# Influence of leaky sewer systems on groundwater resources beneath the City of Rastatt, Germany

# Grundwasserbeeinflussung durch defekte Abwasserkanäle im Gebiet der Stadt Rastatt

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# ABSTRACT

Leakage from defective sewer systems is an omnipresent phenomenon in urban areas but its quantification is associated with high uncertainty. Several studies have identified leaky sewer systems as the dominating contaminant input to urban aquifers (Eiswirth & Hötzl, 1997; Härig & Mull, 1992). On the other hand, contemporary literature also reports the dominating influence of the self-sealing effect which leads to a low risk potential (Blackwood *et al.*, 2005b). Furthermore, attempts to directly measure wastewater impact on groundwater quality have been unsuccessful in several experiments in Germany (Eiswirth *et al.*, 2002a; Hagendorf, 2004) and the results from most other case studies worldwide are ambiguous. It has been the aim of this study to conduct a detailed assessment of the impacts of wastewater exfiltration on the groundwater in a sand and gravel aquifer beneath a city with a detailed sewer condition dataset and a dense network of observation wells. Methods of choice include numerical transport modelling and hydrochemical sampling programs with pharmaceuticals as marker species. Based on this information, the potential, limitations and the uncertainties of the upscaling of laboratory experiments to the city scale are explored. As a case study, the city of Rastatt in Southwestern Germany was chosen. Rastatt has approx. 46000 inhabitants and is located 30km south of Karlsruhe in the Upper Rhine Graben. A reconnaisance mapping of the urban geology and the intersection of sewer information with the thickness of clayey cover layers revealed that most of the sewer network is in direct contact with the Upper Gravel Layer aquifer which is also used for local drinking water supply.

Strong hydrochemical evidence for exfiltration from leaky sewers was found within the experimental field investigations. The placement of 12 new focus observation wells in the direct vicinity of defective sewers with help of the geographically referenced sewer defect database provided opportunities for direct observation of sewage-influenced groundwater. The findings in these focus observation wells were compared to the urban background, the urban fringe and the rural areas. The focus observation wells exhibit significantly elevated values for the parameters ammonium, potassium, sodium, chloride, boron, EDTA, specific electrical conductivity, temperature and phosphate. Elevated concentrations close to leakages were not found for heavy metals and nitrate. The marked contrast between the sewer focus group and the urban background group shows that the obvious quality deterioration is limited to the immediate surrounding of defects rather than the whole network. Among the pharmaceutical substances screened, iodated x-ray contrast media, metoprolol, sotalol, carbamazepine, clofibric acid, bezafibrate, diclofenac und ibuprofene were detected in groundwater samples. The potential of iodated x-ray contrast media as a marker substance was explored in detail. 114 samples from 46 wells were analysed. Positive detects occurred for amidotrizoic acid (30), iothalaminic acid (10), ioxithalaminic acid (10), iopamidol (4), iohexol (4), iomeprol (3), iopromide (2). The microbiological sampling programme comprised *Escherichia coli*, total coliforms, enterococci, faecal streptococci, sulphate reducing clostridia, coliphage and *Pseudomonas aeroginosa* and indicated recent impacts on groundwater quality. Online probes in the focus observation wells which monitored physico-chemical characteristics in high time resolution detected fast responses in the specific electrical conductivity of groundwater in the course of rain events. This proved that the sewers are in close contact with the aquifer and travel times through the unsaturated zone are short (sometimes less than one hour). Monthly samplings at the new monitoring site Danziger Strasse showed elevated concentrations of ammonium and chloride downstream of a leaking sewer. However, the upstream water also seemed to be already influenced by a leaky house connection.

A new method was described for the upscaling of the laboratory results to calculate the amount of sewage exfiltration for the entire sewer network of Rastatt. Besides the defect area known from the visual inspection, the proposed equation takes into account the different exfiltration characteristics during dry weather flow and extreme rain events. Furthermore it considers the water level in the sewer, the wetted proportion of the leak area, the thickness of the colmation layer, the percentage of sewers beneath the groundwater table and the hydraulic conductivity of the colmation layer. The uncertainties in the input parameters were implemented by probability density functions and processed in a Monte Carlo simulation. This enabled to explore the contribution of individual uncertainties to the variance of the overall result. Regarding the boron load to the soil-aquifer system, the uncertanties in the hydraulic conductivity of the clogging layer during dry weather flow exert the strongest influence on the result. Overall, the forecast returned a mean leakage rate of 0.99 mm/a related to the entire city area. The minimum is at 0.02 mm/a and the maximum at 10.38 mm/a. With a probability of 95% the leakage rate will be below 2.24 mm/a. The described upscaling procedure constitutes an easy- to-use alternative to the more sophisticated NEIMO software developed in the course of the AISUWRS project and allows for an assessment of the uncertainty.

Finally, a numerical groundwater flow and transport model was set up to allow for a comparison between measured concentrations and concentrations derived from the forward modelling using the described Monte Carlo simulation approach. The modelling focuses on boron as marker species. The modelling exercises confirm that the impact is limited to the close vicinity (10-50 m) of the leaks. The comparison between measured and modelled concentrations reveals a boron concentration that is slightly too low on average and significant deviations at single wells. This demonstrates the limitation of the applied upscaling procedures. The lacking reliability of CCTV inspections, the unknown distribution of defective private house connections, the existence of other boron sources than sewers and problems in the mesh discretisation were identified as causes for the lack of correlation between measured and modelled concentrations. As an additional checking, a conservative mass balance was set up using the lateral and vertical water flows determined by the numerical flow model. According to this simplified balance approach, an additional input of 53 mm/a sewage into the aquifer is necessary to reach the measured boron concentrations. At last, the findings of the Rastatt case study were compared with statements of 24 international literature sources on sewer leakage rates. Related to the sewer network in Rastatt, the estimates would range from 0.2 mm/a to 1001 mm/a sewer leakage from public sewer networks.

As a summary, it can be stated that consistent evidence was found that significant amounts of wastewater are released to the groundwater beneath the city of Rastatt. The self-sealing effect observed at the laboratory scale does not prevent a qualitative deterioration of the urban groundwater due to sewer leakage. A good knowledge of the groundwater users in the city, which might be both formal and informal, is required to assess the spatial separation of contamination sources and water withdrawals. A more prominent inclusion of environmental factors into the priorisation of sewer rehabilitation works is recommended.

# KURZFASSUNG

Defekte Abwasserkanäle und Abwasserversickerungen sind ein allgegenwärtiges Phänomen in urbanen Räumen, doch die Abschätzung der austretenden Stofffrachten und der resultierenden Beeinflussung des urbanen Grundwassers ist mit großen Unsicherheiten behaftet. Zum einen identifizierten mehrere Studien defekte Abwasserkanäle als Haupteintragspfad von Schadstoffen in urbane Grundwasserleiter (Eiswirth & Hötzl, 1997; Härig & Mull, 1992). Dem entgegen stehen Studien die von einer weitgehenden Selbstabdichtung des Kanalnetzes durch Kolmationsprozesse ausgehen (Blackwood *et al.*, 2005b). Zusätzlich konnten in speziell eingerichteten in situ Versuchsanordnungen (Eiswirth *et al.*, 2002a; Hagendorf, 2004) keine nennenswerten Veränderungen im Grundwasserchemismus festgestellt werden. Ziel der vorliegenden Arbeit ist es daher, an einem leicht zugänglichen Porenaquifer unterhalb einer Stadt mit gut dokumentierter Kanalzustandserfassung und dichtem Messnetz die Beeinflussung des Grundwassers durch defekte Abwasserkanäle mittels hydrochemischer Beprobungen und numerischen Modellrechnungen detailliert zu überprüfen. Dabei werden Möglichkeiten, Unsicherheiten und Beschränkungen der Hochrechnung von Laborversuchen zur Exfiltration auf ein Kanalnetz im Stadtgebietsmaßstab beleuchtet.

Als Untersuchungsobjekt diente die Stadt Rastatt mit ca. 46 000 Einwohnern, die 30 km südlich von Karlsruhe im Oberrheingraben liegt. Durch eine Verschnittoperation von Kanalpositionen und der Mächtigkeit undurchlässiger Deckschichten konnte gezeigt werden, dass der überwiegende Teil des Kanalnetzes in direktem Kontakt mit dem zur lokalen Trinkwasserversorgung genutzten Grundwasserleiter steht. Unter Berücksichtigung der vorhandenen TV-Kanalzustandserfassung wurden 12 Fokus-Messstellen in der unmittelbaren Nähe von Kanalleckagen errichtet. Positive Befunde im Grundwasser dieser Messstellen wurden zum einen mit einem Messstellennetz verglichen, das den urbanen Hintergrund darstellt, sowie zum anderen mit Referenzmessstellen außerhalb des Stadtgebietes. Deutlich gegenüber dem Hintergrund erhöhte Durchschnittswerte der Gruppe der Fokus-Messstellen ergaben sich insbesondere für Ammonium, Kalium, Natrium, Chlorid sowie Bor und EDTA. Im Grundwasser wurden für Nitrat sowie die Schwermetalle Kupfer, Blei und Zink keine auffällig erhöhten Befunde im Umfeld von Kanalleckagen festgestellt. Bei den untersuchten Pharmaka ergaben sich die häufigsten Nachweise im Grundwasser für iodierte Röntgenkontrastmittel (insb. Amidotrizoesäure), Metoprolol, Sotalol, Carbamazepin, Clofibrinsäure, Bezafibrat, Diclofenac und Ibuprofen. Die Belastungen durch Pharmaka variieren dabei sowohl zeitlich als auch räumlich stark. Die Eignung iodierter Röntgen-

kontrastmittel als Abwassertracer konnte aufzeigt werden. Aufgrund der Heterogenität der Konzentrationen im Kanalnetz war eine Bilanzierung jedoch nicht möglich. Mehrere Beprobungen auf mikrobiologische Indikatoren wie *E.Coli*, Enterokokken, Coliphagen und sulfitreduzierende Clostridien ergaben eine deutliche Keimbelastung des urbanen Grundwassers. Zeitlich hoch aufgelöste Messungen der spezifischen elektrischen Leitfähigkeit in den Fokus-Messstellen zeigten Reaktionszeiten von weniger als einer Stunde auf Niederschlagsereignisse und verdeutlichen somit die kurzen Verweilzeiten und das beschränkte Rückhaltevermögen der ungesättigen Zone bei Starkregenereignissen. Monatliche Untersuchungen an 4 Messstellen der Detailmessstrecke Danziger Strasse zeigten deutliche Konzentrationserhöhungen von Ammonium und Chlorid im Abstrom eines Abwasserkanals, jedoch auch Hinweise auf eine Vorbelastung des Grundwasserzustroms durch private Grundstücksentwässerungen.

Zur Hochrechnung der Abwasseraustrittsmengen auf den Stadtmaßstab wurde ein vereinfachter Formelzusammenhang aufgestellt in dem wasserbenetzte Schadensfläche, Kanalfüllstand, Grundwasserstand, Häufigkeit von Starkregenereignissen sowie die hydraulische Durchlässigkeit der Kolmationschicht berücksichtigt werden. Die bestehenden Unsicherheiten bei der Bestimmung der Eingangsparameter wurden als Wahrscheinlichkeitsfunktionen abgebildet und mittels einer Monte Carlo-Analyse in ihrer Wirkung auf das Berechnungsergebnis quantifiziert. Für den Stoffaustrag stellt dabei die hydraulische Durchlässigkeit der Kolmationsschicht während Trockenwetterperioden den einflussreichsten Parameter dar. Die Prognose ergibt eine mittlere Exfiltrationsrate von 0.99 mm/a für das öffentliche Kanalnetz in Rastatt bezogen auf das Stadtgebiet. Mit einer Wahrscheinlichkeit von 95 % ist die Exfiltrationsrate kleiner als 2.24 mm/a. Das Maximum der Prognose liegt bei 10.38 mm/a. Die beschriebene Berechnungsmethode unter Verwendung von Monte Carlo-Simulationstechniken stellt eine effiziente Alternative zu dem im Rahmen des AISUWRS Projektes entwickelten Prognosewerkzeug NEIMO dar und erweitert die Aussage um eine Beurteilung der Unsicherheit.

Die durchgeführten numerischen Modellrechnungen unter der Verwendung von finiten Elementen zeigen die Kleinräumigkeit der Grundwasserbeeinflussung durch einzelne Kanalleckagen. Bei Ansatz der Exfiltrationsmengen aus dem Monte Carlo-Ansatz ergibt sich eine im Schnitt zu niedrige modellierte Bor-Konzentration im Grundwasser im Vergleich mit den hydrochemischen Messungen. Hierin werden die Beschränkungen der durchgeführten Vorwärtsmodellierungen sowie Hochrechnungen von Exfiltrationsmengen im Allgemeinen deutlich. Diese sind gegeben durch die unbekannten Leckagen privater Grundstücksentwässerungen, die Annahme einer homogenen Hintergrundkonzentration sowie die mangelnde Aussagekraft der visuellen Kanalinspektion in Bezug auf die tatsächliche Dichtigkeit des Kanals. Die im Grundwassermodell bestimmten lateralen und vertikalen Wasserflüsse wurden für eine konservative Stoffbilanzierung der Parameter Bor und Chlorid angewendet. Demnach ist ein Eintrag von 53 mm/a Abwasser mit einer Konzentration von 0.45 mg/l Bor durch defekte Kanalsysteme in das Grundwasser nötig. Zur abschließenden Einordnung der Ergebnisse wurden 24 in der Literatur dokumentierte Aussagen zu Exfiltrationsraten auf das Stadtgebiet Rastatt übertragen. Dabei ergab sich eine Bandbreite von 0.2 mm/a bis zu 1001 mm/a.

Während auch im Rahmen der Untersuchungen in Rastatt die Aussageunsicherheiten nicht vollständig ausgeräumt werden konnten, wurden jedoch wichtige Beiträge zum Prozessverständnis geliefert. Zusammenfassend lässt sich feststellen, dass die im Labormaßstab festgestellten Kolmations- und Selbstabdichtungsprozesse im Stadtgebietsmaßstab nicht ausreichend wirksam sind um eine messbare Beeinträchtigung der Grundwasserqualität zu verhindern. Überschreitungen von Grenzwerten der Trinkwasserverordnung wurden jedoch nur für mikrobiologische Indikatoren festgestellt. Der Eintrag fäkaler Verunreinigungen durch Kanalleckagen ins Grundwasser muss daher auch in Zukunft noch stärker bei der Ausweisung von Schutzgebieten berücksichtigt werden. Einer ausreichenden Trennung zwischen Grundwassernutzung (auch private, nicht angemeldete) und defekten Abwassersystemen als Kontaminationsquelle ist Rechnung zu tragen. Eine stärkere Gewichtung und Überarbeitung der umweltrelevanten Faktoren in den Verfahren zur Sanierungspriorisierung könnte hier zusätzliche Sicherheiten schaffen.

## DEDICATION

This thesis is dedicated to my former supervisor PD Dr. Matthias Eiswirth who died together with his son Lennert in a tragic accident on 30/12/2003. Without his many initiatives, his strong dedication to the field of urban hydrogeology and without his support, none of this work would have come into being. Even after his death, his ideas and the memory of him provided strong guidance. But besides his scientific achievements he will always remain known as somebody who cared about the people around him and who provided support wherever possible and appropriate. Without hesitation he took on many responsibilities within his job and his private life, much to the benefit of his fellows. I remain thankful for all the time we shared.

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The key role of PD Dr Matthias Eiswirth in the preparation of this thesis is expressed in the separate dedication on the previous page.

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I am indebted to the members of the AISUWRS research team who provided strong support and personal warmth to me in difficult times. Representative for many, I would like to name Mr. Brian Morris (BGS, UK) for arranging ICP-MS analyses and many discussions; Mr. Stewart Burn & Dr. Dhammika DeSilva (CSIRO, Australia) for common papers and developments, Dr. Joerg Rueedi (Robens Institute, UK) for his focus on content and reviewing of this thesis; Dr. Aidan Cronin (Robens Institute, UK) for arranging microbiological analysis. Also, the major financial support of the European Union is gratefully acknowledged.

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I would like to post a special thanks to the institutions and persons who granted me access to important data sources concerning the city of Rastatt. Especially helpful were Mr. Jürgen Kramp, Mr. Peter Polak and all the other friendly staff from the Department of Civil Engineering. Also, Mr. Volker Tropf from the Umweltamt Rastatt was a constant source of good information. Thanks are also due to the Landesanstalt für Umweltschutz (LfU) fort the access to their very well maintained databases.

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Finally, I would like to express my gratitude to my parents for offering me all their love and wisdom. Most of all, my wife Vera and my daughter Mia were a constant source of joy and understanding, always providing me with the strength to carry on.

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# 1 Introduction

## 1.1 Problem Outline

Leakage from defect sewers is believed to play a major role in the urban water and solute balance, especially with regard to a possible continuous contamination from permanently installed and widespread systems with a complex mixture of substances (Barret *et al.*, 1999; Dohmann *et al.*, 1999; Eiswirth & Hötzl, 1997; Ellis, 2001; Gallert *et al.*, 2001; Sacher *et al.*, 2002). The European Union standard EN 752-2 recognises this problem and therefore demands the structural integrity of urban sewer systems including their water-tightness (Keitz, 2002). German regulations also request the water-tightness of the sewer system, a request which is impossible to fulfil in practice given the financial constraints of urban communities. Despite the significant research work undertaken during the last two decades, the quantification of sewer leakage and the associated risk potential remain uncertain. The main difficulties are the upscaling of laboratory results, problems with multiple sources of employed marker species and heterogeneous hydrogeological boundary conditions.

The total length of the German sewer system amounts to ca. 486 000 km in the year 2001 (Statistisches Bundesamt, 2003). According to the latest national German survey which is based on CCTV inspections, 8.8 % (42 800 km) require immediate rehabilitation while another 10.8 % (52 500 km) require short term action (Berger & Lohaus, 2005). The knowledge of this obviously high number of leaky sewers gave rise to concerns about frequent and widespread detrimental impacts on soil and groundwater. Several large research initiatives were started at national (Dohmann *et al.*, 1999; Hagendorf, 1996; Härig & Mull, 1992) and international level (Ellis *et al.*, 2003; Rauch & Stegner, 1994; Vollertsen & Hvitved-Jacobsen, 2003). Due to the complexity of the involved processes and the limited information about the actual sewer condition, only limited attempts were done to employ physically based models on a real world scale. Existing calculations exhibit a high degree of uncertainty (typically only constrained by worst case and best case scenarios) and have not been compared with measurements of the impact on groundwater quality. Direct evidence of groundwater contamination has been found very scarcely and some authors concluded that they have not been observed at all (Hagendorf 2004).

Recent events documenting the importance of sewer leakage for drinking water quality include the spreading of Giardia lamblia in the drinking water in

Bergen, Norway during September-October 2004 (Hallström, 2005). Leaky sewers upstream of a sewers water reservoir have been identified as the most likely source for the bacteriological contamination. Rapid transport of wastewater components into the lake was fostered by coarse grained sediments in the sewer surrounding and heavy rainfalls. About 1700 people were affected by the sickness, but no fatal cases have been reported.

Furthermore, examples for serious water quality deterioration in connection with excessive pipe leakage have been documented in the UK (Powell *et al.*, 2003). Leaking sewers have caused public water supply contamination and associated gastric illnesses in Britain and Ireland (Misstear *et al.,*1996).

Another likely example of sewage derived contamination of drinking water has been reported from a water work near Cologne, where E.coli was repeatedly detected in water supply wells. The distance to the next settlement was less than 500 meter (Treskatis, 2003).

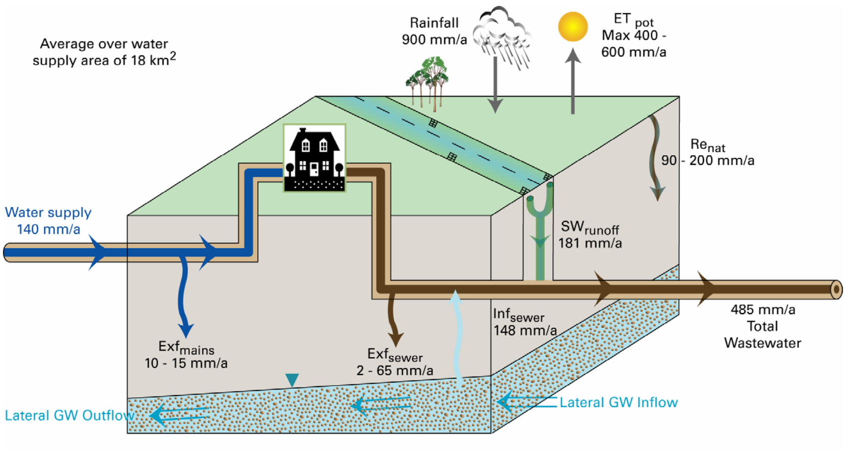



Fig. 1-1: Overview of urban water fluxes, estimated for the entire City of Rastatt, SW-Germany (ET$_{pot}$ = potential evapotranspiration, Re$_{nat}$ = natural groundwater recharge, SW$_{runoff}$ = stormwater runoff into sewers, Inf$_{sewer}$ = Infiltration of groundwater into sewers, Exf$_{sewer}$ = estimated exfiltration of wastewater from leaky sewers, Exf$_{mains}$ = leakage from drinking water mains).

While the problem of sewer leakage is now widely known, major difficulties still exist in the quantification of the volumes exfiltrating from leaky sewers and sometimes contradicting statements can be found in the literature. Even though large exfiltration volumes were predicted for the suburb of Rastatt-Plittersdorf (Eiswirth & Hötzl, 1997), no influence of a leaky sewer on groundwater observation wells within 2 m distance to the leak was detected at a test site built in Rastatt-Kastanienweg (Eiswirth *et al.*, 2002a). Furthermore, estimations of the associated risk to urban groundwater resources are only sparsely available. Hagendorf (2004) concludes in his study that the sewer asset databases of all cities have to be updated with regard to ecological aspects in order to allow an appropriate evaluation of the need for rehabilitation works.

The city of Rastatt has served as a case study city in several research projects related to defect sewers in recent years and relatively dense information is available. It is the purpose of this study to document a detailed assessment of the impact of leaking sewers on the groundwater in Rastatt and to propose a methodology for leakage impact assessment in other urban areas.

## 1.2 Objectives and scope of this study

Even though sewer leakage is a widely recognized problem, laboratory studies suggest a dominating influence of the self-sealing effect which leads to a low risk potential. Furthermore, attempts to directly measure wastewater impact on groundwater quality have been unsuccessful in several experiments in Germany and results from most other case studies worldwide are ambiguous. It has been the aim of this study to merge the laboratory observations with the field investigations and to search for impacts of wastewater exfiltration on the groundwater in a sparsely protected sand and gravel aquifer.

The primary objectives of this study are therefore to:

- Review and compare existing knowledge on exfiltration of wastewater from leaky sewer systems.
- Assess the quantity of exilftrating wastewater in Rastatt based on sewer condition monitoring information for public sewer systems.
- Investigate the hydrochemical impact of exfiltrating wastewater on groundwater resources in Rastatt.
- Perform numerical simulations to model and predict impacts.

While the European Union funded project AISUWRS (Eiswirth *et al.*, 2002b; Wolf et al., in print) follows a broader approach to urban water management, this thesis is limited to the direct assessment of leaky sewer systems. The development of computer models for this assessment is also not discussed in this study but may be reviewed at the homepage of the AISUWRS project (www.urbanwater.de).

## 1.3 Background

### 1.3.1 Related research projects at the University of Karlsruhe

Within the context of leaking sewers, two major research activities were recently concluded out at the University of Karlsruhe. The first one is the research group "Gefährdungspotential von Abwasser aus undichten Kanälen für Boden und Grundwasser (Risk potential of wastewater from leaky sewer systems on soil and groundwater)" and is funded by the German Research Foundation (DFG). Members of the research group are the Institut für Ingenieurbiologie des Abwassers (IBA), Institute for Aquatic Environmental Engineering (ISWW), Engler-Bunte-Institute for Aquatic Chemistry (EBI), Institute for Hydromechanics (IfH), Institute for Mineralogy and Geochemistry (IMG) and the Department of Applied Geology (AGK).

Secondly, a significant part of the European Union funded research project AISUWRS (Assessing and Improving the Sustainability of Urban Water Resources and Systems) is directed to the problem of leaky sewers. Project partners are CSIRO (Australia), British Geological Survey (UK), Robens Centre for Public and Environmental Health (UK), IRGO (Slovenia), GKW (Germany) and University of Karlsruhe (Germany) represented by the Department of Applied Geology as coordinator. The key results of the project are summarized in a book publication (Wolf et al., in print).

