al*., 2003)

(G) Guide value

2. Different long-term standards for un-ionised ammonia

Table IX-2: Different long-term standards for un-ionised ammonia (NH_3).

	Salmonid waters mg/l	Cyprinid waters mg/l	Source
Non-ionized ammonia (NH_3)	≤ 0.025 $^{(M)}$	≤ 0.025 $^{(M)}$	(78/659/EEC)
Un-ionised ammonia (as NH_3)	0.01		(Wimmer *et al*., 2003)
Un-ionised ammonia	highland watercourse 0.02	lowland watercourse 0.02	(AImVF, Draft 1995)
Simplified procedure	< 0.1		(BWK, 2001)
Un-ionised ammonia 99 percentile (mgN/l)	0.04 mg NH_4-N/l = 0.051 mg NH_4/l		(FWR, 1998)
	0.082		(Schwoerbel, 1999)
Un-ionised ammonia	0.002		(Orth *et al*., 2003)

(M) Mandatory

3. Different long-term standards for dissolved oxygen

Table IX-3: Different long-term standards for dissolved oxygen (DO).

	Salmonid waters mg/l	Cyprinid waters mg/l	Source
	50% ≥ 9 mg/l (M) 100% ≥ 7 mg/l (G)	50% ≥ 7 mg/l (M) 100% ≥ 5 mg/l (G)	(78/659/EEC)
	highland waters >80%; <125%; > 7,5 mg/L	lowland waters >80%; > 6,5 mg/ L	(AImVF, Draft 1995)
	80 – 100% (G)		(76/160/EEC)
Sensitive salmonid water	> 8 mg/l		(Orth *et al*., 2003)

(M) Mandatory, (G) Guide value

4. Different long-term standards for total phosphorus

Table IX-4: Different long-term standards for total phosphorus (TP).

		Salmonid waters mg/l	Cyprinid waters mg/l	Source
TP (mg/l PO_4)		(0.2) (G) (≈ 0.065 mg P/l)	(0.4) (G) (≈ 0.13 mg P/l)	(78/659/EC)
TP, dissolved		highland watercourse 0.07	lowland watercourse 0.15	(AImVF, Draft 1995)
TP (values for different ecoregions)		0.01 – 0.07	(0.13)	(US EPA, 2004)
	Mean:	0.034	(0.041)	
	Max:	0.076	(0.128)	
	Min:	0.01	(0.010)	
	Median:	0.032	(0.033)	

(G) Guide value

5. Different long-term standards for copper

Table IX-5: Different long-term standards for copper (Cu).

	Salmonid waters µg/l	Cyprinid waters µg/l	Source
<50 mg $CaCO_3$/l	1.6 = 1.1 [mc] + 0.5 [nb]		
50 – 100 mg $CaCO_3$/l	5.3= 4.8 [mc] + 0.5 [nb]		(Wimmer *et al.*, 2003)
>100 mg $CaCO_3$/l	9.3= 8.8 [mc] + 0.5 [nb]		
10 mg $CaCO_3$/l [dm]	5 (G)		
50 mg $CaCO_3$/l [dm]	22 (G)		(78/659/EEC)
100 mg $CaCO_3$/l [dm]	40 (G)		
500 mg $CaCO_3$/l [dm]	112 (G)		
	highland waters	lowland waters	
< 300 mg $CaCO_3$/l	1	10	(AImVF, Draft 1995)
> 300 mg $CaCO_3$/l	6	30	
CMC [dm, hd]	13		(US EPA, 2002)
CCC [dm, hd]	9		
A1, A2, A3	50		(75/440/EEC)

CMC: Criteria Maximum Concentration
CCC: Criterion Continuous Concentration
[mc] Maximum concentration
[nb] Natural background concentration
[hd] Hardness dependent, limit stated is for 100mg/l hardness (calculation see table below)
[dm] Limit for dissolved metal concentration in the water column
(M) Mandatory
Added risk limit: maximum acceptable concentration [mc] + natural background concentration [nb]

Calculation of water quality criteria by US EPA (2002):

	Conversion Factor freshwater CMC	Conversion Factor freshwater CCC	Hardness dependency m_A	b_A	m_C	b_C
Copper	0.960	0.960	0.9422	-1.7	0.8545	-1.702

Hardness-dependant metals' criteria may be calculated from the following:

$$CMC(dissolved) = e^{mA(\ln(hardness))+bA} \quad (CF)$$

$$CCC(dissolved) = e^{mC(\ln(hardness))+bC} \quad (CF)$$

6. Different long-term standards for zinc

Table IX-6: Different long-term standards for Zinc (Zn).