Already concluded is the project SAM (Sewer condition Assessment using Multi-Sensor Systems) funded by the German Research Foundation in the years 1998-2003. Within the SAM-project, various sensor-techniques have been tested for their ability to provide quantitative information on sewer conditions (Eiswirth, 2003). These included radioactive probes (Heske, 2003), hydrochemical sensors (Held, 2003), geoelectric probes (Wolf, 2003), acoustic resonance testing (Herbst, 2003), microwaves (Munser & Hartrumpf, 2003) and laser supported measuring techniques (Deutscher *et al.*, 2003). Many of these sensors have also been applied to field conditions in the sewer network of Rastatt, where the specifically equipped sewer test site Rastatt-Kastanienweg provided a good monitoring environment. The results of the different sensors have been processed using a newly developed neuro-fuzzy approach by Frey and Kuntze (2003). A general result of the the project is that CCTV inspections of sewer systems may be significantly enhanced by the combination with alternative sensors but that none of the sensors alone allows a unique interpretation of the signal. Inspection costs using alternative sensor systems are still high which inhibits a widespread universal application. However, this might change in future with more research on marketable products.

### 1.3.2 Wastewater and urban drainage

The entire water flow which has to leave the urban area is called urban drainage. Urban drainage is implemented in industrialized cities in the form of a sewer system. In these sewer systems, the water used by the cities inhabitants as well as the rain water from sealed surfaces is carried away. Traditionally, this is achieved by either a combined or a separate sewer system. A separate sewer system incorporates two subsystems, one system for the storm water component (synonymous to rain water) and the other system for conveying the wastewater from households and industries. Wastewater may be subdivided into grey water, black or brown water and also yellow water. Grey water is generated by the use of drinking water in households for processes like showers, baths and washing machines and characterised by a rather low anthropogenic alteration. Black water (sometimes brown water) is generated through use of toilets and is characterised by a high content of human faeces. Yellow water is generated if urine separating toilets are used and is characterised by a high amount of phosphate but a low amount of potentially harmful germs. Conventional separate sewer systems are set up just for the two different species of storm water (sometimes also referred as daywater) and wastewater. Alternative management strategies aim at the separation into more flows in order to allow a better treatment or reuse of the wastewater. For instance, grey water may also be used for irrigation purposes and also yellow water would not require a full scale wastewater treatment plant before it can be used again. Storm water is generally of similar composition to rain water, but may contain significant amounts of contaminants which have settled on the roof and road surfaces by atmospheric deposition or which are present on paved surfaces due to small spillages (e.g. oil and hydrocarbons from parking lots). This contaminant loading is strongly concentrated on the first flush, e.g. the first minutes after the onset of a rain event. Depending on the usage pattern of the drained surfaces, storm water may be discharged into local rivers, infiltrated into the soil or must be conveyed to the wastewater treatment plant or specialised storm water treatment basins.

In Rastatt as well as in most German cities, urban drainage is achieved mainly (ca. 70 %) by combined sewer systems. In a combined sewer system both storm water and wastewater are transported. Consequently, large diameter sewers have to be used. During dry weather flow, only wastewater is flowing in the combined sewer, usually occupying only a few percent of the pipe cross section. During rain events, the combined system is filled with storm water which contains also a small proportion of wastewater. As the central wastewater treatment plants are frequently not capable of handling such large water volumes, the water is released into the environment at several outlets without further treatment. This is done under the assumption that the large proportion of rain water is sufficiently diluting the wastewater, up to a point at which it does not pose an environmental threat. This assumption is often questioned and most cities worldwide strive towards a reduction of these combined sewer overflows (CSOs). Alternative urban drainage systems including a separation of different wastewater types are discussed and tested in pilot studies (Otterpohl, 1998) but rarely put to large scale application.

### 1.3.3 Exfiltration

Härig & Mull (1992) attempted to quantify the effects of sewage exfiltration for the City of Hannover by balancing the water quality data. However, a significant uncertainity remains as the substances considered (e.g. sulphate) could derive from a number of different urban contamination sources. Considering the multitude of sources for typical urban contaminants, Barret *et al.* (1999) tried to use sewage specific marker species to detect the influence of wastewater on groundwater underneath the City of Nottingham. Dohmann *et al.* (1999) attempted to quantify exfiltration rates based on laboratory and field studies. However, the extrapolation to larger scales was based on very general statistical information on the structural condition of the sewer system. Extrapolations, which are based on CCTV-inspection data have been attempted by Wolf *et al.* (2003, 2006). An approach to validate the assumptions on sewer leakage by the measurement of marker species distributions in the urban aquifer has been described in Wolf *et al.* (2004).

Hagendorf (2004) concludes that emissions into the groundwater are very unlikely if clayey and silty sediments of more than 100 cm thickness are present below the sewer. He supposes that this is independent of the leak dimension. Emissions into the groundwater are likely if sand and gravel are present beneath the sewer and the distance to the groundwater table is lower than 100 cm.

Recently several EU research groups addressed sewer leakage. The APUSS project (Assessing Performance of Urban Sewer Systems) focused on processes inside the sewer network and investigated the applicability of artificial tracer measurements to determine network losses (Blackwood *et al.*, 2005b; Ellis *et al.*, 2004; Rieckermann *et al.*, 2005). Within the CARE-S project (Computer Aided Rehabilitation of Sewer Networks) tools for strategic rehabilitation planning were developed. It comprised also the assessment of sewer leakage and sewer deterioration (Schulz & Krebs, 2004), a state of the art description of urban drainage modelling (Freni *et al.*, 2003) and addressed the assessment of structural condition based on visual insections (Knolmar & Szabo, 2003). Groundwater related research was the focus of the AISUWRS project (Asessing and Improving Sustainability of Urban Water Resources and Systems) where a special model for the estimation of sewer infiltration and exfiltration (NEIMO) was developed (DeSilva, 2005; Wolf et al., in print).

### 1.3.4 Infiltration

Sewer and groundwater are in close interaction in many cities worldwide (Ellis & Revitt, 2002). While this thesis focuses on the exfiltration of wastewater, infiltration of groundwater is an even better known problem for the network operators as it increases the costs in the wastewater treatment plant. The proportion of wastewater flow deriving from groundwater infiltration is also termed “parasite“ water (Karpf & Krebs, 2004). The leaky sewers systems, however, often serve as a groundwater drain which effectively prevents cellar flooding and related problems (BWK, 2003). Getta et al (2004) specifically address these problems for German communities and recommend to estimate the consequences of a sewer network rehabilitation before the construction. Numerical groundwater models are the best tool for this task. (Karpf & Krebs, 2004) even refer to the integration of leaky drainage systems into the flood protection plans for historical buildings in the area of Dresden. A good correlation between the groundwater influenced pipe length and the dry weather runoff has been observed for the City of Dresden. A model based on the concept of a leakage factor has been developed and describes the parasite water volumes reasonably well after model calibration. However, the structural condition of the pipes (e.g. cracks) is not taken into account and the calibration effort was quite high (more than 240 time spots used for calibration) (Karpf & Krebs, 2004). Recently a new model was developed which estimates the infiltrating volumes based on groundwater levels and sewer defect area (DeSilva *et al.*, in print).

## 1.4 Legal Background

The problem of leaky sewer systems in Germany is adressed in two legal frameworks, namely the national water management law (Wasserhaushaltsgesetz) and the European Water Framework Directive (EU-WFD). The German national water law requests that:

- § 18 Abs. I: Wastewater must be discharged in a way that does not lessen the public welfare.
  (Abwasser darf nur so beseitigt werden, dass das Wohl der Allgemeinheit nicht beeinträchtigt wird. Die Beseitigungsaufgabe umfasst im Sinne § 18a Abs. I WHG u.a. das Sammeln, Fortleiten, Behandeln, Einleiten, Versickern, Verregnen und Verrieseln von Abwasser.)
- § 25a WHG : A detrimental change of the ecological state must be avoided
  (Durch die Bewirtschaftung muß eine "nachteilige Veränderung des ökologischen Zustandes" vermieden werden)
- § 34, Abs. I: An emission into the groundwater is not allowed if a detrimental change of its characteristics may be possible.
  (Eine Einleitung in das Grundwasser ist nicht statthaft, wenn eine nachteilige Veränderung seiner Eigenschaften zu besorgen ist.)

Further specifications regarding construction, operation and maintenance of sewer systems are given in numerous technical regulations (DIN 1986, DIN 4033, ATV-Arbeitsblatt A 139, ATV-Merkblatt M 143). On a European level the standard EN 752-2 demands the structural integrity of urban sewer systems including their water-tightness (Keitz, 2002).

Despite the clear legal demands it is a known fact that sewer systems of the distant as well as the recent past do exhibit major constructional deficits and therefore do not fulfil the up-to-date requirements. The real extent of the consequences of sewer leakage for soil and groundwater is only partly known and widespread diffuse contamination of the underground has been suspected. The European Water Framework Directive has recently put more stress on the ecological aspects of the water management law.

According to the German standard DIN 1986-30, sewers discharging wastewater from commercial or industrial sources are considered water-tight if they pass the water pressure test described in DIN 4033 in the course of their first inspection. Alternatively, it is also possible to use air as the testing medium. The validity of optical inspection with CCTV cameras is not specified clearly. A repeated optical inspection is required every 5 years for sewers with industrial wastewater upstream of the first treatment plant and 25 years for residential wastewater systems. More strict inspections and requirements are placed upon the sewer operators if the sewer is situated within a water protection zone. Both private house owners and the public utilities have to ensure the correct operation of their sewage systems. The minimum degree of care necessary for this is defined by the commonly accepted state of the art ("allgemein anerkannte Regeln der Technik") which is defined in a number of regulations specific for each federal state.

# 2 Case Study Rastatt

## 2.1 Background information

The city of Rastatt has approximately 50.000 inhabitants and is located 30 km south of Karlsruhe, close to the eastern border of the Upper Rhine Valley, SW-Germany.

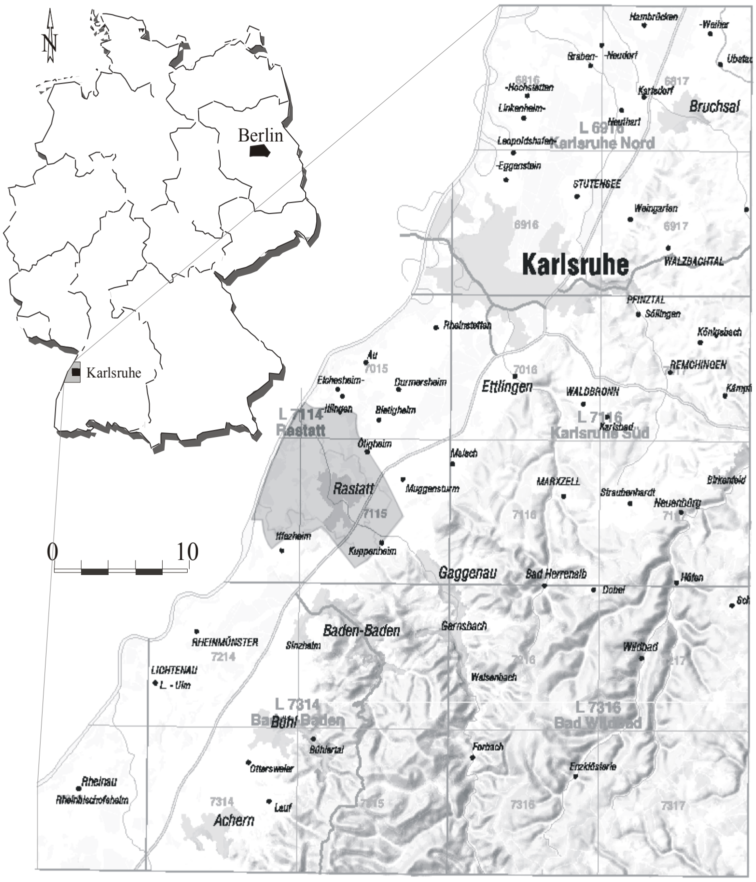



Fig. 2-1: Geographical position of Rastatt in SW-Germany (Eiswirth, 2002)

The city's extension from North to South is 6.7 km, from East to West 7.9 km. The city area is characterised by a rather flat topography at around 115 m a.s.l. in the centre, a minimum of 110.5 m a.s.l. at the location "Große Brufert" and a maximum of 130 m a.s.l. at the castle "Favorite". The mean annual temperature is 10 °C, mean annual precipitation is between 850 mm/m$^3$ and 1000 mm/m$^3$.

Additional general features of the city of Rastatt were described in detail before within the background study of the AISUWRS project (Eiswirth *et al.*, 2003) and are not repeated within this thesis.

## 2.2 Geology

### 2.2.1 General setting in the Upper Rhine Graben

Rastatt is situated close to the eastern margin of the Upper Rhine Graben. The geological structure of the Upper Rhine Graben is a product of the tertiary updoming movement which resulted in a graben structure in its central part, surrounded by the mountain chains of the Vosges in the west and the Black Forest in the east. The Upper Rhine Graben forms the major segment of the Cenozoic Rift system of Western Europe. Although the rift was the target of many seismic and geological investigations, the style of lithospheric extension below the inferred faults, the depth to detachment, and the amounts of horizontal extension and lateral translation are still being debated. An effort to use a finite element model to describe the relevant processes has recently been described (Schwarz, 2005).

The first sediments which filled the graben are of Middle Eocene age and belong to a fresh water facies which was only deposited locally. The main tectonic movements started in the Upper Eocene and resulted in the graben-like sedimentation pattern from the Lower Oligocene onwards. The total thickness of the tertiary and quaternary sediments in the Upper Rhine Graben amounts to ca. 2000 m in the section between Basel and Karlsruhe (Schönenberg & Neugebauer, 1987) and increases to more than 3000 m around Heidelberg an Mannheim (Doebel, 1967).

Tab. 2-1: Sedimentary environments in the Upper Rhine Graben near Rastatt, summarized from different sources (Eiswirth, 2002; Osswald, 2002; Watzel & Ohnemus, 1997).

| System | Series | Sedimentary environment | Sediment sources | Predominant lithology |
|---|---|---|---|---|
| Quarternary | Holocene | Freshwater | Alpine Rhine, Murg River | Sand + gravel |
| | Pleistocene | Freshwater | Alpine Rhine, Kinzig+Murg River | Sand + gravel |
| Tertiary | Pliocene | Freshwater | Pre-Rhine, Black Forest rivers, Vosges rivers | Sand, silt, clay,gravel |
| | Miocene | Periodic marine ingressions | Pre-Rhine, Black Forest rivers, Vosges rivers | Clayey limestone, sand + gravel, evaporities |
| | Oligocene | Periodic marine ingressions | Pre-Rhine, Black Forest rivers, Vosges rivers | Clayey limestone, sand + gravel, evaporities |
| | Eocene | Periodic marine ingressions | Pre-Rhine, Black Forest rivers, Vosges rivers | Clayey limestone, sand + gravel, evaporities |

Tab. 2-1 summarizes the sedimentological history of the upper Rhine Graben filling. The sedimentary sequence is inhomogeneous and the lithology depends on:

- Connection to the Thethys or the North Sea marine basin.
- Slope of the surrounding mountains / rate of subsidence.
- Functioning of the Kaiserstuhl as watershed.

The geology of the Upper Rhine Graben in the area of Rastatt can be divided into four main tectonic units: a graben block, a downfault block, a marginal block (fore hills), and an outlier zone (basement) (Fig. 2-2). The graben is bordered by the Black Forest and the Vosges. Only at the graben block it is possible to differentiate between younger and older Quaternary sediments.

### 2.2.2 Tertiary sediments

From late Eocene to the Miocene, the depositional environment in the area of Rastatt as well as within the whole Upper Rhine Graben was characterised by marine or brackish conditions. In this period marls, clays and evaporites are dominating the sequence (Geyer & Gwinner, 1991). Typical units are the Lymnäenmergel / Grüne Mergel, the Pechelbronner Schichten as well as the Graue Mergel, Foraminiferenmergel, Fischschiefer, Meletta-Schichten and Cyrenen-Mergel. The marine influence decreases in the late Miocene where fresh water sediments have been deposited in significant thicknesses, documenting fluviatile and limnic environments. The Pliocene sediments are solely of fluviatile and limnic character. With this final retreat of the tertiary sea and the onset of the graben volcanism in the Miocene, the basis of the modern surface water drainage pattern is set (Bartz, 1976; Eitel & Blümel, 1990).

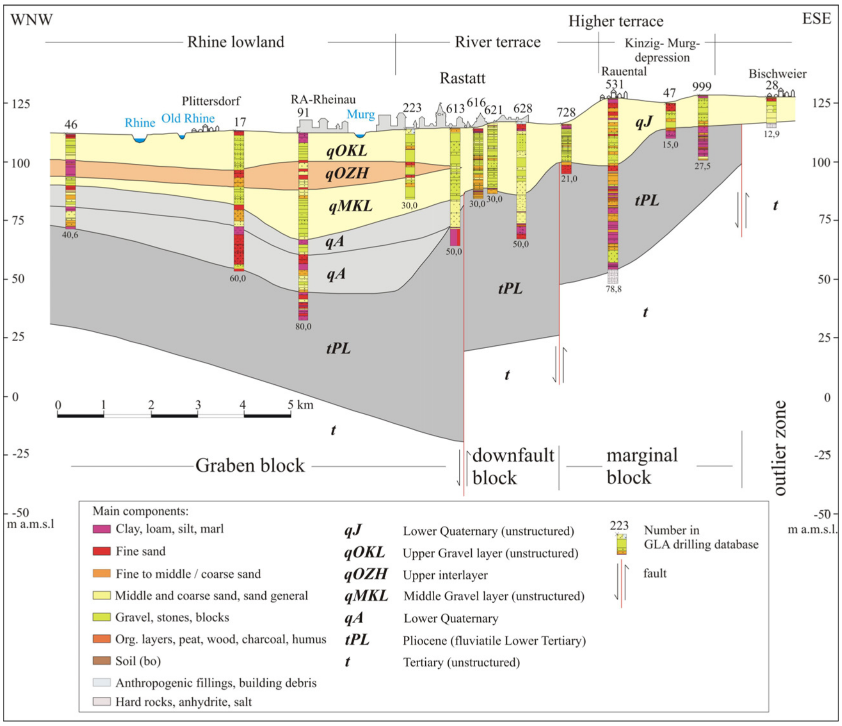



Fig. 2-2: Geological cross section (NW-SE) through the Upper Rhine valley in the area of Rastatt. Modified after (Eiswirth *et al.*, 2004).

The Pliocene sediments are dominantly fluviatile sands with occassional gravel insets. The overall silt content is high and also clay lenses are frequent. Close to the graben border, especially at the outlet of the Murg valley, the sediments contain large blocks of bleached Buntsandstein. The thickness of the Pliocene sediments varies considerably due to (i) the strong topographical relief existing in the Pliocene and (ii) erosion before the deposition of the quaternary sediments.

### 2.2.3 Quaternary sediments

The tectonic activities which created the formation of the individual blocks are still ongoing. The sedimentary units in the centre of the graben are subsiding relatively to the uplifted mountain ranges. The thickness of the younger quaternary sediments increases from 20-30 m on the marginal block to 25-40 m on the downfaulted block and finally to 45-70 m on the graben block.

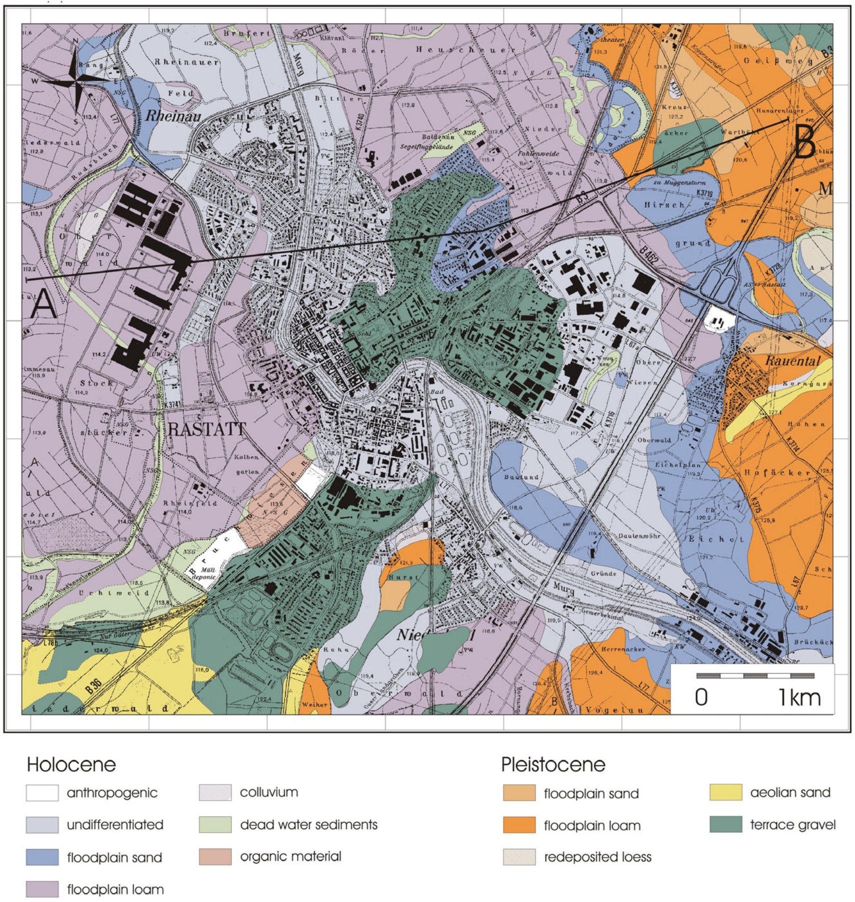



Fig. 2-3: Geological map of the urban area of Rastatt. Redrawn after Osswald (2002).

Only on the graben block it is possible to introduce a finer differentiation of the younger quaternary sediments. The younger quaternary is divided into the Upper and Middle Gravel Layer (Oberes und Mittleres Kieslager) which are separated by a sandy-silty horizon (Oberer Zwischenhorizont) (Huppmann, 1978).

### 2.2.4 Holocene landscape

Rastatt lies in the lowlands of the River Rhine at an altitude of 111 m a.s.l. on the western edge of the main fault of the Upper Rhine Graben. East of the lowlands follows the lower terrace, which is morphologically clearly separated by an 8-10 m high margin north of Rastatt (Fig. 2-4). The lower terrace is mainly set up by gravel and sands of the Würm ice age and partly covered with sand drift. In the Holocene, the river Rhine cut channels up to 20 m deep into the surface of the lower terrace. This erosive material was partly redeposited in the lowlands of the river Rhine and sporadically superposed by flood deposits resulting in 1-2 m thick loam. To the east of the lower terrace runs the Kinzig-Murg channel. This system eroded in the sediments of the lower terrace and is partly filled with postglacial sand, clay and peat (Bartz, 1976).

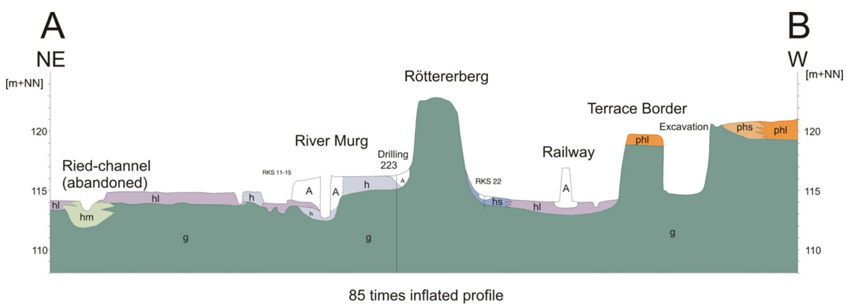



Fig. 2-4: Cross section in the urban area of Rastatt, redrawn after Osswald (2002). g= Pleistocene terrace gravel; phs= Pleistocene floodplain sand; phl= Pleistocene floodplain loam; hl= Holocene loam; h= Holocene undifferentiated; hs= Holocene sand; A= anthropogenic fill; Starting points A and B defined in Fig. 2-3.

During the Holocene epoch, the river Rhine excavated the Lower Terrace Formations within the present Rhine depression and filled it up with recent material in various thicknesses. With generally a clearly rising slope, the Rhine depression borders the valley terrace that in this region corresponds to the uniform flat Lower Terrace plain. Along the mountain border, the rivers coming down from the mountains created a Holocene channel which was running parallel to the river Rhine (the so called Kinzig-Murg-Rinne).

The fluviatile depositional facies leads to a pronounced heterogeneity of the sediments on a small to medium scale, which is also visible in the depth to groundwater table maps of the local authorities (HGK, 1978). The Holocene sedimentary cover clearly exhibits the influence of the meandering rivers. Besides the common sand and gravel sediments, also silty and clayey overbank fines and trench fillings are found frequently.

## 2.3 Hydrogeology

### 2.3.1 General set up

As shown schematically in Fig. 2-5 and Fig. 2-6, a maximum of four aquifer bodies can be distinguished in the Rastatt area: the Upper Gravel Layer (OKL), the Middle Gravel Layer (MKL), the Older Quaternary (qA) and the pliocene aquifer. The aquifer sequence is overlain by holocene cover sediments of varying grain sizes and thicknesses (Fig. 2-3, Fig. 2-4). The Upper Gravel Layer and the Middle Gravel Layer are separated by the Upper Interlayer which consists of more fine grained sediments. The Upper Interlayer does not occur in the entire study area and does only partially constitute a hydraulic barrier. Therefore the Upper Gravel Layer and the Middle Gravel Layer are frequently grouped together as Younger Quaternary aquifer. The names Upper and Middle Gravel Layer suggest that there would also be a Lower Gravel Layer and a Middle Interlayer but this system is stemming from different parts of the Upper Rhine Graben, where more frequent fine grained horizons allow these subdivisions.

In the Rastatt area, the Midlle Gravel layer is underlain by the Older Quaternary aquifer, which is distinguished by a finer grained lithology. The Older Quaternary itself is underlain by the Pliocene aquifer.

The existing geological information may be combined with the knowledge on major water flows to derive a conceptual hydreogeological model for the Rastatt area (Fig. 2-6).

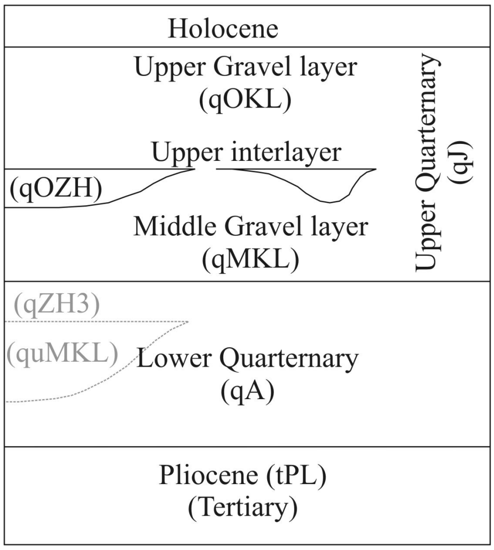



Fig. 2-5: Hydrogeological elements in the Rastatt area (Eiswirth, 2002), redrawn from (Watzel & Ohnemus, 1997).

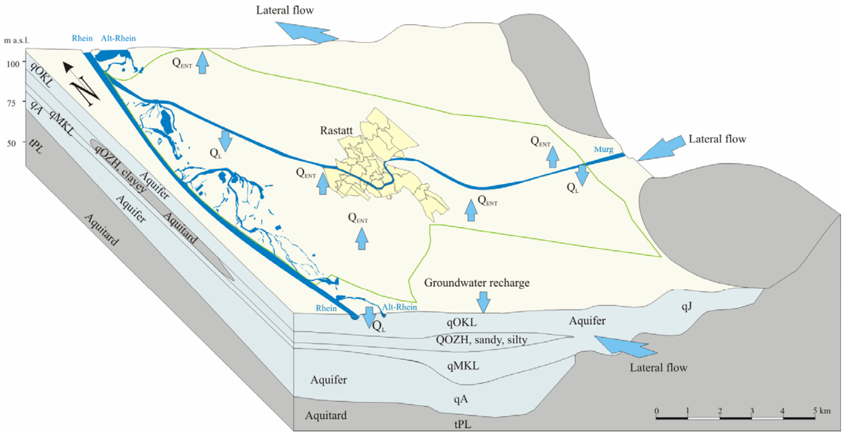



Fig. 2-6: 3d-sketch of the hydrogeological setting in Rastatt. Translated from Eiswirth (2002).

### 2.3.2 Hydrogeological units

#### 2.3.2.1 Younger Quaternary (Jungquartär, qJ)

As shown in Fig. 2-2, no differentiation of the Younger Quaternary sediments can be deduced from the drilling logs on the on the downfault and marginal block in the eastern part of Rastatt. In these parts, the Younger Quaternary (qJ) exhibits thicknesses between 10 m and 40 m. Pumping tests have indicated hydraulic conductivities between $1 \cdot 10^{-3}$ m $s^{-1}$ and $4 \cdot 10^{-4}$ m $s^{-1}$. Public waterworks which tap this aquifer are located in Rauental (365,000 m³ $a^{-1}$) and Niederbühl (630,000 m³ $a^{-1}$) according to Stadtwerke Rastatt (2000).

#### 2.3.2.2 Upper Gravel Layer (Oberes Kieslager, OKL)

The Upper Gravel Layer can be distinguished clearly in the western section of the study area. It is characterised by a high hydraulic conductivity of $2.5 \times 10^{-3}$ m $s^{-1}$ and is the main source of the drinking water production.

The Upper Gravel Layer is tapped by the waterworks in Ottersdorf (around 1,800,000 m³ $a^{-1}$) and by the waterworks Rheinwald which is also the major source of drinking water for the city of Karlsruhe (approx. 270,000 inhabitants). All groundwater observation wells used for this thesis represent the Upper Gravel Layer or the Younger Quaternary aquifer.

#### 2.3.2.3 Upper Interlayer (Oberer Zwischenhorizont, OZH)

As stated in the paragraphs above, the Upper Interlayer does not cover the entire area of Rastatt but is only present in the western part as a hydraulic barrier. The Upper Interlayer is typically characterised by fine sand, silt and clay. However, also a sand and gravel facies of the Upper Interlayer has been specified in some maps. The recognition of this horizon in the drilling logs is usually achieved by the identification of slightly more fine grained sediment material. While traditional literature conceives the interplay of fine grained horizons and thick gravel layers as the product of the changes between Pleistocene cold and warm climatic periods, this view is not supported by more recent palaeontological investigations (Bludau, 1993; Engesser & Münzing, 1992) which point to an ordinary genesis as a distant floodplain facies. The maximum thickness of the Upper Interlayer is 8 m (Eiswirth, 2002). The vertical hydraulic conductivity has been assessed in a view pumping tests is cited with mean values of $6 \cdot 10^{-7}$ m $s^{-1}$ (Ministerium, 1998).

#### 2.3.2.4 Middle Gravel Layer (Mittleres Kieslager, MKL)

The Middle Gravel Layer can be distinguished only in the western part of the study area. It reaches its maximum thickness of approx. 20 m on the graben block. The upper boundary is set between 90-100 m a.s.l., the lower boundary between 70-85 m a.s.l. At the base of the Middle Gravel layer also silty lenses are encountered frequently, marking the transition to the silty sediments of the underlying Older Quaternary. The hydraulic conductivities vary between $3 \cdot 10^{-4}$ m $s^{-1}$ and $1.6 \cdot 10^{-3}$ m $s^{-1}$ with a mean of ca. $1 \cdot 10^{-3}$ m $s^{-1}$ (Ministerium für Umwelt, 1988).

#### 2.3.2.5 Older Quaternary aquifer (Altquartärer Grundwasserleiter, qA)

The Older Quaternary is only present on the graben block as displayed in Fig. 2-2. Also its extension in North-South direction is limited. The deepest point of the lower boundary has been found at approx. 55 m a.s.l. The Older Quaternary aquifer is mainly made up of sands with high silt and clay content. Its hydraulic conductivity has been estimated as $1 \cdot 10^{-4}$ m $s^{-1}$. The Older Quaternary aquifer is only used in conjunction with the other aquifers for water abstraction.

#### 2.3.2.6 Pliocene aquifer (Pliozäner Grundwasserleiter, tPL)

The Pliocene aquifer constitutes the basis of the described hydrogeological sequence and is present in the entire study area. The aquifer is made up of thick gravel and sand beds but exhibits high content of silt and clay. The hydraulic conductivities have been assessed between $2 \cdot 10^{-5}$ m $s^{-1}$ and $5 \cdot 10^{-4}$ m $s^{-1}$ (Ministerium für Umwelt, 1988). As significant silt and clay layers are present at the top of the Pliocene sequence, this aquifer does not have major effects on the water movement in the upper system and is therefore usually omitted in the numerical groundwater models.

### 2.3.3 Unsaturated zone

The unsaturated zone comprises a central part in the risk assessment of sewer leakage as the majority of the pipes are placed in the unsaturated zone. Consequently almost all sewage needs to pass the unsaturated zone. The characteristics of the unsaturated zone exert a major control on i) the quantity of exfiltrating sewage and ii) the decomposition, adsorption, absorbtion of contaminants migrating downward to the groundwater. The unsaturated zone properties in the City of Rastatt are known from consultation of the state drilling log archives as well as own observation drillings. In the course of his Diploma-thesis, J.Osswald created a map of the spatial distribution of low permeability cover layers (Osswald, 2002). This map was later intersected with the depth of the sewer bottom and additionally, areas in which the sewer system is in direct contact with the high permeability zone of the aquifer were determined.

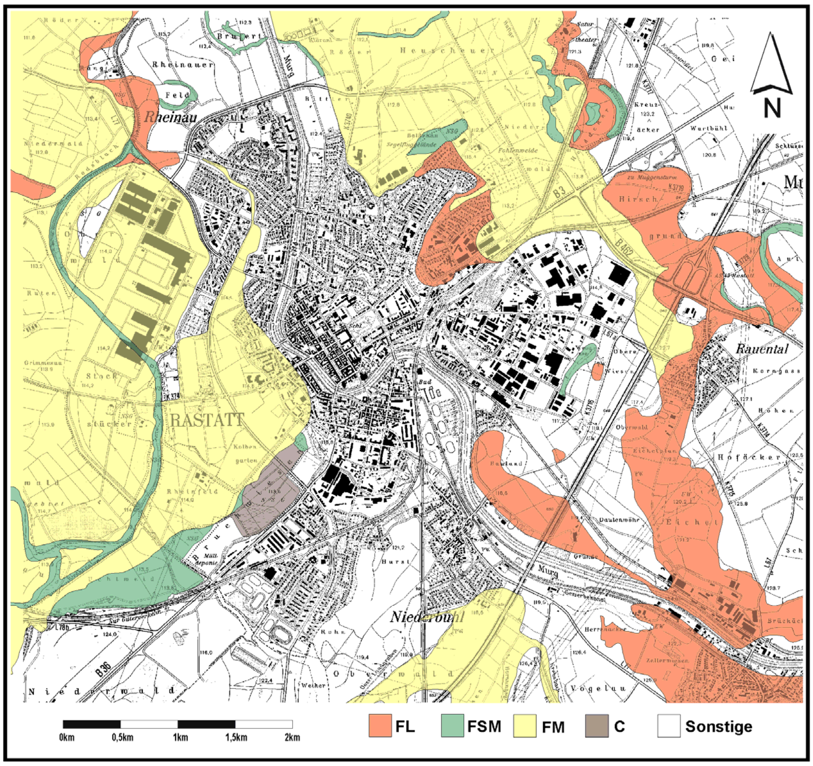



Fig. 2-7: Holocene facies zones as mapped by Osswald (2002) (FL=Fine Laminated, FSM=Fine Silty Massive, FM=Fine Massive, C=rich in carbon, Sonstige=No Holocene sediments).

However, the drilling works for the focus observation wells showed that the variability of the soil properties is very high in the city area. As with most fluviatile deposits, the sediments are characterized by short-distance variations between sandy, gravely and silty layers and regionalization of the information is not possible. Therefore another approach will be adopted.

The unsaturated zone in the investigation area is situated within the two stratigraphical units of the Holocene and the Pleistocene. Inside the Holocene as well as inside the Pleistocene sediments facies zone were mapped out (see Fig. 2-7).

Tab. 2-2: Schematic profiles for each geological unit.

| Profile type | Depth [m] | Sediment type | Sediment type corresponding to Carsel & Parrish (1988) |
|---|---|---|---|
| Pleistozäne Terrasenschotter (Pleistocene Terrace Gravel) | 15 | Sand, Gravel | S |
| Holozäner Auensand (Holocene Floodplain Sand) | 0.5 | Sand, Silt | SL |
| | 15 | Sand | S |
| Holozän ungegliedert (Holocene, memberless) | 0.5 | Sand, Silt | SL |
| | 2 | Sand | S |
| | 3 | Sand, Silt | SL |
| | 15 | Sand, Gravel | S |
| Holozäner Auenlehm (Holocene Floodplain Loam) | 1 | Silt | SI |
| | 2 | Sand | S |
| | 3 | Clay, Sand | SCL |
| | 15 | Sand, Gravel | S |

For the description of the unsaturated soil properties in the demonstration catchment Rastatt-Danziger Strasse all available drilling logs were reviewed again. Four different geological units are distinguished in this area:

- Holocene Floodplain Loam (Auenlehm).
- Holocene Floodplan Sand (Auensand)
- Holocene unclassified (ungegliedert)
- Pleistocene Terrace Gravels (Terassenschotter)

Based on a general knowledge of the sedimentological situation the information from the drilling logs was condensed to just one representative profile type for each of the four geological units as shown in Tab. 2-2.

The sediment types noted in the drilling logs were correlated to the sediment types cited in (Carsel & Parish, 1988). In the empirical studies of Carsel & Parish (1988) the van Genuchten parameters which describe the flow-relevant soil properties were determined. Using this data it was also possible to attribute van-Genuchten parameters to the individual soil layers described in Tab. 2-2. This resulting profiles with soil characteristics were then used as input for the travel time estimations within the AISUWRS project (Mohrlok & Bücker-Gittel, 2005).

The immediate surrounding of the sewer, namely the utility trench, is anthropogenically refilled. However it is assumed that the trenches are in almost all cases refilled with the same material which is dug out as this is the most economical way to do it. Only for a minority of sewers which were built in recent years a special sand layer was placed under the sewer to ensure better soil stability.

### 2.3.4 Groundwater protection zones

The protection of the drinking water quality in the delivering areas is incumbent on the Federal States. The Federal Water Act (§19) regulates the creation of water protection areas. The following zones are distinguished:

- Zone I: Well Head Protection Area (Fassungsbereich), minimum 10 meters from the well head, in the case of negative subsoil surroundings up to 50 meters towards the arriving groundwater.
- Zone II: Closer Protection Zone (Engere Schutzzone), from the limit of Zone I to a line from which the groundwater needs around 50 days to reach the well head

- Zone III: Wider Protection Zone (Weitere Schutzzone), from the limit of Zone II up to the limit of the catchment area of the used water supplies. If the Zone II area limit is greater than two kilometers distance from the limit of the catchment area, the Zone III area can be divided into Zone IIIA and Zone IIIB with the latter being the catchment area beyond two kilometers from the Zone II area limit.

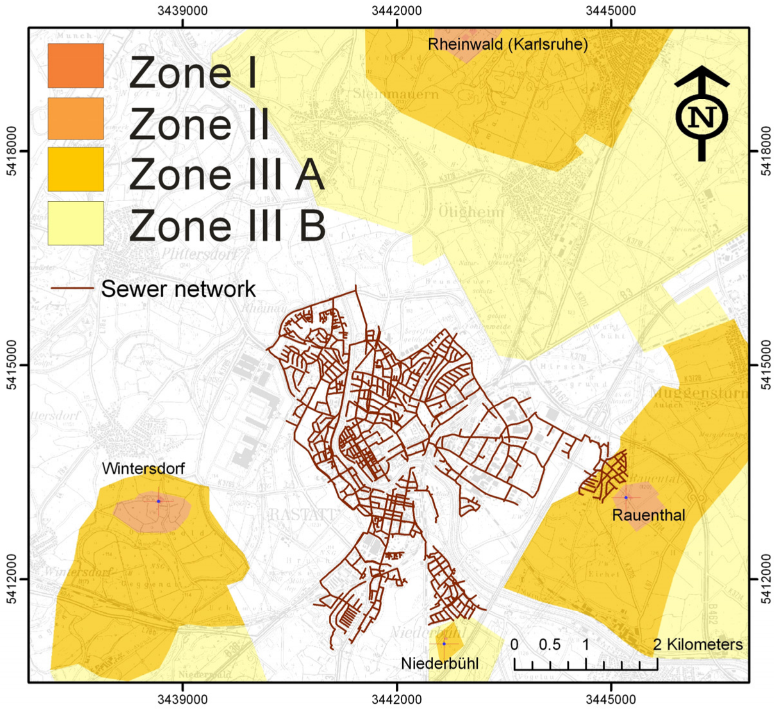



Fig. 2-8: Water protection areas at the waterworks. Zone I is limited to the immediate surrounding of the wells.

The water protection areas for the waterworks in the study area correspond to the particular catchment areas. The city of Rastatt is surrounded by the water protection zones of the water works Rheinwald, Ottersdorf, Rauental and Niederbühl (Fig. 2-8). Only the water work Rheinwald which is the main supplier fort the 300000 inhabitants of Karlsruhe, is situated downstream of Rastatt. The outer border of protection zone IIIb touches the Rastatt city limits. In Zone III B the following activities are forbidden:

- To leach or infiltrate water coming from streets or traffic areas

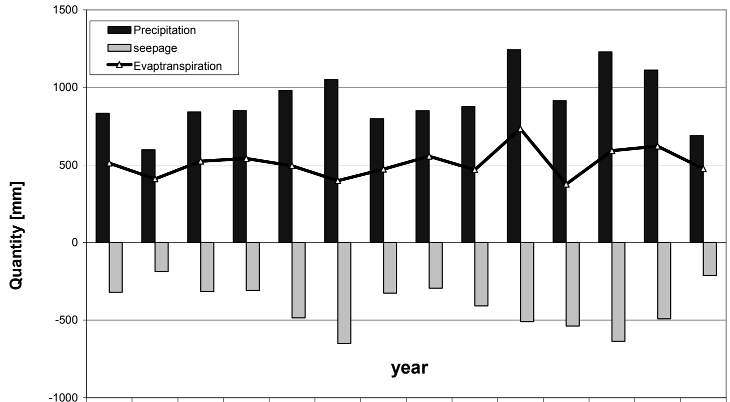
- To leach or infiltrate wastewater
- To operate long distance pipelines containing water damaging substances
- To store radioactive or other water damaging substances
- To construct or enlarge installations that produce, use, store, turn over or treat radioactive and water damaging waste or wastewater.

### 2.3.5 Groundwater recharge

The groundwater recharge is a rather complex parameter in the hydrological cycle as it depends on numerous parameters like land use, unsaturated zone thickness, soil type, precipitation and potential evapotranspiration. Furthermore, it is time variant. Groundwater recharge needs to be determined in its spatial distribution for the whole study area of Rastatt. For the modelling purposes of the AISUWRS project, the groundwater recharge was determined using different methodologies for rural and urban areas. This separation had to be done due to the different processes (e.g. runoff from paved areas) effective in both compartments. For agricultural areas, data like evapotranspiration of the respective plants, seepage through the soil body etc. were measured at lysimeters and the spatial distribution of groundwater recharge was calculated according to the land use and groundwater levels.

For the populated areas the determination is much more difficult. No lysimeters exist in the urban area of Rastatt. Most existing lysimeters are located in rural areas because the data are usually used for agriculture. Urban town planners, however, are traditionally mainly interested in runoff coefficients. Furthermore, the urban surface is covered by houses, streets, pathways and parks. This complex structure with different land uses and changing runoff behaviour complicates regionalisation.

In the rural area of Rastatt groundwater recharge below pervious areas from rain water is monitored in several lysimeters controlled by the LfU. They are situated in the close surroundings of Rastatt. The longest time series to relate precipitation, temperature and seepage is available from the Rauental Lysimeter. Between 1990 and 2003 the groundwater recharge (or seepage) measured at the lysimeter varied between 187 mm/a and 651 mm/a.




| | 1990 | 1991 | 1992 | 1993 | 1994 | 1995 | 1996 | 1997 | 1998 | 1999 | 2000 | 2001 | 2002 | 2003 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Precipitation | 832,8 | 597,5 | 841,4 | 850,6 | 980,1 | 1050,3 | 798,5 | 849,7 | 876,7 | 1242,6 | 914,4 | 1228,8 | 1112 | 689 |
| seepage | -320,85 | -187,49 | -316,48 | -309,54 | -486,01 | -651,68 | -325,03 | -292,9 | -407,97 | -509,31 | -538,04 | -636,74 | -491,17 | -213,65 |
| Evaptranspiration | 511,95 | 410,01 | 524,92 | 541,06 | 494,09 | 398,62 | 473,47 | 556,8 | 468,73 | 733,294 | 376,36 | 592,06 | 620,83 | 475,35 |

Fig. 2-9: Observations at the Rauental Lysimeter maintained by LfU Baden Württemberg (Wolf *et al.*, 2005b).

For the implementation of the groundwater recharge into the 2D-steady state groundwater flow model the punctual measurements had to be transferred into a spatial distribution. Therefore, a model estimating groundwater recharge was applied to each of these units and the results were validated using lysimeters and soil moisture measurements (Pfuetzner, 1996). As a basis for the transient numerical groundwater model of the water work Rheinwald, a detailed modelling of the groundwater recharge between 1964 and 1990 was performed (Pfützner, 1996). For this purpose the area was divided into different units depending on the soil type, the land use like meadows, acres, forests, lakes and rivers, and the distance between the ground surface and the groundwater table. The model results were validated with measurements from the lysimeters.

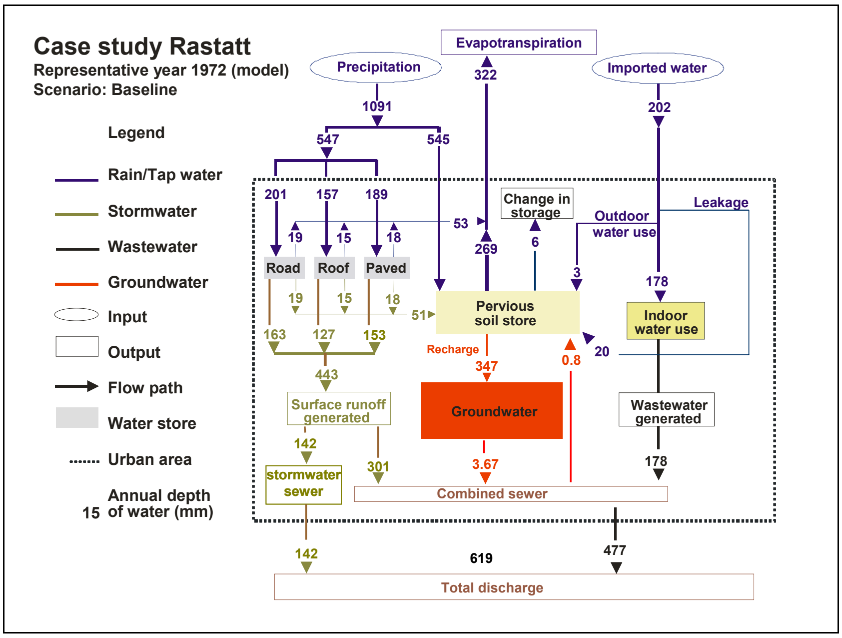



Fig. 2-10: Water balance diagram calculated with the AISUWRS DSS model chain (Klinger *et al.*, 2006; Wolf et al., in print).

For the city area of Rastatt the recharge assessment methodology used by Pfuetzner (1996) can not be applied because the structure of the land parcels and the associated land use pattern is very complex (land use changes within short distances between houses, streets, driveways and open spaces like parks or gardens). In the existing groundwater models of Kühlers (2000) and Klinger (2003), groundwater recharge values were applied which represent an average over the whole urban area and depend on the general structure of the settlement. This value most likely does not represent the real situation beneath the city of Rastatt as it neglects all other urban recharge sources like leaky water mains and sewers. However, the regional groundwater models are operating based on imprecise groundwater recharge data in urban areas and inconsistencies between observed and calculated groundwater levels were alleviated by changes in hydraulic conductivities of the aquifer.

A refined assessment of the urban groundwater recharge was produced within the AISUWRS project (Fig. 2-10).

### 2.3.6 Long term analysis of groundwater levels

An extensive dataset on groundwater levels was acquired from the environmental agency of the state of Baden-Württemberg (LfU). It includes 210 groundwater observation wells, each with a time series of piezometric head.