	Salmonid waters µg/l	Cyprinid waters µg/l	Source
< 50 mg $CaCO_3$/l	9.6 = 8.6 [mc] + 1.0 [nb]		(Wimmer *et al.*, 2003)
50 – 100 mg $CaCO_3$/l	21= 20 [mc] + 1.0 [nb]		
> 100 mg $CaCO_3$/l	29.6= 28.6 [mc] + 1.0 [nb]		
10 mg $CaCO_3$/l	30 [(M)]	300 [(M)]	(78/659/EEC)(AImVF, Draft 1995)
50 mg $CaCO_3$/l	200 [(M)]	700 [(M)]	
100 mg $CaCO_3$/l	300 [(M)]	1000 [(M)]	
500 mg $CaCO_3$/l	500 [(M)]	2000 [(M)]	
	highland waters	lowland waters	
< 300 mg $CaCO_3$/l	15	70	(AImVF, Draft 1995)
> 300 mg $CaCO_3$/l	60	180	
CMC [dm, hd]	120		(US EPA, 2002)
CCC [dm, hd]	120		
A1	3000		(75/440/EEC)
A2, A3	5000		

CMC: Criteria Maximum Concentration
CCC: Criterion Continuous Concentration
[mc] Maximum concentration
[nb] Natural background concentration
[hd] Hardness dependent, limit stated is for 100mg/l hardness (calculation see table below)
[dm] Limit for dissolved metal concentration in the water column
[(M)] Mandatory
Added risk limit: maximum acceptable concentration [mc] + natural background concentration [nb]

Calculation of water quality criteria by US EPA (2002):

	Conversion Factor freshwater CMC	Conversion Factor freshwater CCC	Hardness dependency m_A	b_A	m_C	b_C
Zinc	0.978	0.986	0.8473	0.884	0.8473	0.884

Hardness-dependant metals' criteria may be calculated from the following:

$$CMC(dissolved) = e^{mA(\ln(hardness))+bA} \quad (CF)$$

$$CCC(dissolved) = e^{mC(\ln(hardness))+bC} \quad (CF)$$

7. Different long-term standards for cadmium

Table IX-7: Different limits found for cadmium (Cd).

	Salmonid waters µg/l	Cyprinid waters µg/l	Source
	1		(Wimmer *et al.*, 2003)
	highland waters	lowland waters	
< 300 mg $CaCO_3$/l	0.1	1	(AImVF, Draft 1995)
> 300 mg $CaCO_3$/l	0.5	2	
	1		(LAWA, 2003)
in coastal waters	0.5		
CMC dm, hd	2.0		(US EPA, 2002)
CCC dm, hd	0.25		
A1, A2, A3	5		(75/440/EEC)

CMC: Criteria Maximum Concentration
CCC: Criterion Continuous Concentration
hd - Hardness dependent, limit stated is for 100mg/l hardness (calculation see table below)
dm - Limit for dissolved metal concentration in the water column

Calculation of water quality criteria by US EPA (2002):

	Conversion Factor freshwater CMC	Conversion Factor freshwater CCC	Hardness dependency m_A	b_A	m_C	b_C
Cadmium	1.136672-[(ln hardness)(0.041838)]	1.101672-[(ln hardness)(0.041838)]	1.0166	-3.924	0.7409	-4.719

Hardness-dependant metals' criteria may be calculated from the following:

$$CMC(dissolved) = e^{mA(\ln(hardness))+bA} \quad (CF)$$

$$CCC(dissolved) = e^{mC(\ln(hardness))+bC} \quad (CF)$$