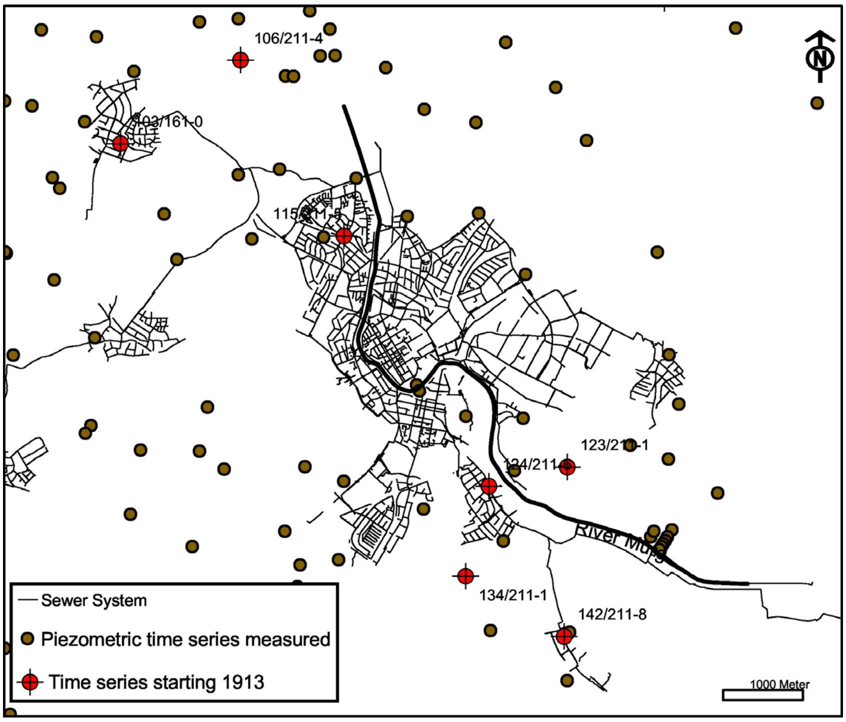



Fig. 2-11: Observation points with time series of piezometric head extracted from the LfU-database.

The longest time series start in 1913 and were only interrupted 1924–1931 and 1936–1945. Observation wells in the surrounding of the City of Rastatt are displayed in Fig. 2-11. The available readings were screened for long term trends in groundwater levels. For the majority of wells monitored since 1913, a decrease in groundwater levels is apparent (Fig. 2-12) following the second world war (1939–1945). Well 115/211-5 is the only long time series well which is situated beneath the urban area of Rastatt. More precisely, it is located in the sub-district of Rheinau, a residential area which was mainly developed after 1945. Well 106/211-4 is situated downstream of Rastatt between the suburbs Plittersdorf and Steinmauren. Well 123/211-3 is situated upstream of Rastatt.

The long term decrease of water levels amounts to approx. 0.7 m for the two wells downstream of Rastatt and approx. 1 m for the upstream well. However, water levels appear to be stable since 1950. Different factors might have influenced the evolution of water levels in the observation wells:

- Rising groundwater abstraction rates.
- Increased degree of sealing, increased surface runoff.
- Construction of dams at the river Rhine.

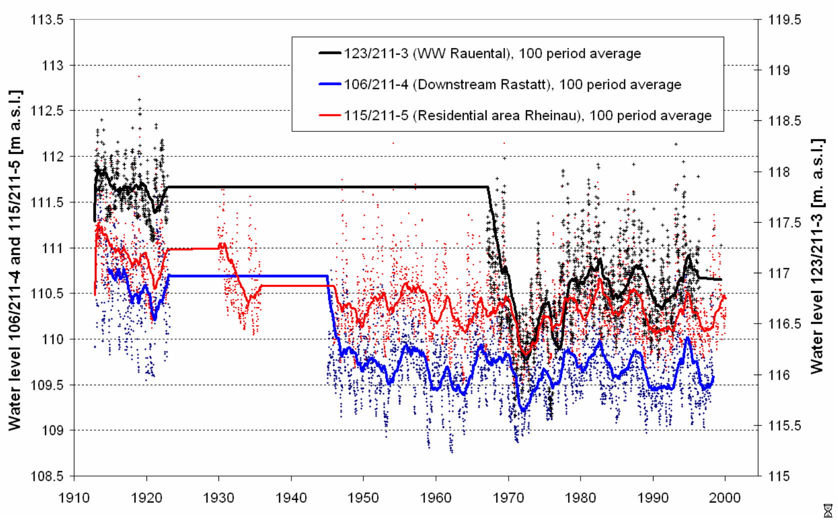



Fig. 2-12: Long term evolution of hydraulic heads since 1913. Raw data extracted from LfU-database.

Closer inspection of groundwater level time series in the years 1970-2000 shows that only the well 158/211-0 beneath the residential area of Rheinau has a slightly reclining trend (Fig. 2-13).

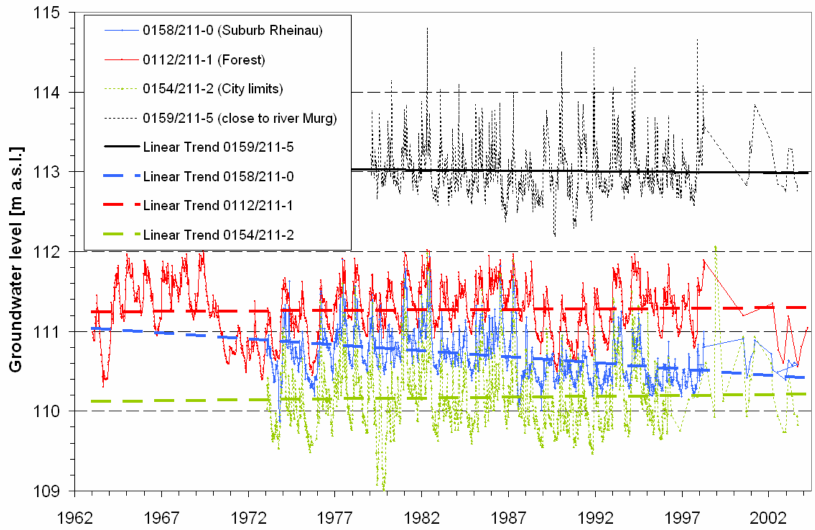



Fig. 2-13: Long term evolution of groundwater level at reference wells inside and outside the city area (Data from LfU and own measurements).

While well 0159/211-5 is also situated in the city centre, it must be considered that it has been drilled within a distance of 20 m to the river Murg and that it is heavily influenced by the river water level.

Drawing up a closer comparison between the time series evolution of the well in Rastatt-Rheinau and a well located in a forest north of Rastatt, it is obvious that the difference between the groundwater levels in both wells is increasing (Fig. 2-14). The change is pronounced following the year 1988 and may be linked to the construction of a very large automobile production plant in the immediate vicinity (observed in well 0158/211-0).

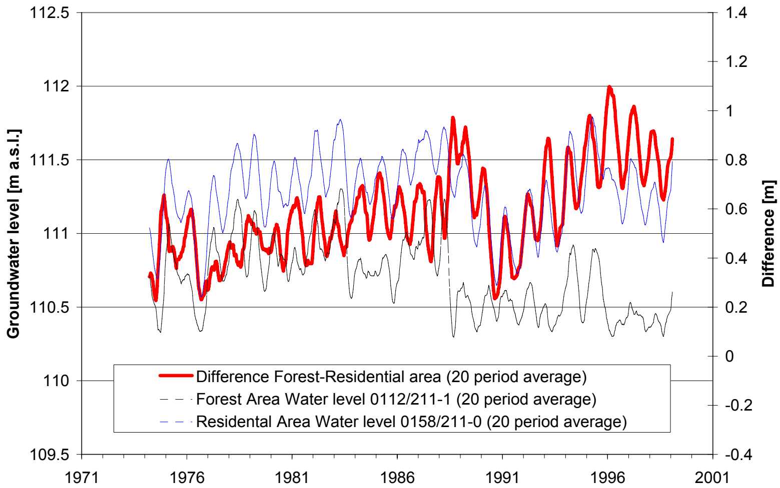



Fig. 2-14: Different evolutions of hydraulic heads beneath a residential area and a forest area (raw data from LfU and own measurements).

However, observed changes in groundwater levels in the area of Rastatt during the last 90 years indicate that the anthropogenic impact on groundwater levels in this area is hardly noticeable. This is due to the very productive younger quaternary aquifer, the high recharge rates and the exchange with the surface waters of the Rhine and Murg systems.

# 3 Hydrochemical evidences of wastewater exfiltration

## 3.1 Generalities / Marker species

In order to demonstrate the effects of exfiltrating wastewater on the urban groundwater, monitoring wells in the city area were tested for several wastewater specific marker species. The challenge in using marker species is to link them unambiguously to a wastewater source alone or to locate and quantify all other sources. In addition, the low marker species concentrations between µg/l and ng/l require high analytical effort.

Wastewater from leaky sewers does not only contain illegally discharged hazardous substances like chlorinated hydrocarbons. In addition, a constant and highly diverse mixture of legally used chemicals is running through the sewer system, containing many substances for which no adequate toxicological information is available. These so called "emerging pollutants" include for instance pharmaceutical residues, endocrine disruptors and personal care products (PCP). Much research work is currently going on to detect these emerging pollutants in surface waters (Hirsch *et al.*, 1999; Hirsch *et al.*, 2000; Hirsch *et al.*, 2001; Richardson & Ternes, 2005; Ternes *et al.*, 2002). Degradation is highly specific for each substance (Kümmerer, 1998). The incomplete removal in the wastewater treatment plants is regarded as the main entrance pathway of those substances to the surface water. However, wastewater leaking from defect sewer has never passed a treatment plant and so the pollutants are directly released into the urban drinking water resource. It is still an open question whether this direct release might have an even worse ecological impact than the emission into surface waters. The ecotoxicology and mobility of endocrine disruptors is also the topic of current research (Helmreich, 2001; Weltin & Bilitewski, 2001). Most of the adverse endocrine effects reported up to now deal with feminization of fishes and amphibians (Kloas, 2001; Oetken, 2005).

Personal Care Products (PCP) such as liquid bath additives, soaps, skin-care products, shampoos and dental care products are produced and used in large amounts world wide. In the early 1990s, the annual production volume was more than 550,000 metric tonnes for Germany alone. Considerable persistence and bioaccumulation potential was shown for a number of PCPs like musk fragrances, disinfectants, antiseptics, some repellents and sunscreen agents (Ternes *et al.*, 2002). Pharmaceutical residues have already been detected in the drinking water of Berlin, Germany (Heberer & Stan, 1996; Heberer, 2002; Stan & Linkerhägner, 1992; Stan *et al.*, 1994). Clofibric acid, used as a blood lipid regulator was found in 64 tap water samples in Berlin with concentrations between 10 and 165 ng/l (Heberer, 2002; Stan *et al.*, 1994). Likewise, it has also been detected in deeper groundwaters below the outskirts of Berlin as a result of sewage farm operation (Scheytt *et al.*, 2000). Iodated organic compounds already constitute an important fraction of urban wastewater (Jekel & Wischnack, 2000), especially downstream of hospitals (Larsen *et al.*, 2000).

Caffeine was used recently as an anthropogenic marker for wastewater contamination of surface waters in Switzerland (Buerge *et al.*, 2003). The concentrations measured in Lake Zurichsee correlated with precipitation data. Combined sewer overflows are believed to be the cause of this correlation.

Boron was applied as wastewater trace substance also in previous studies (Gäbler & Bahr, 1999; Härig & Mull, 1992) and a substantial amount of literature on boron concentrations in the aquatic environment is available. Much of this literature stems from the discussion on lowering the drinking water limit to 0.3 mg/l Boron (currently 1 mg/l) in the late 1990s (Abke *et al.*, 1997; Coughlin, 1998; Metzner *et al.*, 1999; Wiecken & Wübbold-Weber, 1995). The most important intake route for the aquatic environment is that involving detergents and cleaners as these are legally discharged into the wastewater (Metzner *et al.*, 1999). A statistical analysis of 3028 boron measurements in Baden-Württemberg showed a median concentration of 0.03 mg/l boron, mean concentration of 0.08 mg/l boron and the 90 % percentile was found at 0.15 mg/l boron (Metzner *et al.*, 1999). These values correspond fairly well with the Rastatt data set.

Other possible marker species proposed (Barret *et al.*, 1999; Ellis & Revitt, 2002) like chlorination by-products (THM), faecal steroids (coprostanol), synthetic estrogens, detergents, chlorinated solvents and stable isotopes ($^{15}N$) have not been screened for in Rastatt but may be incorporated in future analytical programmes.

A broad screening exercise for pharmaceutics and hormones in the aquatic environment was conducted between the years 2000 and 2002 in Baden-Württemberg by the environmental agengy (LfU, 2002), parallel to the measurements of this thesis in Rastatt. In total 180 samples were taken from 105 different groundwater observation wells and analysed for a spectrum of 74 substances. The most prominent findings are recorded for solatol, phenazone, propyphenazone, carbamazepine, diclofenac and dehydrato-erythromycon and are summarized in Tab. 3-1.

Tab. 3-1: Results of a screening for 74 pharmaceutical substances in 105 groundwater observation wells in Baden-Württemberg. Data extracted from LfU (2002).

| Substance Group | Substance | Positive detects > 10 ng/l | Maximum concentration measured [ng/l] |
|---|---|---|---|
| Betablocker | Metoprolol | 1 | 110 |
| | Bisoprolol | 1 | 12 |
| | Sotalol | 3 | 560 |
| | Terbutalin | 1 | 12 |
| Blood lipid regulator | Gemfibrozil | 1 | 14 |
| Analgesic | Phenazon | 5 | 25 |
| Analgesic | Propyphenazon | 2 | 19 |
| Analgesic | Diclofenac | 4 | 590 |
| Analgesic | Indometacin | 1 | 22 |
| Iodated X-Ray Contrast Media | Iopamidol | 5 | 300 |
| Iodated X-Ray Contrast Media | Amidotrizoesäure | 21 | 1100 |
| Anticonvulsant | Carbamazepin | 13 | 900 |
| Makrolid-Antibiotica | Dehydrato-Erythromycin | 10 | 49 |
| Makrolid-Antibiotica | Roxithromycin | 1 | 26 |
| Sulfonamide-Antibiotics | Sulfamethoxazol | 11 | 410 |
| Sulfonamide-Antibiotics | Sulfadiazin | 2 | 17 |
| | Ronidazol | 1 | 10 |
| Phyto-Estrogenes | beta-Sitosterol | 21 | 260 |
| Alkylphenole | Bisphenol A | 26 | 1600 |
| | Bisphenol F | 4 | 270 |
| | iso-Nonylphenol | 68 | 7100 |

Steroids like 17-α-ethinylestradiol, 17-β-estradiol, estriol, estron, mestranol and norethisteron have been screened for in 105 groundwater observation wells in Baden-Württemberg without any positive detects, even though the analytical detection limit was at 1 ng/l. Likewise, no positive detects were found for the group of penicillin. In contrast to the widespread occurrence of clofibric acid in Berlin, no traces could be detected in the 105 observation wells screened in Baden-Württemberg. In parallel, also boron concentrations have been determined and a positive correlation between boron concentration and pharmaceutical residues was found in the groundwater. The LfU authors assume that the influence of wastewater is the most common cause for the positive detects (LfU, 2002).

In contrast to the findings of the LfU study, no pharmaceuticals were detected in 8 groundwater samples in Rastatt which have been screened for a spectrum of 15 substances (Wolf *et al.*, 2004). To explore the representativeness of this limited sampling program, additional pharmaceuticals were investigated in selected wells in Rastatt in June 2005. Ten samples were analysed for "group one" pharmaceuticals, including betablockers and analgesics.

## 3.2 Methods

### 3.2.1 Sampling

Up to 60 groundwater monitoring wells in and around the urban area of Rastatt were selected for area-wide screening campaigns. Using this observation network six large sampling campaigns and additional monthly samplings covering groundwater, surface water, and wastewater were conducted by the Department of Applied Geology during the years 2001-2005.

Groundwater sampling was performed using submersible pumps. Samples were taken after stabilisation of the field parameters (sp. el. conductivity, temperature, pH, dissolved oxygen) in the volume flow. Caution was taken to avoid contact of samples designated for boron analysis with glass (interaction with the borosilicate).

The analysis of long term data series on groundwater levels in the Rastatt area shows that the typical range of water level variations is about 1.8 m. The hydrochemical samplings conducted in the years 2001-2005 were performed during comparatively dry periods. With only one exception, groundwater levels in well 112/211-1 were below the long term median during the sampling periods. This ensures that the samples do not represent a time with significantly lower concentrations due to strong dilution with recent precipitation.

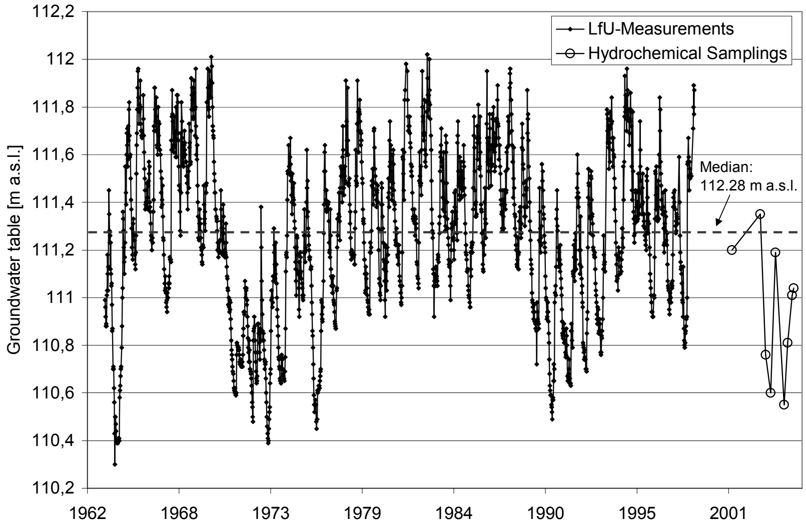



Fig. 3-1: Groundwater level at the reference well 112/211-1 during the last 40 years in relation to the hydrochemical samplings (data from LfU and own measurements).

### 3.2.2 Analytical methods

All water samples were analysed for major anions using ion chromatography while major cations were analysed using Flame Absorption Spectrometry at the AGK laboratory in the Environmental Research Centre Karlsruhe (FZU).

Analyses for boron and heavy metals were conducted at the Institute for Mineralogy and Geochemistry (IMG) in Karlsruhe using Inductive Coupled Mass Spectrometry (AXIOM). A second series of boron samples were analysed by the British Geological Survey (BGS) in Wallingford, UK using the same ICP-MS technique. Gadolinium measurements were accomplished by Prof. Peter Möller (GFZ Potsdam) using solid phase extraction and GC-MS as described in Bau & Dulski (1996).

Measurement of pharmaceutical residues and iodated x-ray contrast media was performed at the Centre for water technology (TZW) in Karlsruhe using solid phase extraction and LC-MS-MS as described in Sacher & Brauch (2001). The detection limits for pharmaceuticals and iodated x-ray contrast media are at 10 ng/l in groundwater. Additional measurements of pharmaceutical residues and x-ray contrast media were performed at the Engler-Bunte-Institute for Aqueous Chemistry using using solid phase extraction and LC-MS-MS-measurements of microbiological parameters were performed for selected samples as documented in Paul *et al.* (2004) or analysed according to German industrial standards (DIN EN ISO 9308-1).

## 3.3 Sampling well networks

### 3.3.1 City wide sampling network

A large network of observation wells is maintained for the entire state of Baden-Württemberg by the environmental agency (LfU). The overall status of the groundwater resources with respect to water quality and quantity is summarized in annual reports (e.g. LfU 2002, LfU 2003, LfU 2004). The Department of Applied Geology received an extract from this large database. Water quality observation wells maintained by the LfU in the Rastatt area are depicted in Fig. 3-2. Most of the sampling points are located outside the urban area and information on the northern part of the city is sparse.

As the existing network of LfU sampling points was not sufficiently dense for the purpose of this study, additional wells were sampled by the Department of Applied Geology. The positions of these wells were supplied by the local environmental agency (Umweltamt Rastatt) or were taken from the data generated by Arcadis (1999).

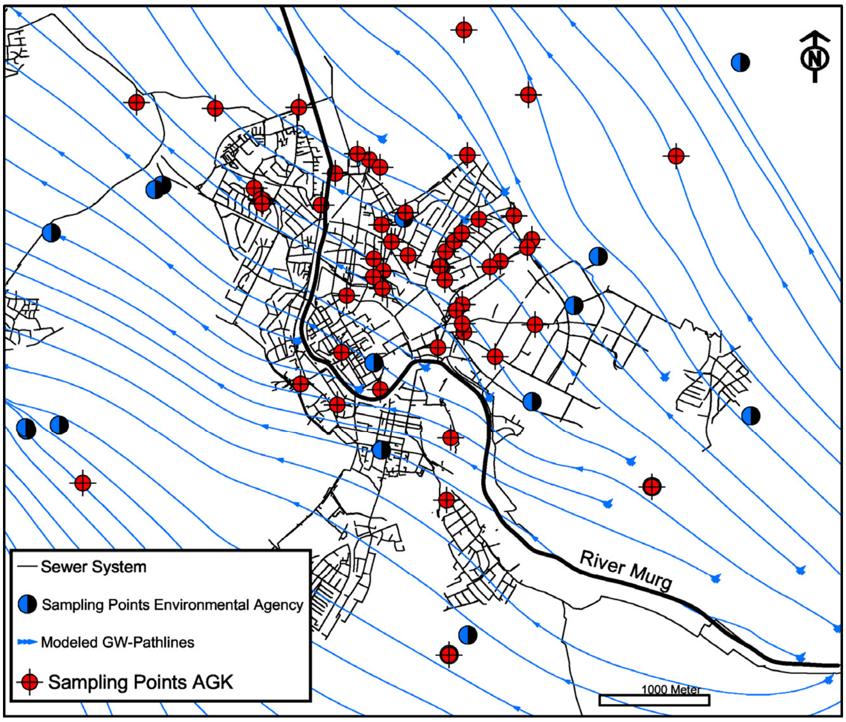



Fig. 3-2: Groundwater observation wells used for sampling during the years 2001-2005. Half filles circles mark sampling points of the environmental agency, Circles with crosshairs mark the additional wells sampled in this study.

### 3.3.2 Focus observation wells

The sampling campaigns during the years 2001 and 2002 indicated that no widespread influences of leaky sewers were detectable. Therefore it was decided to build new observation wells close to known sewer leaks. For this reason, the sewer defect database was searched for large diameter sewers with severe defects (Fig. 3-3). In a second step the pre-selected sites were evaluated in terms of their accessibility and potential conflicts with other urban infrastructure (gas or water pipelines, telecommunications and electricity).

Finally, eight new groundwater monitoring wells were built. The wells were drilled with a diameter of 225 mm and equipped with 50 mm PVC pipes and screens. The annular space was filled with gravel and sealed with bentonite against contamination from the surface.

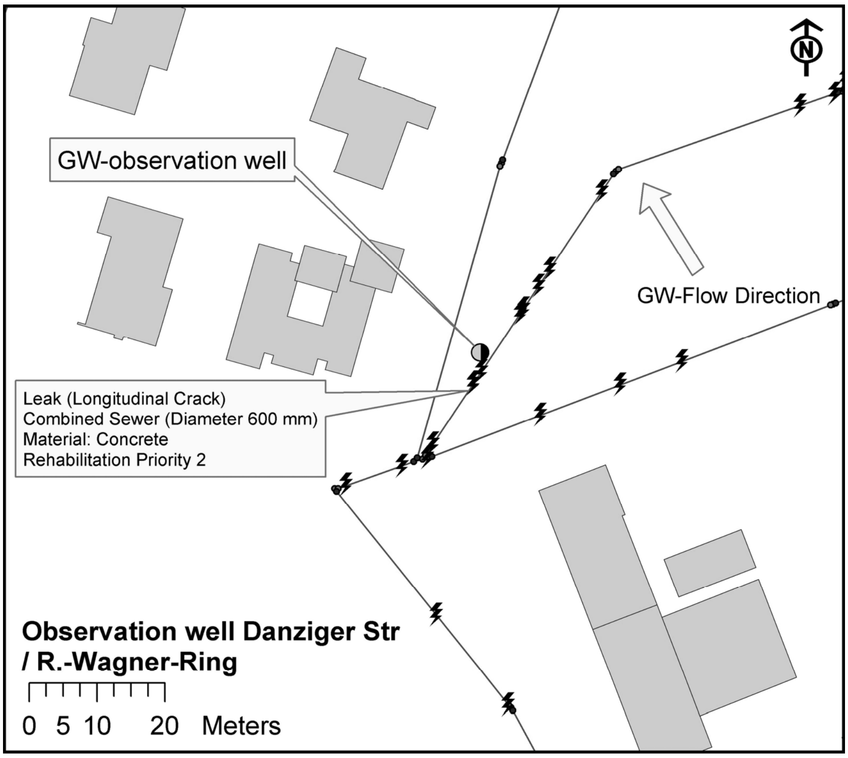



Fig. 3-3: Map used for site selection of the sewer focus observation wells.

One additional sewer focus well has been constructed in the vicinity of a building, whose owner currently sues the City of Rastatt on the topic of sewer leakage or drinking water mains leakage. The law suit is still open and the causes for the water ingress into the cellars as well as the cause for the building subsidence are not proven in court.

Three further sewer focus wells have been constructed in summer 2004 at the sewer test site Danziger Strasse and are detailed in chapter 3.5. In this study, all of these newly constructed groundwater monitoring wells, will be referred to as focus observation wells. Their positions are given in Fig. 3-4.

### 3.3.3 Concept of analysis using well groups

In order to compare the quality of groundwater beneath the urban area of Rastatt with the surrounding rural area, the observation wells are grouped according to their spatial position (Fig. 3-4). Well group 0 comprises wells which are situated outside the city area ('Out of town'), either in forests or on agricultural land. Well group 1 is assigned to wells which are situated on the outskirts of the city ('City limits'). Well group 2 comprises wells which are located directly beneath the settlements and is dubbed as urban background. Well group 3 consists of wells which have been specifically drilled to monitor the effects of wastewater exfiltration ('sewer focus') and are described in chapter 3.3.2.

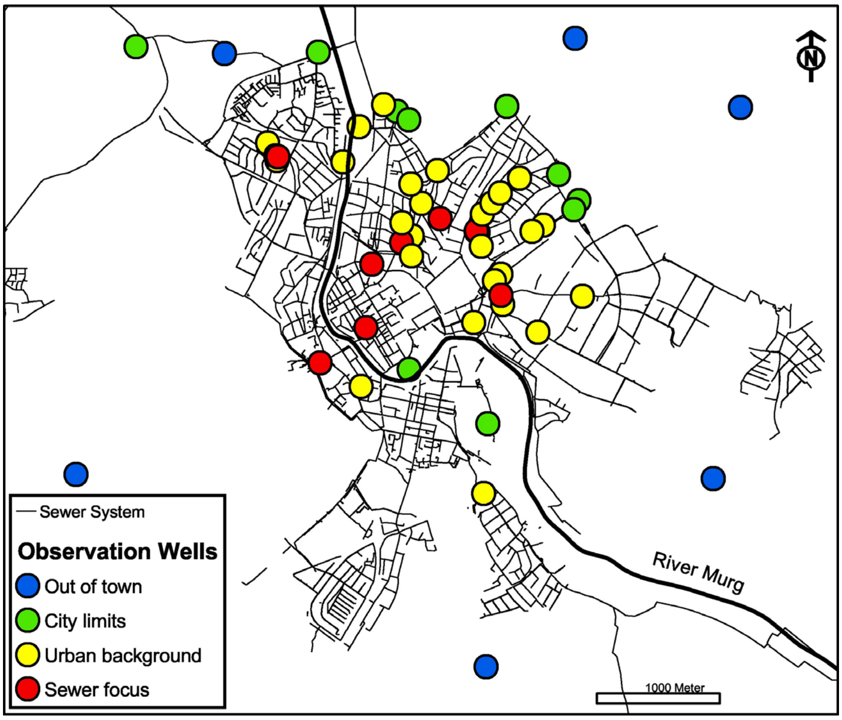



Fig. 3-4: Classifying observation wells based on their exposure to urban pollution sources.

## 3.4 Results

### 3.4.1 Overview and statistical parameters

Within the years 2001 to 2005, 6 city-wide sampling campaigns and 14 focused sampling campaigns were performed in the City of Rastatt.

Tab. 3-2: Statistical parameters of selected parameters during the samplings 2001-2005.

| Parameter | Unit | Number of stations | Number of analyses | Min | Max | Arith. mean | 84,1 % Qt. | 95 % Qt. |
|---|---|---|---|---|---|---|---|---|
| Next Sewer | [m] | 61 | | 1.0 | 1150.0 | 98.2 | 55.8 | 750.0 |
| Next Up-stream Sewer | [m] | 61 | | 1.0 | 1700.0 | 178.6 | 344.0 | 1000.0 |
| Water Level | [m.b.g.l] | 57 | 300 | 0.91 | 8.45 | 3.75 | 5.56 | 7.58 |
| S. E. C. | [µS/cm] | 60 | 321 | 193 | 1053 | 563 | 680 | 880 |
| Temp | [°C] | 59 | 324 | 8.9 | 19.1 | 12.9 | 14.3 | 14.9 |
| pH | [-] | 57 | 259 | 6.4 | 7.7 | 7.1 | 7.4 | 7.6 |
| Oxygen | [mg/l] | 54 | 280 | 0.05 | 7.40 | 1.79 | 3.08 | 4.89 |
| Redox [mV] | [mV] | 46 | 72 | -146 | 260 | 112 | 217 | 250 |
| $Na^+$ | [mg/l] | 58 | 302 | 5.90 | 38.51 | 16.23 | 21.37 | 33.03 |
| $K^+$ | [mg/l] | 58 | 302 | 1.10 | 16.30 | 4.78 | 7.99 | 10.76 |
| $Ca^{2+}$ | [mg/l] | 58 | 301 | 31.4 | 194.2 | 102.6 | 133.2 | 156.6 |
| $Mg^{2+}$ | [mg/l] | 58 | 299 | 2.21 | 21.30 | 7.31 | 10.25 | 14.00 |
| $Cl^-$ | [mg/l] | 58 | 234 | 6.90 | 53.90 | 20.36 | 26.20 | 43.33 |
| $NO_3^-$ | [mg/l] | 56 | 214 | 0.17 | 70.06 | 12.29 | 21.34 | 34.26 |
| $NO_2$ | [mg/l] | 15 | 20 | 0.30 | 3.55 | 1.76 | 2.41 | 3.51 |
| $PO_4^{3-}$ | [mg/l] | 45 | 96 | 0.50 | 3.66 | 1.01 | 1.35 | 1.94 |
| $SO_4^{2-}$ | [mg/l] | 58 | 232 | 1.54 | 73.09 | 33.25 | 43.21 | 57.78 |
| $F^-$ | [mg/l] | 37 | 50 | 0.30 | 2.85 | 0.99 | 1.30 | 2.62 |
| $NH^{4+}$ | [mg/l] | 51 | 218 | 0.01 | 2.71 | 0.26 | 0.38 | 1.02 |
| B | [mg/l] | 54 | 258 | 0.02 | 0.23 | 0.06 | 0.09 | 0.12 |
| $CO_2$ | [mg/l] | 49 | 84 | 5.5 | 120.2 | 35.1 | 58.8 | 84.8 |
| $HCO_3^-$ | [mg/l] | 54 | 103 | 80.8 | 640.5 | 310.1 | 393.7 | 530.9 |
| AOX | [mg/l] | 40 | 48 | 1.71 | 21.87 | 6.21 | 9.16 | 10.81 |
| DOC | [mg/l] | 43 | 83 | 0.36 | 5.30 | 1.36 | 2.29 | 4.23 |
| AOX/DOC | [mg/l] | 40 | 48 | 0.87 | 26.21 | 6.67 | 11.57 | 14.76 |

A total of 7019 data entries deriving from 379 samples from 61 different sampling points were generated. In a time-consuming effort the results have been compiled into one data structure and analysed for their plausibility. The statistical parameters of the key elements are listed in Tab. 3-2. With few exceptions who tap both Upper and Middle Gravel Layer, all wells tap the Upper Gravel Layer with typical depths of max. 10 m below ground level.

### 3.4.2 Major elements and field parameters

#### 3.4.2.1 Groundwater

The vast majority of samples was analysed for the major cations potassium, sodium, magnesium and calcium as well as for the anions sulphate, chloride, nitrate, phosphate and hydrogen carbonate. Key statistical parameters are listed in Tab. 3-2.

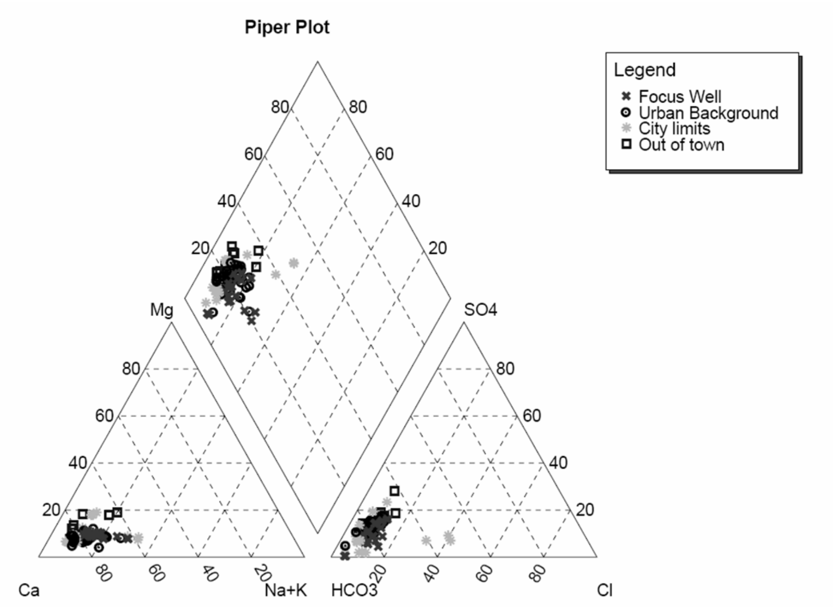



Fig. 3-5: Piper plot for samples from the Upper Gravel Layer with complete ion balance sampled 2001-2005.

The interpretation of the boron analysis is included in this chapter. Several processes affect the distribution of groundwater constituents:

- Sewer leakage
- Infiltration from open spaces (private gardens, public parks)
- Fertiliser application
- Road salting
- Temperature sealing
- Pollutant spillages associated with traffic accidents
- Pollutant spillages from industrial sites
- Landfills

The hydrochemical composition of the groundwater samples taken in Rastatt is dominated by calcium and bi-carbonate, corresponding to the high degree of calcite in the Quaternary sediments of the Upper Gravel layer. The relationships between the major ions have been displayed in a Piper plot (Fig. 3-5). A Piper plot is used to identify groups of groundwater samples with similar evolution or aquifer systems. In a Piper plot, the analysis of any mixture of waters appears in a straight line. Due to the homogenous structure of the Upper Gravel layer, all samples from Rastatt plot close to each other. Elevated concentrations of chloride, sulphate, sodium, potassium and magnesium point towards unspecific anthropogenic sources. Considering the different well groups sampled, it can be seen that samples from the sewer focus observation wells plot with slightly elevated sodium and potassium values in the diagram. However, no clear separation between the well groups is visible in the plot. Individual wells belonging to the well groups are influenced by different anthropogenic sources. Wells outside the city area are affected by agricultural practices including fertiliser application. Individual wells in the city area are affected by contaminated sites with old spillages. Additional interpretations using Piper plots are specified in Eiswirth (2002).

Tab. 3-3: Selected hydrochemical parameters related to well groups (0 = out of town, 1 = city limits, 2 = urban background, 3 = sewer focus) and identified trends (++ = straight increase from group 0 toward group 3, + = increase from group 0 towards group 3 with one exception, o = ambiguous pattern, - = inverse correlation). Data from own samplings 2001–2005.

| Well Group | Category | Distance Nearest Sewer [m] | Distance Upstream Sewer [m] | Cond [µS/cm] | Temp [°C] | Diss. $O_2$ [mg/l] | Na [mg/l] | K [mg/l] | Cl [mg/l] | NO3 [mg/l] | PO4 [mg/l] | SO4 [mg/l] | NH4 [mg/l] | B [mg/l] |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **0** | Arith. Mean | 841.7 | 1116.7 | 484.9 | 11.1 | 3.1 | 10.5 | 2.1 | 17.8 | 27.2 | 0.8 | 31.1 | 0.06 | 0.032 |
| **1** | | 44.0 | 257.3 | 556.8 | 12.5 | 1.0 | 14.5 | 3.3 | 19.8 | 5.1 | 0.7 | 34.5 | 0.16 | 0.056 |
| **2** | | 14.8 | 47.8 | 489.2 | 13.1 | 1.9 | 13.3 | 4.7 | 14.1 | 10.0 | 0.8 | 30.4 | 0.14 | 0.061 |
| **3** | | 2.6 | 10.2 | 706.9 | 14.0 | 1.2 | 23.1 | 7.3 | 30.5 | 12.0 | 1.1 | 38.1 | 0.66 | 0.082 |
| **0** | Median | 875.0 | 1000.0 | 424.0 | 11.2 | 2.3 | 9.1 | 1.2 | 15.7 | 18.6 | 0.7 | 31.2 | 0.03 | 0.024 |
| **1** | | 24.0 | 270.0 | 585.0 | 12.4 | 0.6 | 11.3 | 2.9 | 17.2 | 4.8 | 0.7 | 32.1 | 0.08 | 0.037 |
| **2** | | 9.0 | 37.0 | 499.6 | 13.2 | 2.0 | 13.0 | 3.9 | 14.8 | 7.9 | 0.7 | 33.5 | 0.04 | 0.060 |
| **3** | | 2.0 | 3.0 | 622.5 | 14.1 | 0.8 | 20.7 | 6.1 | 26.2 | 8.6 | 1.0 | 37.5 | 0.38 | 0.093 |
| **0** | Min | 410.0 | 1000.0 | 232.6 | 9.9 | 0.1 | 5.9 | 1.1 | 8.3 | 1.6 | 0.7 | 11.5 | 0.03 | 0.015 |
| **1** | | 7.0 | 13.0 | 216.5 | 9.8 | 0.1 | 7.4 | 1.5 | 6.9 | 0.2 | 0.7 | 11.6 | 0.02 | 0.018 |
| **2** | | 1.5 | 1.5 | 193.1 | 8.9 | 0.4 | 7.8 | 1.6 | 8.1 | 1.3 | 0.5 | 13.0 | 0.02 | 0.036 |
| **3** | | 1.0 | 1.8 | 462.2 | 12.6 | 0.4 | 11.3 | 3.1 | 11.7 | 1.4 | 0.7 | 16.1 | 0.01 | 0.036 |
| **0** | Max | 1150.0 | 1700.0 | 723.0 | 12.1 | 7.4 | 22.0 | 5.6 | 28.4 | 70.1 | 1.3 | 55.3 | 0.13 | 0.076 |
| **1** | | 90.0 | 620.0 | 876.0 | 19.1 | 3.4 | 33.2 | 10.3 | 53.9 | 8.9 | 1.0 | 73.0 | 0.78 | 0.231 |
| **2** | | 50.0 | 190.0 | 649.0 | 14.6 | 3.6 | 21.3 | 13.6 | 21.4 | 26.9 | 1.6 | 43.3 | 1.14 | 0.109 |
| **3** | | 8.3 | 80.0 | 1052.7 | 15.1 | 6.2 | 38.5 | 16.3 | 53.6 | 54.1 | 2.3 | 73.1 | 2.71 | 0.143 |
| **0** | Measure-ments considered | 6 | 6 | 31 | 34 | 30 | 30 | 30 | 20 | 18 | 6 | 19 | 23 | 27 |
| **1** | | 22 | 22 | 35 | 35 | 30 | 35 | 35 | 35 | 31 | 12 | 35 | 15 | 27 |
| **2** | | 23 | 23 | 88 | 87 | 74 | 82 | 82 | 72 | 63 | 28 | 72 | 58 | 67 |
| **3** | | 14 | 14 | 137 | 138 | 121 | 121 | 121 | 79 | 74 | 37 | 78 | 102 | 111 |
| **0** | Increase towards focus wells | Mean | | ++ | ++ | - | + | ++ | + | o | o | o | + | ++ |
| **1** | | Median | | + | ++ | o | ++ | ++ | + | o | o | ++ | + | ++ |
| **2** | | Min | | + | + | o | ++ | ++ | o | o | o | ++ | - | ++ |
| **3** | | Max | | + | o | o | o | ++ | o | o | + | o | ++ | + |
| **0** | Focus above average [%] | Mean | | 38.5 | 14.4 | -38.9 | 80.5 | 116.0 | 77.3 | -14.6 | 42.6 | 19.0 | 446.5 | 65.2 |
| **1** | | Median | | 23.8 | 15.1 | -53.3 | 86.4 | 128.7 | 65.2 | -17.7 | 46.0 | 16.4 | 616.8 | 131.7 |
| **2** | | Min | | 115.9 | 32.2 | 103.0 | 59.6 | 122.7 | 51.4 | 35.1 | 10.0 | 34.2 | -52.6 | 54.2 |
| **3** | | Max | | 40.5 | -1.1 | 28.3 | 51.1 | 66.0 | 55.1 | 53.3 | 82.0 | 27.8 | 296.6 | 3.3 |

The comparison between the different well groups is listed in Tab. 3-3 for selected parameters of the sampling programme. It can be seen that the focus observation wells (Well Group 3) exhibit significantly elevated values compared to the average of the other three well groups for the parameters electrical conductivity, temperature, sodium, chloride, potassium, phosphate, ammonium and boron. The highest contrast can be observed for ammonium (+ 446.5 % in the arithmetic mean) potassium (+ 116 % in the arithmetic mean), sodium (+ 80.5 % in the arithmetic mean), chloride (+ 77.3 % in the arithmetic mean) and boron (+ 65.2 % in the arithmetic mean). An inverse relationship is found for dissolved oxygen. Nitrate concentrations are lower in the urban wells, most probably due to the influence of fertilizer application on rural areas.

The data set can be further analysed with respect to the straightness of the increase from rural area to sewer focus observation wells. A straight increase is given if the concentrations at the sewer focus wells (group 3) exceed the concentrations of the city background wells (group 2) which exceed the concentrations of the outskirts wells (group 1) which themselves are above the concentrations of the rural background wells (group 0). A straight increase in both arithmetic mean and median values has only been observed for potassium, boron and temperature.

The marked contrast between the sewer focus group and the urban background group shows that the quality deterioration is connected to large defects rather than to the whole network. However, the elevated concentrations of the urban background group compared to the rural group demonstrates the diffuse pollution originating from the superposition of a large number of leaks.

For further analysis of selected parameters, scatter plots have been prepared. As an example the correlation between sodium and chloride is shown in Fig. 3-6. It shows a reasonable good correlation which is typical for constituents which are enriched by the same sources. It is obvious that the wells in the rural areas ("out of town") are characterised by lower concentrations whereas the sewer focus wells plot widely distributed over the graph. Surprisingly, wells located at the city limits exhibit a stronger scattering than the urban background wells. Typical urban sources for sodium and chloride are road salting and sewage. Chloride may also enter via fertilisers which contain potassium and chloride and which are applied both to private gardens and agricultural areas. A natural source of sodium and chloride would be the dissolution of evaporates, which are however not present in the sediments of the Upper Gravel Layer.

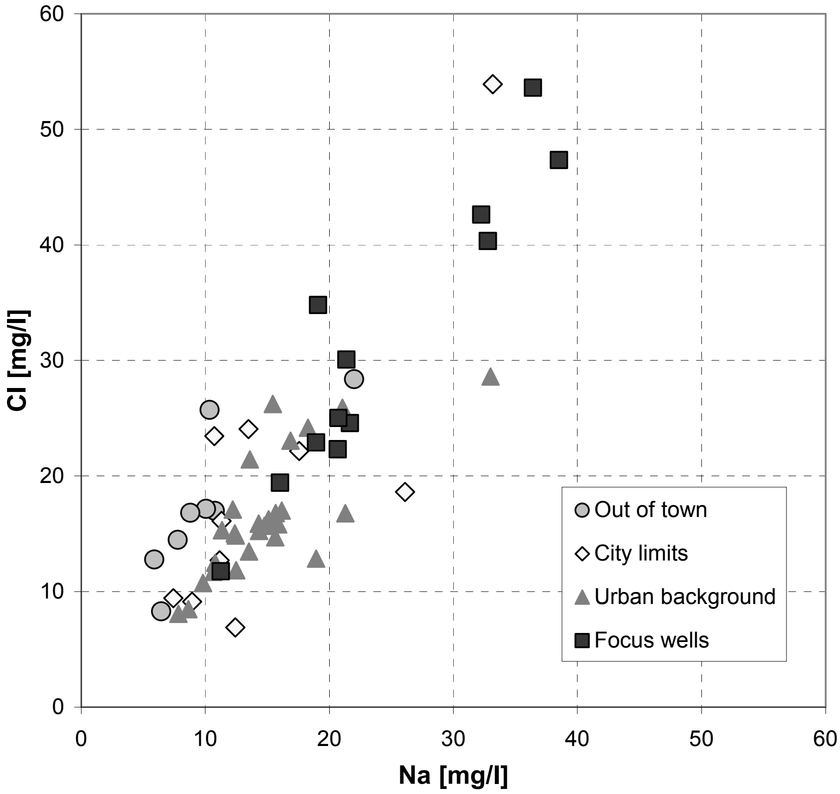



Fig. 3-6: Sodium/chloride scatter plot for average concentrations at wells in the Upper Gravel Layer 2001–2005.

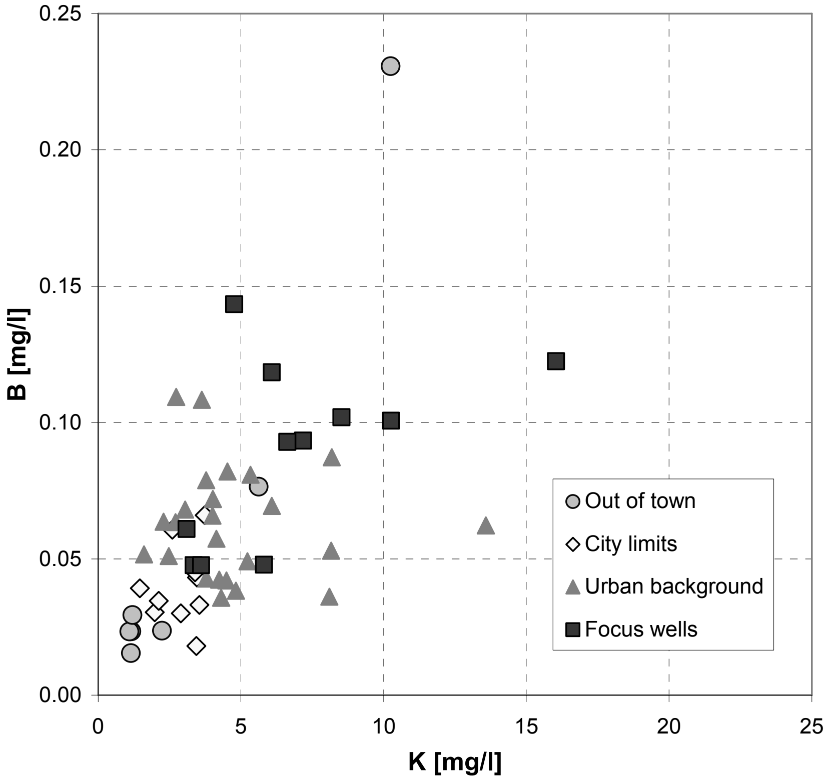



Fig. 3-7: Potassium/boron scatter plot for average concentrations at wells in the Upper Gravel Layer 2001-2005.

Potassium and boron, which exhibit clear tendencies between the different well groups regarding their average and median values are plotted in Fig. 3-7. While it can be observed that the sewer focus wells are characterised by elevated concentrations of the two constituents, there is also a significant number of samples which does not follow this pattern. The scatter plot shows the large bandwidth of possible concentrations. Unlike the scatter plot for sodium and chloride, which showed a rather good correlation between the two parameters, the potassium-boron scatter plot shows that there is only a minor connection between high potassium values and high boron values. Obviously, different processes and sources must contribute to the potassium and boron concentrations.

Potassium is a major constituent of fertilisers and occurs frequently in elevated concentrations beneath agricultural areas (Eiswirth, 2002). Other sources are seepage water from landfills or sewer leakage.

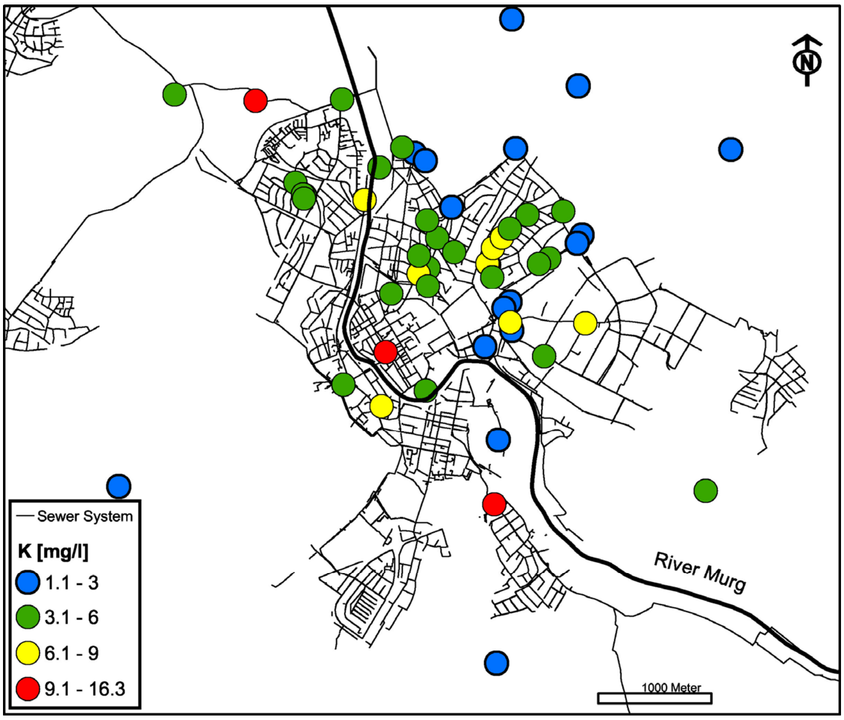



Fig. 3-8: Potassium (in mg/l) in groundwater of the Upper Gravel layer in Rastatt. Averages based on 302 samples from 54 wells taken between 2001 and 2005.

Even though potassium is also a common ingredient in fertilisers, the spatial distribution of potassium concentrations (Fig. 3-8) shows that the concentrations on the agricultural land outside the city remain generally low. However, there are two exceptions: The well GWM-Rheinau-Nord and the well BK 1/355. Observation well BK 1/355 does not show any other sign of sewage influence and it is proposed that fertiliser application on the adjacent sports field is responsible for the high potassium concentrations. Well GWM

Rheinau-Nord does also exhibit high concentrations of boron, but the source of the generally high mineralization of the groundwater at this well is not known yet.

While Boron can also be present in some fertilizers, the major source in the urban environment are detergents which enter the urban aquifer chiefly via leaky sewers. Additional sources of boron occur through industrial production processes.

Groundwater samples showed boron concentrations in the range of 0.01-0.23 mg/l with a mean concentration of 0.06 mg/l. Fig. 3-9 shows a histogram of the boron contents. Concentrations at the outskirts of the city area tend to be below 0.03 mg/l boron.

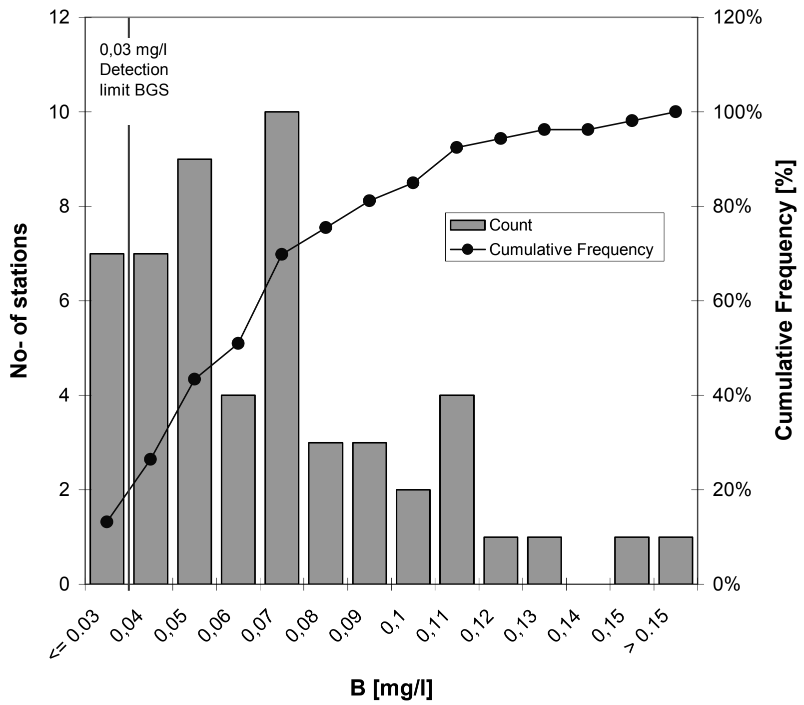



Fig. 3-9: Histogram of the measured Boron concentrations in Rastatt based on 284 samples taken from 53 different stations between 2001 and 2005. Detection limit is at 0.03 mg/l.

With respect to the spatial distribution, there is a clear tendency towards elevated boron concentrations beneath the city centre (Fig. 3-10) which is most likely caused by exfiltrating wastewater. However, the highest concentration of boron was found in a well close to a former industrial spillage. At this spot it is not possible to distinguish between influence from leaky sewers and influence from the industrial spillage.

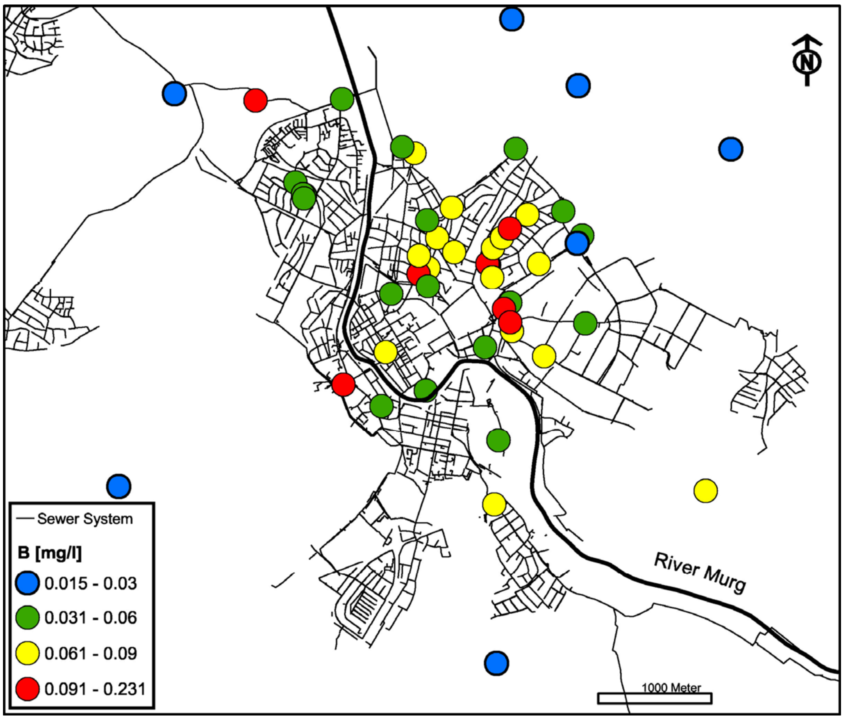



Fig. 3-10: Boron (in mg/l) in groundwater of the Upper Gravel layer in Rastatt. Averages based on 258 samples from 54 wells taken between 2001 and 2005.

Another good indicator for sewage influence seems to be ammonium. Ammonium is usually transformed rapidly into nitrate and occurs therefore only in close proximity to the sources. Significant ammonium concentrations have been found predominantly in the sewer focus wells as indicated in Fig. 3-11 and Tab. 3-3. The effect is also known from other wastewater infiltration studies (MacQuarrie & Sudicky, 2001; MacQuarrie *et al.*, 2001). Elevated ammonium concentrations are well known from contaminated sites where microbiological processes are degrading hydrocarbon pollution (Schilling, 2002).

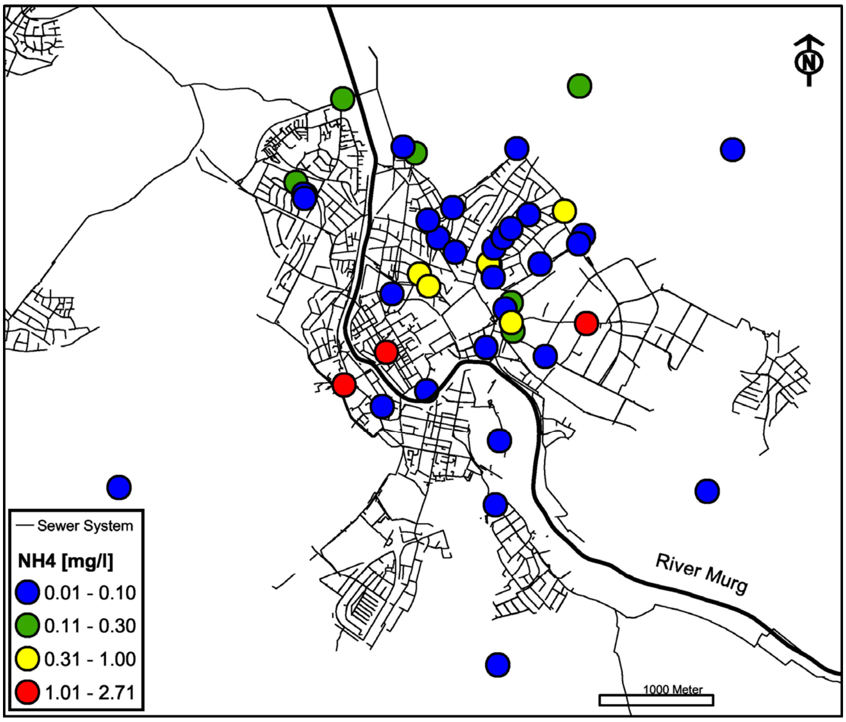



Fig. 3-11: Ammonium (in mg/l) in groundwater of the Upper Gravel layer in Rastatt. Averages based on 218 samples from 51 wells taken between 2001 and 2005.

#### 3.4.2.2 Surface water

In the surface water of the river Murg boron was found in concentrations between 12 µg/L (before passing the city) and 14 µg/L (downstream of the city) in measurements at 7th of February 2002. However, boron concentrations from the database of the environmental agency at Steinmauern three kilometres downstream of Rastatt are significantly higher (see Fig. 3-12). This is most likely due to the position of the sampling point close to the outlet of the Rastatt wastewater treatment plant into the river Murg.

The low concentrations measured upstream of the wastewater treatment plant demonstrate that an infiltration of surface water from the river Murg into the ground water is not a major source of boron for the urban aquifer.

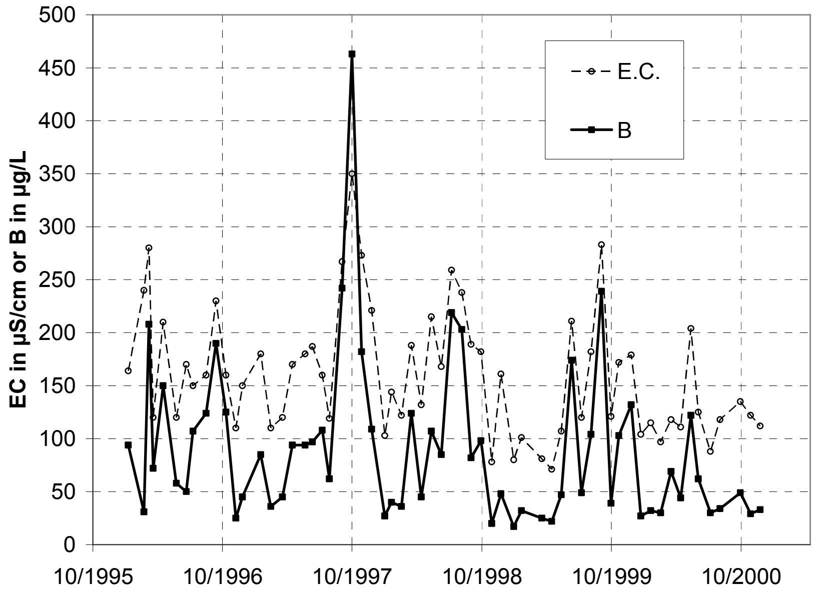



Fig. 3-12: Boron concentrations and specific electrical conductivity in the Murg River 3 km downstream of Rastatt (data source: LfU).

Boron concentrations in the inflow of the wastewater treatment plant in Rastatt range between 0.690 mg/l (7$^{th}$ of February 2002) and 2.480 mg/l (5$^{th}$ of March 2003). This corresponds to (Metzner *et al.*, 1999; Vengosh *et al.*, 1999), who stated that typical concentrations of boron in treated wastewater are around 1 mg/l. Only a small fraction (<10 %) of boron is removed during the passage through a wastewater treatment plant (Metzner *et al.*, 1999).

Apart from the elevated boron concentrations in the wastewater, high boron concentrations are also reported in groundwater subject to hydrothermal influence, evaporative concentration, dissolution of evaporates, the presence of residual seawater or mineral weathering. For instance, desorption of boron from mineral surface as freshwater flushing displaces saline waters in aquifers has led to high concentrations up to 2.1 mg/l in Bangladesh (Ravenscroft & McArthur, 2004).

#### 3.4.2.3 Wastewater

Within the AISUWRS project an intensive monitoring campaign of sewage and groundwater was conducted with the aim to correlate temporal fluctuations in SEC in the groundwater of the focus observation wells with the sewage characteristics, on-line monitoring of pH, SEC and temperature was performed in sewage (Danziger Strasse, Ottersdorfer Strasse, Zaystrasse) and in groundwater (Danziger Strasse, Zaystrasse) a for a period of 10 days (13.10.-23.10.2003).

Tab. 3-4: Summary of hydrochemical characteristics of the sewage sampled in Rastatt.

| Sampling Point | | $Ca^{2+}$ | $Mg^{2+}$ | $K^+$ | $Na^+$ | $F^-$ | $Cl^-$ | $NO_2$ | $NO_3^-$ | $PO_4^{3-}$ | $SO_4^{2-}$ | B |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | mg/l | mg/l | mg/l | mg/l | mg/l | mg/l | mg/l | mg/l | mg/l | mg/l | mg/l |
| Danziger Str. | Arith.Mean | 80.0 | 8.6 | 20.7 | 192.6 | 4.44 | 238.2 | 63.0 | 8.0 | 20.1 | 44.5 | 0.57 |
| Kastanienweg | | 89.0 | 6.6 | 18.7 | 88.6 | - | 68.3 | - | 4.8 | 7.4 | 62.0 | 1.93 |
| Kehler Str. | | 88.4 | 11.3 | 31.9 | 138.8 | - | 152.1 | - | 1.8 | 11.7 | 55.5 | - |
| Ottersdorfer Str. | | 74.2 | 8.4 | 18.6 | 68.5 | 1.90 | 45.6 | 21.2 | 6.1 | 11.4 | 34.2 | 0.63 |
| ZayStr. | | 59.5 | 6.5 | 22.2 | 75.9 | 1.06 | 34.2 | 33.8 | 6.0 | 11.9 | 28.9 | 0.91 |
| WWTP | | 94.6 | 11.5 | 23.6 | 92.9 | - | 127.3 | - | - | 15.5 | 54.7 | - |
| Danziger Str. | Standard Deviation | 29.4 | 3.3 | 8.7 | 108.8 | 1.56 | 170.1 | 28.7 | 3.8 | 12.3 | 18.6 | 0.37 |
| Kastanienweg | | 8.2 | 4.7 | 4.6 | 8.7 | | 32.8 | | 5.8 | 3.8 | 42.6 | 0.57 |
| Kehler Str. | | 5.4 | 1.9 | 9.7 | 77.1 | | 151.2 | | 0.1 | 2.1 | 12.3 | |
| Ottersdorfer Str. | | 34.4 | 4.8 | 13.6 | 50.3 | 0.07 | 38.2 | 13.0 | 3.0 | 7.5 | 21.6 | 0.24 |
| ZayStr. | | 26.2 | 3.3 | 14.5 | 42.0 | 0.11 | 20.3 | 42.7 | 10.1 | 9.5 | 17.9 | 0.84 |
| WWTP | | - | - | - | - | - | - | - | - | - | - | - |
| Danziger Str. | Measure-ments considered | 28 | 28 | 28 | 28 | 7 | 23 | 12 | 12 | 23 | 23 | 24 |
| Kastanienweg | | 3 | 3 | 3 | 3 | 0 | 3 | 0 | 3 | 2 | 3 | 2 |
| Kehler Str. | | 4 | 4 | 4 | 4 | 0 | 4 | 0 | 2 | 4 | 4 | 0 |
| Ottersdorfer Str. | | 14 | 14 | 14 | 14 | 11 | 12 | 8 | 10 | 12 | 12 | 23 |
| ZayStr. | | 13 | 13 | 13 | 13 | 10 | 12 | 8 | 7 | 12 | 12 | 20 |
| WWTP | | 1 | 1 | 1 | 1 | 0 | 1 | 0 | 0 | 1 | 1 | 0 |
| Total Average | | 80.9 | 8.8 | 22.6 | 109.5 | 2.5 | 111.0 | 39.3 | 5.3 | 13.0 | 46.6 | 1.0 |
| Max | | 94.6 | 11.5 | 31.9 | 192.6 | 4.4 | 238.2 | 63.0 | 8.0 | 20.1 | 62.0 | 1.9 |
| Min | | 59.5 | 6.5 | 18.6 | 68.5 | 1.1 | 34.2 | 21.2 | 1.8 | 7.4 | 28.9 | 0.6 |
| Stand.Deviaton | | 12.8 | 2.2 | 4.9 | 47.5 | 1.8 | 77.6 | 21.4 | 2.3 | 4.3 | 13.1 | 0.6 |

Besides the multi-parameter monitoring, sewage samples were taken with automatic samplers in regular intervals for chemical analyses. In all three sewers, a flowmeter was temporarily installed for sewage flow measurements in the manhole closest to the groundwater monitoring well. All three sewers are combined sewers with concrete pipes. The sewers Zaystrasse and Ottersdorfer are DN300, whereas the Danziger Strasse is a DN600.

The results of the sampling campaign are summarized in Wolf et al (2005a). In order to provide a better characterisation of the sewage with respect to the concentrations of major elements, the results of all available sewage analyses have been compiled in **Tab. 3-4**. It is apparent that the variation between different sub-catchments is limited for calcium but pronounced for boron or chloride. The average concentrations serve as input to the modelling exercises in the following chapter.

### 3.4.3 Heavy metals

Heavy metals such as heavy metals like cadmium, chromium, copper, lead, nickel and zinc occur in elevated concentrations in most wastewaters. They may occur in toxic concentrations especially if industrial wastewater discharges are present. Laboratory experiments conducted with soil columns detected heavy metal concentrations exceeding national guideline values (Stögbauer, 2004) in selected samples.

A limited number of groundwater samples from Rastatt were screened for common heavy metals as displayed in Tab. 3-5. However, as the mobility of heavy metals in groundwater is very low, also the detected concentrations ranged far below the guideline values of the WHO (2004). No clear relationships or trends can be seen among the different well groups. In contrast to the elevated concentrations of major ions, the heavy metal concentrations in the sewer focus wells are even below the urban background concentrations.

The observed values are supported by literature sources (Hagendorf, 1996; Hagendorf, 2004; Stögbauer, 2004) which report that the enrichment of heavy metals due to sewage exfiltration is limited to the first decimetres of soil passage.

Tab. 3-5: Summary of heavy metal concentrations according to different well groups (0 = out of town, 1 = city limits, 2 = urban background, 3 = sewer focus) and identified trends (++ = straight increase from group 0 toward group 3, + = increase from group 0 towards group 3 with one exception, o = ambiguous pattern, - = inverse correlation).

| Well Group | Category | Distance Nearest Sewer | As | Cd | Cu | Cr | Ni | Pb | Zn |
|---|---|---|---|---|---|---|---|---|---|
| **0** | Arith. Mean | 842 | 2.35 | 0.54 | 6.59 | 0.87 | 3.43 | 3.17 | 3.30 |
| **1** | | 44.0 | 2.77 | 1.34 | 5.85 | 1.90 | 1.31 | 1.14 | 4.59 |
| **2** | | 14.8 | 2.25 | 0.63 | 3.34 | 1.75 | 1.91 | 1.27 | 4.52 |
| **3** | | 2.6 | 1.14 | 0.05 | 2.81 | 0.65 | 2.20 | 0.93 | 3.74 |
| **0** | Median | 875 | 1.64 | 0.41 | 6.59 | 0.71 | 3.99 | 1.21 | 3.46 |
| **1** | | 24.0 | 1.91 | 1.15 | 4.43 | 1.46 | 1.08 | 0.98 | 2.93 |
| **2** | | 9.0 | 2.08 | 0.68 | 2.94 | 1.78 | 1.09 | 1.24 | 4.23 |
| **3** | | 2.0 | 1.00 | 0.06 | 2.81 | 0.20 | 0.43 | 1.00 | 1.13 |
| **0** | Min | 410 | 1.00 | 0.04 | 5.36 | 0.11 | 0.29 | 0.51 | 1.99 |
| **1** | | 7.0 | 1.00 | 0.02 | 2.36 | 0.73 | 0.84 | 0.80 | 1.76 |
| **2** | | 1.5 | 1.00 | 0.01 | 1.28 | 0.02 | 0.26 | 0.01 | 0.29 |
| **3** | | 1.0 | 1.00 | 0.01 | 2.81 | 0.02 | 0.42 | 0.00 | 0.76 |
| **0** | Max | 1150 | 5.12 | 1.30 | 7.82 | 1.93 | 6.00 | 9.74 | 4.44 |
| **1** | | 90.0 | 7.76 | 3.72 | 9.85 | 5.52 | 2.09 | 1.75 | 14.7 |
| **2** | | 50.0 | 4.35 | 1.32 | 6.45 | 5.06 | 8.35 | 2.98 | 19.1 |
| **3** | | 8.3 | 2.12 | 0.06 | 2.81 | 5.04 | 5.75 | 2.33 | 9.32 |
| **0** | Measure-ments conside-red | 6 | 6 | 7 | 2 | 7 | 3 | 7 | 3 |
| **1** | | 22 | 16 | 20 | 9 | 22 | 15 | 22 | 14 |
| **2** | | 23 | 31 | 41 | 17 | 41 | 27 | 40 | 26 |
| **3** | | 14 | 10 | 12 | 1 | 12 | 3 | 11 | 3 |
| | Increase towards focus wells | Mean | o | o | o | o | o | - | o |
| | | Median | o | o | o | o | o | o | o |
| | | Min | o | o | o | o | o | o | o |
| | | Max | - | o | o | o | o | o | o |
| | Focus above average [%] | Mean | -53.6 | -94.4 | -46.5 | -57.0 | -0.7 | -49.8 | -9.6 |
| | | Median | -46.7 | -92.0 | -39.5 | -84.8 | -79.0 | -12.7 | -68.1 |
| | | Min | 0.0 | -56.1 | -6.3 | -93.0 | -9.9 | -99.3 | -43.5 |
| | | Max | -63.1 | -97.2 | -65.0 | 20.7 | 4.9 | -51.8 | -26.8 |
| WHO Guideline (2003) | | | 10.00 | 3.00 | 2000 | 50.00 | 20.00 | 10.00 | nil |

### 3.4.4 EDTA

The chelating agent EDTA (ethylenediaminetetraacetic acid) is a compound of massive use world wide with household and industrial applications, being one of the anthropogenic compounds with highest concentrations in inland European waters. (Oviedo & Rodríguez, 2003). It is a powerful complexing agent of metals and a highly stable molecule, offering a considerable versatility in industrial and household uses (Sinex, 2004). Since it is applied predominantly in aqueous medium, it is released into the environment through wastewaters. EDTA´s main use is in detergents, where it acts as a builder (chelates metals) especially as a replacement for phosphates, a major nutrient in wastewater. However, a problem with EDTA is its inability to biodegrade in the environment. EDTA is found in many natural waters and occurs at higher levels in wastewater effluents. Western European countries have banned the use of EDTA in detergents (Sinex, 2004). Concerns about the ecotoxicological effects of EDTA are reported concerning the mobilization of heavy metals and the possible contribution of EDTA to eutrophication water processes.

Tab. 3-6: Industrial and household uses of EDTA (Oviedo & Rodríguez, 2003).

| Use | % of world market |
|---|---|
| Detergents | 33 |
| Water treatment | 18 |
| Pulp and Paper Industry | 13 |
| Photography | 5 |
| Metal Cleaning | 5 |
| Cosmetics, foodstuffs, pharmaceuticals | 5 |
| Agrochemicals | 4 |
| Textile Industry | 4 |
| Printing inks | 3 |
| Concrete admixtures | 2 |
| Miscellaneous | 8 |

Within a final sampling round conducted on the 9$^{th}$ of May 2005 also the suitability for EDTA as a marker species for sewage was assessed. 17 samples were analysed at the Environmental Research Centre with a detection limit of 0.2 µg/l. The focus of the sampling campaign was placed on wells close to known sewer defects. Only few wells outside the city area were sampled for reference purposes.

Tab. 3-7 shows the results of the campaign in comparison with boron concentrations and electrical conductivity. While EDTA was generally found in elevated concentrations at the sewer focus wells (focus group 3), the samples from the sewer test site Rastatt Danziger Strasse exhibit very low EDTA concentrations. The correlation with boron concentrations and electrical conductivities is weak.

Tab. 3-7: Results of the EDTA sampling campaign in comparison with averages of boron and electrical conductivity.

| Observation Well | Focus Group | Next Sewer [m] | Next Upstream Sewer [m] | EDTA [µg/l] | B [mg/l] | Cond [µS/cm] |
|---|---|---|---|---|---|---|
| 0021/211-6 | 0 | 750 | 1000 | 0.5 | 0.08 | 233 |
| 152/211-3 | 0 | 1150 | 1700 | 1.4 | 0.02 | 387 |
| GWMS N-Rheinau | 0 | 60 | 270 | < 0.2 | 0.23 | 876 |
| 171/211-0 | 1 | 90 | 350 | 0.5 | 0.02 | 602 |
| BK 1/102 | 2 | 5 | 6 | 1.2 | 0.05 | 193 |
| Elf P1 | 2 | 10 | 25 | < 0.2 | 0.08 | 538 |
| UWA 1 | 2 | 7 | 40 | 2.3 | 0.07 | 579 |
| D1 | 3 | 2.2 | 80 | < 0.2 | 0.12 | 959 |
| D2 | 3 | 8.3 | 9.2 | 0.5 | 0.09 | 712 |
| D3 | 3 | 2 | 2 | < 0.2 | 0.10 | 1053 |
| D4 | 3 | 3 | 3 | < 0.2 | 0.10 | 1010 |
| Gartenstr. | 3 | 2 | 2 | 1.2 | 0.09 | 592 |
| Ottersdorfer Sr. | 3 | 1 | 8 | 3.3 | 0.14 | 815 |
| Rauentaler/Lochfeldstr. | 3 | 1.8 | 1.8 | 7.1 | 0.12 | 581 |
| Wussler | 3 | 2 | 2 | 4.2 | 0.06 | 462 |
| Zaystr. | 3 | 3.5 | 3.5 | 2.4 | 0.05 | 511 |

The comparison of the average concentrations of EDTA in the different well groups shows that the wells outside or at the fringe of the urban area (focus groups 0 and 1) exhibit a mean concentration of 0.67 µg/l EDTA. For the urban background (focus group 2) an average of 1.2 µg/l EDTA was measured. Finally the sewer focus wells (focus group 3) exhibited an average of 2.14 µg/l EDTA. This trend suggests that EDTA concentrations are linked to sewage exfiltration. It is remarkable that EDTA still occurs in elevated concentrations even though its use in detergents has been banned in Europe. The inclusion of EDTA in future sampling programmes is therefore recommended.

### 3.4.5 Pharmaceutical residues

Pharmaceutical residues were found in many German rivers and locally also in groundwater (Sacher & Brauch, 2001). In Baden-Württemberg, 105 groundwater samples were screened for 74 pharmaceutical residues and endocrine disruptors. Pharmaceutical residues were found in ca. one third of the samples (LfU, 2002).

A first sampling campaign during the year 2002 in Rastatt comprised seven groundwater samples, one soil water samples and two wastewater samples. The samples were analysed for 15 pharmaceutical substances.

However, within the 2002 sampling campaign none of the substances were found in the groundwater samples in Rastatt, although significant loads were present in the wastewater (Tab. 3-8). One soil water sample taken from a suction cup (SK7) beneath a sewer leak at the test site Kastanienweg showed bezafibrate concentrations of 440 ng/l but the groundwater of observation well GWM 6 at a distance of 4 m to the suction cup showed no concentrations above the detection limit (20 ng/l). It is possible that the pharmaceutical residues are subject to microbiological decomposition and adsorption effects on their passage through the unsaturated zone. This has also been observed in column experiments (Hua *et al.*, 2003) where the passage through a 125 cm sand column led to a removal of 77 % bezafibrate and similar high elimination rates for other substances. Another explanation might be that the leakage rate is too small and the dilution with the unaffected groundwater is lowering concentrations below the detection limits. No results for pharmaceuticals exist for the new focus observation wells yet.

Corresponding to the results of the LfU screening in 2002, additional pharmaceuticals were investigated in selected wells in Rastatt in June 2005 (Tab. 3-9). Positive findings displayed bold. Values in brackets indicate a detection of the substance which falls slightly below the official detection limit: Ten samples were analysed for "group one" pharmaceuticals, including betablockers and analgesics. Out of the seven groundwater wells sampled, two wells exhibited traces of pharmaceuticals. The betablocker Metoprolol occurred at a concentration of 30 ng/l in the observation well Gartenstrasse. Sotalol, another betablocker, was detected at a concentration of 920 ng/l at the observation well Ottersdorfer Strasse. This is exceeding the highst concentration of 560 ng/l found in the screening of 105 wells in Baden-Württemberg documented by (LfU, 2002).

Tab. 3-8: Pharmaceutical residues detected in groundwater, soil water, and wastewater in Rastatt (12.3.2002); positive findings displayed bold.

| | **Groundwater** | | | | | | | **Soil water** | **Wastewater** | |
|---|---|---|---|---|---|---|---|---|---|---|
| | GWM 6 | GWM 5 | GWM 1 | 112/ 221 | AGIP P1 | Metz P9 | ELF P1 | SK 7 | Inflow WWTP | WW sewer above SK7 |
| Substance | [ng/l] | [ng/l] | [ng/l] | [ng/l] | [ng/l] | [ng/l] | [ng/l] | [ng/l] | [ng/l] | [ng/l] |
| Bezafibrate | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | **440** | **1900** | **6000** |
| Carbama-zepine | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | **42** | - | **970** |
| Clofibric acid | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | **340** | < 50 |
| Diazepam | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | **310** | **120** |
| Diclofenac | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | **260** | **4100** | **8400** |
| Etofibrate | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 50 | < 50 |
| Fenofibrate | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | - | < 50 |
| Fenofibric acid | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | **94** | **740** | **190** |
| Fenoprofen | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 50 | < 50 |
| Gemfibrozil | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | **81** | **190** | < 50 |
| Ibuprofen | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | **120** | **3800** | **2100** |
| Indometacin | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | **220** | **76** |
| Ketoprofen | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 50 | < 50 |
| Naproxen | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | **540** | < 50 |
| Pentoxifyllin | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 20 | < 50 | < 50 |

Tab. 3-9: Pharmaceutical residues in samples from Rastatt (11.6.2005)

| **Substance** | **Groundwater** | | | | | | | **Seepage water** | **Wastewater** | |
|---|---|---|---|---|---|---|---|---|---|---|
| | D 1 | D 2 | D 3 | D 4 | Garten-Str. | Zay-Str. | Ottersdorfer-Str. | Test site Kehler Strasse | Kehler Strasse | WWTP |
| | ng/L | ng/L | ng/L | ng/L | ng/L | ng/L | ng/L | ng/L | ng/L | ng/L |
| ***Analgesics:*** | | | | | | | | | | |
| Phenazon | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Dimethylaminophenazon | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | **150** | < 50 | < 100 |
| Propyphenazon | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| ***Betablocker:*** | | | | | | | | | | |
| Atenolol | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | **450** | **620** | **480** |
| Betaxolol | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Bisoprolol | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Metoprolol | < 10 | < 10 | < 10 | < 10 | **30** | < 10 | < 10 | **1100** | **540** | **500** |
| Pindolol | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Propranolol | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Sotalol | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | **920** | **1200** | **1200** | **620** |
| ***Broncholytics, Secretolytics:*** | | | | | | | | | | |
| Clenbuterol | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Salbutamol | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Terbutalin | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| ***Zytostatics:*** | | | | | | | | | | |
| Ifosfamid | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Cyclophosphamid | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| ***Antibiotics:*** | | | | | | | | | | |
| Azithromycin | | | < 10 | | < 10 | < 10 | < 10 | < 20 | < 50 | (54) |
| Dehydrato-Erythromycin A | | | < 10 | | **(5)** | < 10 | < 10 | **98** | (39) | **240** |
| Clarithromycin | | | < 10 | | < 10 | < 10 | < 10 | **45** | < 50 | **150** |
| Roxithromycin | | | < 10 | | < 10 | < 10 | < 10 | **31** | < 50 | < 100 |
| Clindamycin | | | < 10 | | **(6)** | < 10 | **(8)** | **230** | < 50 | (65) |
| Ronidazol | | | < 10 | | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Metronidazol | | | < 10 | | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Sulfadiazin | | | < 10 | | < 10 | < 10 | < 10 | **21** | < 50 | < 100 |
| Sulfamerazin | | | < 10 | | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Furazolidon | | | < 10 | | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Sulfadimidin | | | < 10 | | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Sulfamethoxazol | | | < 10 | | < 10 | < 10 | < 10 | **50** | **170** | **440** |
| Dapson | | | < 10 | | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Trimethoprim | | | < 10 | | < 10 | < 10 | < 10 | **41** | **120** | **190** |
| Amoxicillin | | | < 10 | | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Penicillin G | | | < 10 | | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Penicillin V | | | < 10 | | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Cloxacillin | | | < 10 | | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Oxacillin | | | < 10 | | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Nafcillin | | | < 10 | | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Dicloxacillin | | | < 10 | | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Oleandomycin | | | < 10 | | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Chloramphenicol | | | < 10 | | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Spiramycin | | | < 10 | | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |
| Tylosin | | | < 10 | | < 10 | < 10 | < 10 | < 20 | < 50 | < 100 |

The group of antibiotics did not exhibt any major concentrations in the groundwater wells sampled. Dehydrato-Erythrom and Cliodamicin were identified in the spectrographic analysis, but at concentrations slightly below the official detection limit. The most prominent pharmaceutical components of the wastewater sampled are the betablockers Metoprolol, Atenolol, Sotalol and the antibiotics Dehydrato-Erythromycin, Clarithromycin, Sulfamethoxazol, Trimethoprim. As the comparison with the seepage water from the test site Kehler Strasse (approx. 50 cm soil passage) suggests, the betablockers are less subject to attenuation or degradation than the antibiotics. Together with the lower concentrations of antibiotics in the sewage, this is responsible for the lack of positive detects in the groundwater. Surprisingly, no pharmaceutics have been detected in the wells at the monitoring site Danziger Strasse, which are otherwise clearly affected by sewage.

Metoprolol and Sotalol are beta-adrenergic blocking agents (beta-blockers). Metoprolol is prescribed for patients with high blood pressure (hypertension). It is also used to treat chest pain (angina pectoris) related to coronary artery disease. Metoprolol is also useful in slowing and regulating certain types of abnormally rapid heart rates (tachycardias). Other uses for Metoprolol include the prevention of migraine headaches and the treatment of certain types of tremors (http://www.medicinenet.com). Sotalol is used to treat irregular heartbeats. It works by acting on the heart muscle to improve the heart's rhythm. Both drugs are usually prescribed on a regular basis. Therefore a regular input into the wastewater can be expected and also a high total mass flux is present. The prescription practice might be a reason for the detection of Metoprolol and Sotalol in the absence of other substances in the groundwater.

In conclusion it can be stated that pharmaceutics were found with elevated frequency in the Rastatt groundwater samples compared to the state-wide survey undertaken by (LfU, 2002). The findings correspond to other parameters indicating sewer leakage.

At the test site Kehler Strasse it can be seen that the degradation or attenuation of most pharmaceutics is very limited on the 50 cm seepage route. All substances which were identified in the sewage were also recorded in the seepage water in the same dimension. Sometimes the seepage water concentrations have been above the sewage concentrations. This demonstrates that the sewage composition is extremely variable and the seepage water collected at time A does not stem from the sewage water collected at time B (B must be before A). In order to establish a mass balance in this strongly transient system, a denser monitoring scheme is obviously necessary.

### 3.4.6 Iodated X-ray contrast media

In total 114 samples were taken from 46 groundwater observation wells in Rastatt and analysed for iodated X-ray contrast media during the years 2002-2004. Two samples were taken from the river Murg, three wastewater samples were taken at the wastewater treatment plant and at the test site Kastanienweg and one soil water sample was taken from suction cups at the test site Kastanienweg.

Iodated X-ray contrast media have a high potential as an excellent wastewater marker for several reasons:

- The background environmental concentration is zero. Natural sources are not known.
- Typical concentrations in wastewater are 100-1000 times above the detection limit.
- The complexes are very stable and behave conservatively (iodated x-ray contrast media are designed to pass the human body without any interactions)

Concentrations of iodated contrast media in wastewater are extremely variable and within the scope of the research project only a small number of wastewater samples could be taken to account for this variability. The two wastewater samples taken at the inflow of the wastewater treatment plant in Rastatt and the wastewater sample taken from a sewer in a residential area (at the test site Kastanienweg) show significant differences to the findings of (Ternes & Hirsch, 2000) for the wastewater at a Hessian treatment plant. The wastewater in Rastatt is characterised by very high maximum values of 21.000 ng/l iomeprol and up to 5.900 ng/l ioxithalaminic acid. On the other hand, the concentrations recorded for amidotrizoic acid (which is the most widespread substance in the groundwater samples) are below the concentrations measured by Ternes et al. (2003). It is questionable whether the ongoing change in the application of iodated X-ray contrast media, namely the shift from ionic substances toward anionic substances, has already had an effect on the wastewater composition in Rastatt.

Raw influents of the wastewater treatment plant in Rastatt showed concentrations of 520 - 1200 ng/l amidotrizoic acid. This is lower than the 3300 ng/l +/- 700 ng/l amidotrizoic acid measured by Ternes and Hirsch in Hessia and also below the 2500 ng/l amidotrizoic acid measured in the wastewater treatment plant Karlsruhe (Hua *et al.*, 2003). However, amidotrizoic acid concentrations in the part of the sewer system directly draining the municipal hospital in Rastatt are not known. Therefore it remains difficult to assess the degree of groundwater pollution by comparing concentrations of amidotrizoic acid in groundwater and wastewater.

Tab. 3-10: Summary of iodated x-ray contrast media analysis in Rastatt compared to the measurements of Ternes & Hirsch (2000).

| | Groundwater | | | Wastewater | | | Riverwater | |
|---|---|---|---|---|---|---|---|---|
| Substance | Samples total [-] | Positive [-] | Max. [ng/l] | Wastewater (Ternes & Hirsch 2000) [ng/l] | WWT inflow (Rastatt) [ng/l] | WW Residential Area [ng/l] | River Murg upstream Rastatt [ng/l] | River Murg downstram Rastatt [ng/l] |
| Iopamidol | 114 | 4 | 78.8 | 4300 ± 900 | < 100 - 1000 | <10 | 54 | 46 |
| Iopromide | 114 | 2 | 38.9 | 7500 ± 1500 | < 100 - 130 | <10 | <10 | <10 |
| Iomeprol | 114 | 3 | 1655 | 1600 ± 400 | 400 - 21000 | **165** | 34 | 46 |
| Amidotrizoic acid | 114 | 30 | 1703 | 3300 ± 700 | 520 - 1200 | <10 | <10 | <10 |
| Iodipamide | 113 | 0 | < 25 | | | | <10 | <10 |
| Iohexol | 114 | 4 | 187.4 | | | | <10 | <10 |
| Iopanic acid | 22 | 0 | < 10 | | | | <10 | <10 |
| Iothalamic acid | 114 | 10 | 238.2 | 180 ± 100 | < 100 - 930 | <10 | <10 | <10 |
| Ioxalagic acid | 21 | 0 | < 10 | | | | <10 | <10 |
| Ioxitalamic acid | 113 | 10 | 204 | 170 ± 100 | 120 - 5900 | <10 | <10 | <10 |
| Iotrolan | 47 | 0 | < 100 | | | | | |

In the water samples from the River Murg, only iopamidol and iomeprol were detected in maximum concentrations of 54 ng/l. It is therefore assumed for further interpretations that the river Murg is not a major source of iodated x-ray contrast media in Rastatt.

As shown in Tab. 3-10, amidotrizoic acid (synonymous to diatrizoate) shows by far the highest number of positive detects in groundwater. It has been found in about every fourth sample. The highest concentration measured in groundwater was 1703 ng/l amidotrizoic acid at the focus observation well Zaystrasse. The focus observation well Zaystrasse is situated in 5 m distance of a 300 mm diameter sewer with a crack in the bottom section. Within the catchment of the sewer in the Zaystrasse lies also the city hospital of Rastatt, which is a source for high concentrations of iodated contrast media in the sewage.

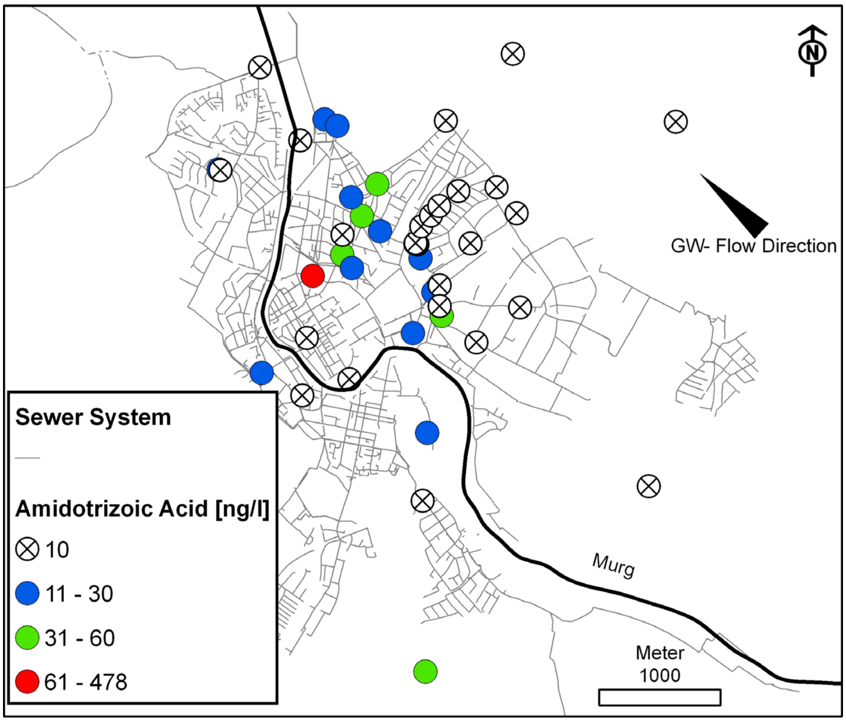



Fig. 3-13: Amidotrizoic acid (in ng/l) in groundwater of the Upper Gravel layer in Rastatt. Mean concentrations over samples from 2002 to 2004.

Both iothalaminic acid and ioxithalaminic acid were positively detected in 10 groundwater samples. In almost all of the wells with positive results, both substances were found. All positive findings are within the city area. Unlike amidotrizoic acid, they have not been detected in the focus observation well Zaystrasse. Therefore it is assumed that the city hospital is not a major source for these substances but that they are used by several free physicians in the city area.

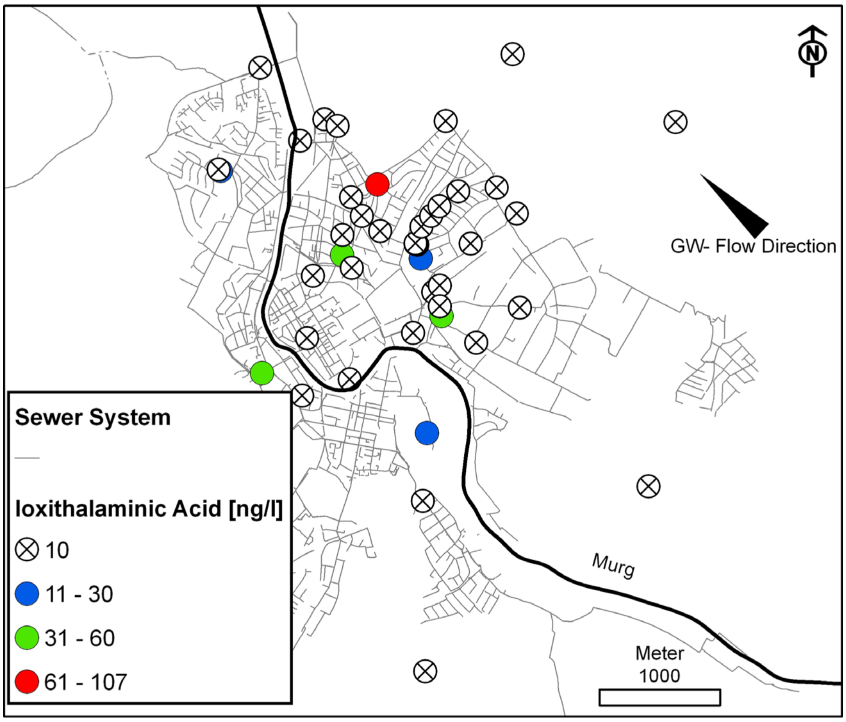



Fig.3-14: Ioxithalaminic acid (in ng/l) in groundwater of the Upper Gravel layer in Rastatt. Mean concentrations over samples from 2002 to 2004.

Very high concentrations of up to 1655 ng/l in groundwater have been found for iomeprol. All positive detects originate from the focus observation well Ottersdorfer Strasse which was drilled within 3 m distance to a defective 300 mm diameter sewer and within 1 m distance to a house connection of unknown condition. The concentrations of iomeprol at the observation well Ottersdorfer Strasse exhibits a strong temporal variation (Fig. 3-15). However, the temporal pattern cannot be correlated with the temporal variations of amidotrizoic acid at the observation well Zaystrasse. It must be concluded, that the concentrations in the sewage as the source are highly variable, depending on the application pattern of iodated x-ray contrast media in the medical services attached to the sewer network. In order to establish an appropriate mass balance, a very high sampling frequency is necessary which was not possible given the analytical effort required.

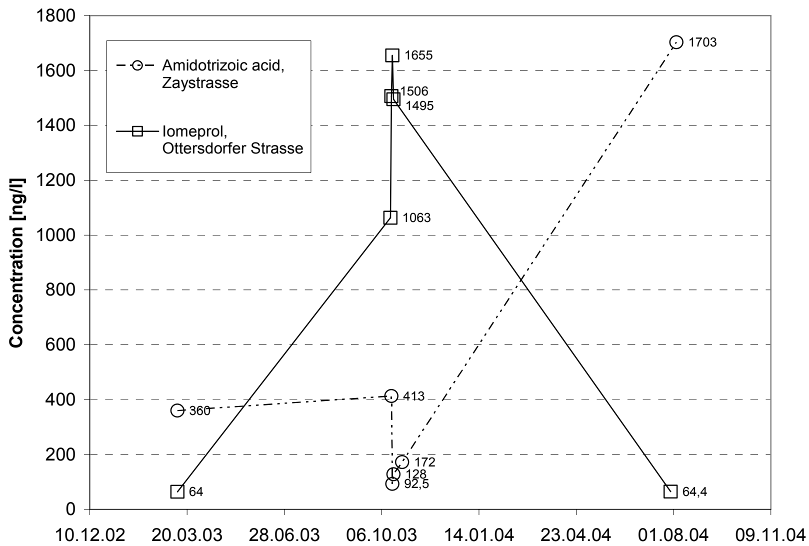



Fig. 3-15: Temporal variations of iodated x-ray contrast media in groundwater samples.

Regarding the spatial distribution of the positive detects of iodated x-ray contrast media in groundwater, it could be noticed that iodated x-ray contrast media are only found in the vicinity of sewage sources. Fig. 3-16 depicts the results of the sampling campaign in October 2003 in relation to the distance to the nearest upstream sewer. It can be seen that no x-ray contrast media have been found beyond a distance of 60 m to the nearest upstream sewer. However, in a following sampling round in 2004, x-ray contrast media were also found at a single well outside the city area (well 0134/211-1). This well has been free of x-ray contrast media in previous samplings and further investigations are needed to control this finding. The well is situated within 50 m from frequently used railway tracks and emissions from train lavatories must be taken into consideration.

In addition to the iodated x-ray contrast media, the wastewater marker species examination in the year 2001 also included gadolinium. Gadolinium is a rare earth element with paramagnetic properties used as a contrast media in computer tomography. It has been successfully measured as a conservative environmental tracer before (Elbaz-Poulichet *et al.*, 2001; Möller *et al.*, 2000). However, no significant anomalies could be detected in the 32 groundwater samples taken during a campaign in 2001. The most likely explanation would be the lack of medical facilities equipped with tomography devices. A more detailed analysis of the results is provided in (Osswald, 2002).

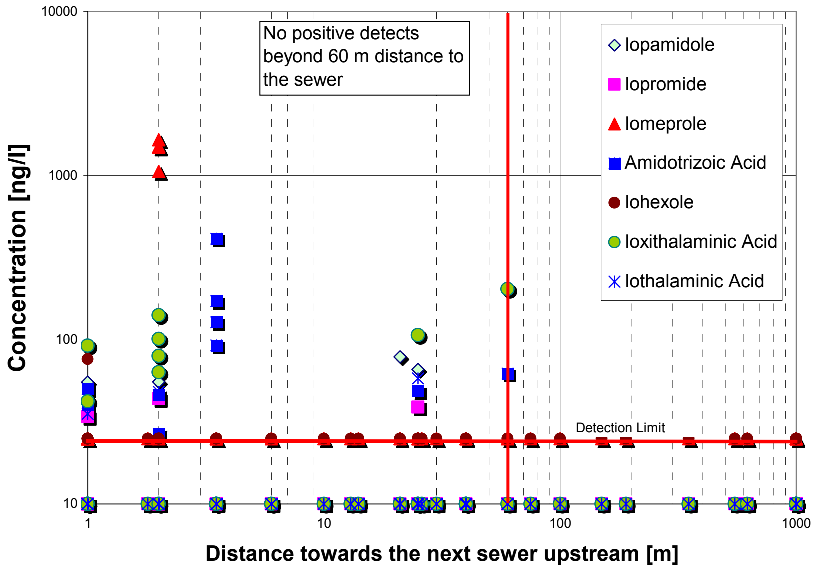



Fig. 3-16: Detections of iodated x-ray contrast media in groundwater samples in relation to the sewer distance.

The spatial distribution of pharmaceutical residues in the sewer system is expected to be very heterogeneous. The sources are private individuals, medical practioners and hospitals. Private consumption can be assumed to be distributed across the town according to population density. Rastatt possesses one large hospital which is located directly in the city centre. Other major sources are the medical ambulances where (i) a high number of sick persons using pharmaceutics are present every day (ii) excessive pharmaceutics are discarded in the drain. Fig. 3-17 displays the distribution of medical practitioners in Rastatt. For this compilation, the 37 practioners listed in the telephone directory were given spatial reference in the GIS and classed according to medical sectors. Most of the medical facilities are concentrated in the city centre west of the castle and close to the pedestrian zone.

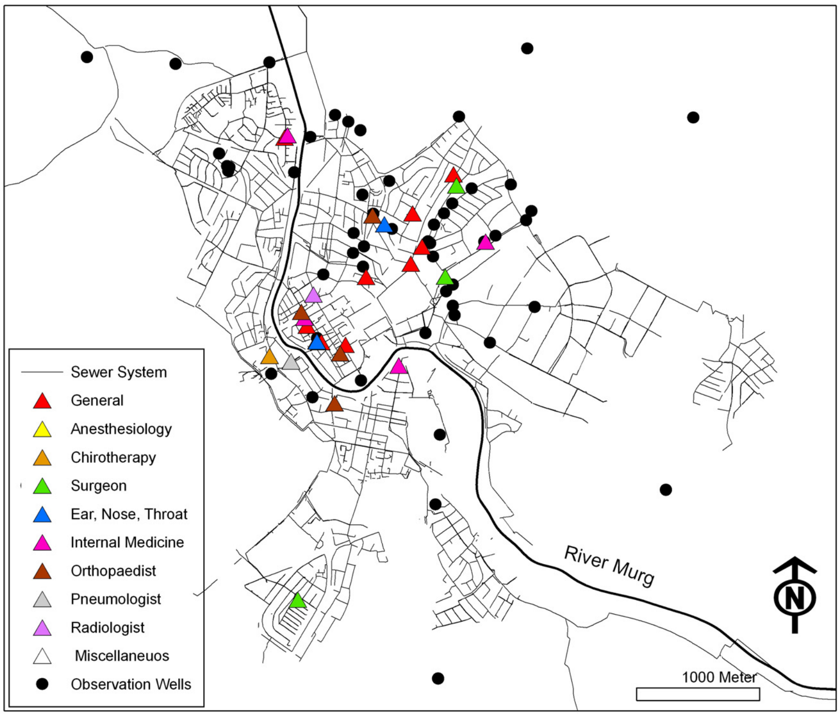



Fig. 3-17: Medical practioners in Rastatt as listed in the telephone directory (2003).

Conclusions:

- Among the pharmaceutical residues, iodated x-ray contrast media were screened in greatest depth in Rastatt (114 samples from 46 wells were analysed)
- Positive detects occurred for amidotrizoic acid (30), iothalaminic acid (10), ioxithalaminic acid (10), iopamidol (4), iohexol (4), iomeprol (3), iopromide (2). The frequency of the positive detects as well as the maximum concentrations exceed the values from the state-wide sampling campaign in Baden-Württemberg (LfU, 2002).
- With only one exception, no x-ray contrast media have been found in wells which have a distance of more than 60 m to the nearest upstream sewer.
- The concentrations in individual wells are subject to strong variations.

- The distribution in the urban aquifer depends not only on the existence of leaks but also on the application of iodated x-ray contrast in medical facilities in the sewer catchment.
- A significant proportion of wastewater is present in the urban aquifer.
- Iodated x-ray contrast media are well suited as a marker species provided that sufficient application in the medical facilities is present.

### 3.4.7 Microbiological parameters

#### 3.4.7.1 Background

Groundwater is a common transmission route for waterborne infectious diseases (Borchardt *et al.*, 2003). Surveillance data since 1981 have shown that approximately half of all waterborne disease outbreaks in the United States were associated with contaminated groundwater (Levy *et al.*, 1998). Several studies on microbiological impacts of wastewater on groundwater are available from research on artificial groundwater recharge schemes and wastewater reuse facilities. Specific investigations on the microbiological groundwater quality beneath urban areas were conducted in the United Kingdom in Nottingham (Barret *et al.*, 1999; Cronin *et al.*, 2003), Birmingham and most recently in Doncaster (Cronin *et al.*, 2005). The studies by Cronin *et al.* (2005) in Doncaster compare the concentrations of faecal indicator organisms in sewage and groundwater. The sampling in Doncaster included total thermotolerant coliforms, *Escherichia coli*, total coliforms, faecal streptococci, sulphite reducing clostridia spores, coliphages and enteric viruses. The high detection rate of sulphite reducing clostridia and faecal streptococci showed their value as markers (Cronin *et al.*, 2005).

In Rastatt, selected wells were screened for microbiological indicator organisms during the years 2003-2005. The results of the samplings during 2003 are documented in (Paul *et al.*, 2004) but not elaborated further as it is assumed that the results were partly influenced by an improper disinfection practice. At the sampling campaigns following October 2004, the entire pumping equipment was sterilised using 70 Vol % ethanol and rinsed with deionised water before the deployment into a new well. Groundwater sampling was performed using submersible pumps. Samples were taken only after the stabilisation of the parameters measured in the field (i.e. electric conductivity, temperature, pH-value, dissolved oxygen). Samples for microbiological analysis were taken and stored in freshly autoclaved 2 l glass bottles. Immediately prior to each sample collection event the bottle caps and the tube of the pump were sterilised in the field using ethanol at a flame of a small camping stove.

It must be kept in mind that the parameters screened for in Rastatt do not allow a complete microbiological risk assessment as viruses are missing in the sampling programme. The WHO (Ashbolt *et al.*, 2001) reports in this context: Numerous epidemiological studies of waterborne illness in developed countries indicate that the common aetiological agents are more likely to be viruses and parasitic protozoa than bacteria (Levy *et al.*, 1998). Given the often lower persistence of vegetative cells of the faecal bacteria compared to the former agents, it is not surprising that poor correlations have been reported between waterborne human viruses or protozoa and thermotolerant coliforms (Kramer *et al.*, 1996). For the German and Suisse national context it is concluded that the knowledge about the origin and transport of pathogenic microorganisms must be deepened in future (Auckenthaler, 2003).

#### 3.4.7.2 *Escherichia coli* and total coliforms

The detection of organisms such as *Escherichia coli* or enterococci is assumed to indicate faecal contamination. Both groups of bacteria are considered to originate from the intestinal tract of humans and a wide variety of warmblooded animals and therefore can be used as indicators of contamination with wastewater. *Escherichia coli* is assumed to survive up to 21 days outside the intestine and thus indicates recent faecal contamination (Carillo et al., 1985). Alexander and Seiler (1985) could detect *Escherichia coli* in wells for more than 150 days, Althaus et al. (1982) performed investigations at different groundwater wells and observed a total elimination of *Escherichia coli* after 9 to 12 months.

The results show that the detection of *Escherichia coli* (Tab. 3-11) and total coliforms (Tab. 3-12) is subject to strong temporal and spatial variations. Problems like the unexpected appearance of indicator organisms which are known from microbiological groundwater sampling in Doncaster (Morris *et al.*, 2005) have also been observed in Rastatt.

5 out of 12 groundwater observation wells exhibit positive detects of *Escherichia coli*. In contrast to the samplings documented in (Paul *et al.*, 2004), all reference wells outside the city area were found to be free from *Escherichia coli*. Of the three urban background wells, only well Elf P1 showed continuously positive detects. Out of the nine sewer focus observation wells, five were found to contain *Escherichia coli*. However, all bacteriological

counts which indicate a strong influence were found at the test site Danziger Strasse.

Tab. 3-11: Results of the screenings for *Escherichia coli*. Well types: 0 = Reference, 1 = City border, 2 = Urban Background, 3 = Focus well

| Sampling Point | Well Type | Lab A 27/10/04 | Lab B 27/10/04 | Lab B 03/02/05 | Lab C 09/05/05 |
|---|---|---|---|---|---|
| | | E.Coli as 1/100ml | | | |
| 0112/211-2 | 0 | 0 | 0 | 0 | |
| 0021/211-6 | 0 | 0 | | 0 | 0 |
| BK1/102 | 2 | 0 | | 0 | |
| UWA 1 | 2 | 0 | | 0 | 0 |
| D1 | 3 | 0 | | 10000 | 8000 |
| D2 | 3 | 0 | | 2500 | 1000 |
| D3 | 3 | 0 | | 0 | 0 |
| D4 | 3 | 0 | | 284 | 20 |
| ELF P1 | 2 | 8 | | 7 | 10 |
| Zaystr. | 3 | 0 | 1 | 0 | |
| Gartenstr. | 3 | 4 | | 8 | |
| Ottersdorfer-Str | 3 | 0 | | 0 | |
| Wussler | 3 | 0 | | 0 | |
| Rauentaler/Lochfeld Str. | 3 | 0 | | 0 | |
| 152/211-3 | 0 | | | | 0 |
| Wastewater Danziger Str | - | | | | 1.E+06 |

No explanation could be found for the negative findings at the sewer focus well D3 for which a direct connection to the wastewater sewer was proven by other hydrochemical tracers (e.g. boron, ammonium, chloride) and online monitoring of water level and electrical conductivity (cf. chapter 3.4.2). Similar to the *Escherichia coli* counts, also the amount of total coliforms is very low at this sampling spot.

In contrast to the focus observation well D3, the focus observation well D1, which showed the highest counts of *Escherichia coli* in two subsequent samplings, is located 3 m upstream of the nearest public sewer. In this case, it must be considered, that a significant influence of leaky house connections is present.

Tab. 3-12: Results of the screenings for total coliforms. Well types:: 0 = Reference, 1 = City border, 2 = Urban Background, 3 = Focus well

| | | Lab A | Lab B | Lab B |
|---|---|---|---|---|
| | | 27/10/04 | 27/10/04 | 03/02/05 |
| | | | Total Coliforms | |
| Sampling Point | Well Type | 1/100ml | 1/100ml | 1/100ml |
| 0112/211-2 | 0 | 0 | 0.2 | 120 |
| 0021/211-6 | 0 | 36 | | 177 |
| BK1/102 | 2 | 5 | | 13000 |
| UWA 1 | 2 | 6 | | 19000 |
| D1 | 3 | >200 | | 33000 |
| D2 | 3 | >200 | | 47000 |
| D3 | 3 | >200 | | 610 |
| D4 | 3 | >200 | | 120000 |
| ELF P1 | 2 | >200 | | 1070 |
| Zaystr. | 3 | 0 | 0.3 | 300 |
| Gartenstr. | 3 | 6 | | 30 |
| Ottersdorfer-Str | 3 | 0 | | 30 |
| Wussler | 3 | 0 | | 4 |
| Rauenta-ler/Lochfeld Str. | 3 | 0 | | 3 |

### 3.4.7.3 Faecal streptococci and Enterococci

While the British literature usually refers to faecal streptococci in contrast to widespread German practice of analysis for enterococci, a short comparison of both terms is given below.

Tab. 3-13: Results of the screenings for enterococci, faecal streps, sulphite reducing clostridia (SRC), Pseudomonas aeruginosa and coliphages. Positive detects displayed bold.

| | | Lab A | Lab B | Lab B | Lab C |
|---|---|---|---|---|---|
| | | 27/10/04 | 27/10/04 | 03/02/05 | 09/05/05 |
| | | | Enterococci | | Fecal Streps |
| Sampling Point | Well Type | | 1/100ml | | |
| 0112/211-2 | 0 | 0 | 0 | 0 | |
| 0021/211-6 | 0 | 0 | | **3** | 0 |
| BK1/102 | 2 | 0 | | 0 | |
| UWA 1 | 2 | 0 | | 0 | 0 |
| D1 | 3 | **4** | | 0 | **26** |
| D2 | 3 | 0 | | **1** | 0 |
| D3 | 3 | **6** | | **1** | 0 |
| D4 | 3 | 0 | | 0 | 0 |
| ELF P1 | 2 | 0 | | **1** | 0 |
| Zaystr. | 3 | 0 | 0 | 0 | |
| Gartenstr. | 3 | 0 | | **1** | |
| Ottersdorfer-Str | 3 | 0 | | 0 | |
| Wussler | 3 | 0 | | 0 | |
| Rauentaler/Lochfeld Str. | 3 | 0 | | 0 | |
| 152/211-3 | 0 | | | | 0 |
| Wastewater Danziger Str | - | | | | **2.E+06** |

(i) Faecal streptococci (FS) are Gram-positive, catalase-negative cocci from selective media (e.g. azide dextrose broth or m Enterococcus agar) that grow on bile aesculin agar and at 45°C, belonging to the genera Enterococcus and Streptococcus possessing the Lancefield group D antigen (WHO, 2001). Fecal streptococci generally occur in the digestive systems of humans and other warm-blooded animals (EQC, 2005). In the past, fecal streptococci were monitored together with fecal coliforms and a ratio of fecal coliforms to streptococci was calculated. This ratio was used to determine whether the contamination was of human or nonhuman origin. However, this is no longer recommended as a reliable test.

(ii) Enterococci are a subgroup within the fecal streptococcus group. Enterococci are distinguished by their ability to survive in salt water, and in this respect they more closely mimic many pathogens than do the other indicators. Enterococci are typically more human-specific than the larger fecal streptococcus group. The US-EPA recommends enterococci as the best indicator of health risk in salt water used for recreation and as a useful indicator in fresh water as well (USEPA, 1997). Enterococci are well adapted for survival and persistence in a variety of adverse environments, including sites of infection and inanimate hospital surfaces. Enterococci have been estimated to account for 110,000 urinary tract infections (UTI) annually in the United States (Huycke *et al.*, 1998). The rapid emergence of antimicrobial resistance among enterococci undoubtedly also contributes to their emergence as prominent nosocomial pathogens, making them among the most difficult to treat (Hancock & Gilmore, 2000)

Within the 30 samples, seven positive detects of enterococci were found. The findings were restricted to urban background and sewer focus wells with one exception. In February 2005 also the reference well 0021/211-6, which is located on agricultural land upstream of Rastatt showed a positive detect. The enterococci counts generally remain on a rather low scale and corresponded to the findings of the *Escherichia coli* screening. In internal analyses carried out by the Institute for Engineering Biology and Biotechnology of Waste Water on samples from the same wells, enterococci were exclusively found at the sewage influenced wells, if false-positive strains on Bile-Esculine Agar were excluded, and not in the reference well (Gallert *et al.*, 2005)

The screening for faecal streptococci resulted in only one positive detect, which was 26 fecal streptococci in the focus observation well D1.

The wastewater sample taken from a combined sewer at the monitoring site Danziger Strasse contained $1.7\times10^6$ faecal streptococci, $1\times10^6$ *Escherichia coli* and $1\times10^5$ total coliforms. This represents a very sharp contrast to the groundwater samples. The vast majority of the faecal coliforms are usually eliminated already within the first centimenters of the soil passage. Hua *et al.* (2003) performed laboratory sewage percolation tests using sand columns and found that 99.9 % of the total coliforms and 98.6 % of the feacal coliforms were eliminated within the first 25 cm of the column. Enterococcus, once excreted, can be detected up to 140 days in the environment, although these survival times are highly variable.

Tab. 3-14: Results of the screenings for sulphite reducing clostridia (SRC), Pseudomonas aeruginosa, coliphages and total number of colony forming units cell number (GKZ). Positive detects displayed bold.

| | Lab A | Lab C | Lab A | Lab D | Lab B |
|---|---|---|---|---|---|
| | 27/10/04 | 09/05/05 | 27/10/04 | 27/10/04 | 03/02/05 |
| | SRC | | P.aeruginosa | Coli-phage | CFU |
| Sampling Point | 1/100ml | | 1/100ml | 1/1000 ml | 1/100 ml |
| 0112/211-2 | 0 | | 0 | 0 | 8.10E+05 |
| 0021/211-6 | **4** | 0 | 0 | 0 | 8.30E+07 |
| BK1/102 | 0 | | 0 | 0 | 3.00E+08 |
| UWA 1 | 0 | **1** | 0 | 0 | 2.30E+08 |
| D1 | 0 | 0 | 0 | 0 | 4.20E+08 |
| D2 | 0 | 0 | 0 | 0 | 3.80E+07 |
| D3 | 0 | **1** | 0 | 0 | 1.50E+08 |
| D4 | 0 | 0 | 0 | 0 | 1.50E+08 |
| ELF P1 | **13** | **47** | 0 | 0 | 8.60E+05 |
| Zaystr. | 0 | | 0 | 0 | 8.70E+06 |
| Gartenstr. | 0 | | 0 | 0 | 1.80E+08 |
| Ottersdorfer-Str | 0 | | 0 | 0 | 1.00E+06 |
| Wussler | 0 | | 0 | 0 | 5.60E+07 |
| Rauentaler/Lochf.Str. | 0 | | 0 | 0 | 3.40E+05 |
| 152/211-3 | | 0 | | | |
| Wastewater Danz. Str | | **1.E+05** | | | |

#### 3.4.7.4 Sulphite reducing clostridia / Clostridium perfringens

Clostridum perfringens was detected in two wells with four counts in well 0021/211-6 and with 13 counts in well Elf P1. While well Elf P1 is situated in the midst of the city and has a well construction susceptible to contamination, well 0021/211-6 is located on farmland outside the city area. None of the focus observation wells showed a positive detect of Clostridium perfringens, even when they were loaded with significant amounts of *Escherichia coli*. A subsequent sampling on the wider group of sulphite reducing Clostridia (SRC) confirmed the findings in well Elf P1 and detected low counts of SRC in two of the focus observation wells Danziger Strasse.

#### 3.4.7.5 Antibiotic resistances

Using samples from the sewer focus observation wells constructed for this thesis, analysises for antibiotic resistances were carried out by the Institute for Engineering Biology and Biotechnology of Waste Water. It was found that the antibiotic resistances of pseudomonads, faecal coliforms and enterococci from groundwater samples varied to a high extent. Many pseudomonads from groundwater samples were resistant to more antibiotics than those from the sewage. However, the pseudomonads from non-polluted groundwater were the most resistant isolates of all (Gallert *et al.*, 2005). While it appears premature to postulate a correlation between sewage exfiltration and antibiotic resistances in the groundwater, the fact that soil and groundwater beneath leaky sewers may foster the acquisition of resistances constitutes a research need for the future.

#### 3.4.7.6 Conclusions from the microbiological sampling

Summarizing it can be stated that:

- The results are extremely sensitive to the sampling technique.
- Indicators of faecal contamination have been found in several groundwater samples. The groundwater from 6 out of 12 wells is not suitable for human consumption according to german and international drinking water guidelines.
- Viruses and other pathogenic organisms were not included in the sampling programme but are expected to be present as some members of these groups are more persistent than the measured indicator organisms.
- The reproducibility of the results is limited. Wells with strong positive detects during one sampling campaign might be clean in the next one.
- No correlation between leak characteristics, distance to the leak and microbiological indicators was found within the limited number of samples.
- No correlation between other wastewater indicators (e.g. ammonium, boron, pharmaceuticals) and microbiological indicators was found within the limited number of samples.

- Wastewater in Rastatt contains high numbers of all the organisms screened for. However, an effective removal is taking place within the sediments surrounding the sewer.
- The acquisition of antibiotic resistancies in soil and groundwater beneath sewer leaks should receive further attention.

## 3.5 Focus groundwater monitoring Danziger Strasse

### 3.5.1 Introduction

Very few examples of groundwater observation wells under an unambiguous influence of a leaky sewer are documented in the literature. More frequent are examples, where specifically drilled observation wells did not show any signs of altered groundwater quality, such as experienced by the Umweltbundesamt (Hagendorf, 2004) or in Rastatt at the sewer test site Kastanienweg (Eiswirth *et al.*, 2002a). However, the drilling of a series of sewer focus wells with spot selection based on CCTV sewer condition data, produced a number of wells with obvious sewage influence. The first groundwater observation well (D3) at the spot "Danziger Strasse" was especially suspicious due to the detected short term variations in physico-chemical parameters corresponding to the flow regime in the sewer (Wolf *et al.*, 2004). As the well "D3" was only measuring groundwater quality downstream of the sewer, it was decided to built three new wells in the immediate vicinity of the sewer (two upstream, one additional downstream). The current setting is depicted in Fig. 3-18. The site Danziger Strasse was originally planned to be equipped like the test site Kehler Strasse. However, this plan had to be dropped because of high groundwater levels (rising up to the sewer bottom level and possibly flooding the instrumentation) and high construction costs due to the deep lying sewer (5.5 m b.g.l.).

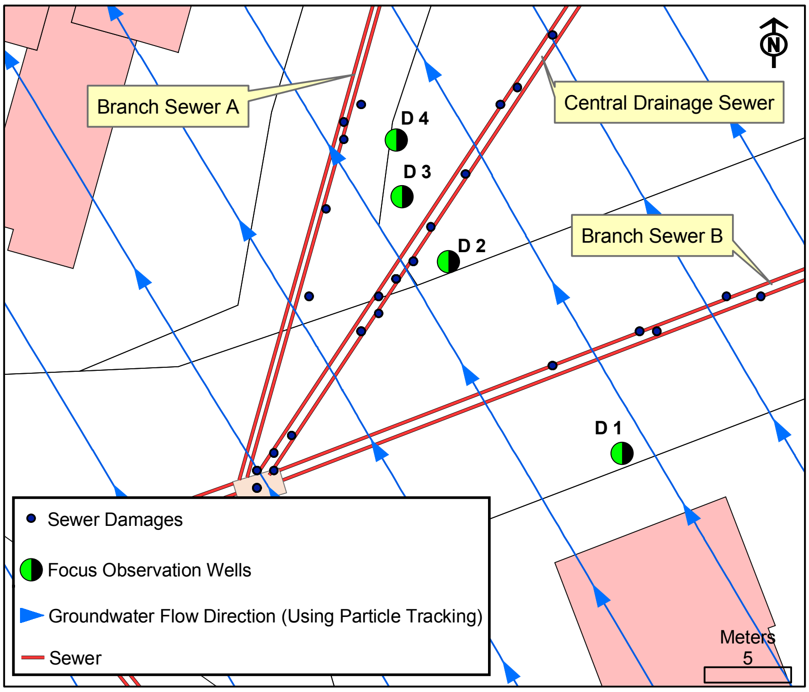



Fig. 3-18: Position of groundwater observation wells D1-D4 in relation to sewer position and groundwater flow direction. Groundwater flow direction was calculated with the numerical groundwater model using particle tracking.

The objectives of the groundwater monitoring site Danziger Strasse are as follows:

- Observe the impact of a heavily damaged sewer on groundwater quality in its immediate surrounding under natural and undisturbed conditions.
- Allow comparison between upstream and downstream calculations.
- Provide a validation data set for numerical models on sewer leakage and for approximations on exfiltrating sewage volumes.
- Quantify the impact of storm events and high hydraulic loads on the exfiltration process and on groundwater quality.

### 3.5.2 Condition of the sewers at the test site Danziger Strasse

Three sewers are potential sources of wastewater in the immediate area. The central sewer which also receives the highest hydraulic load is sewer No 42149182 with a diameter of 600 mm. The building material is concrete and the sewer shows a high number of cracks and root intrusions. (Fig. 3-19) The joints are in poor condition, mostly as a result of the installation process and the outdated construction method. Two CCTV inspections were conducted on behalf of AGK in the years 2003 and 2004 and showed several large root intrusions. The root intrusions were cut using a rotating chain with attached razor blades. This intervention was necessary to allow the double-packer system to be inserted.

The sewer system at the site Danziger Strasse is also a known bottleneck of Rastatts drainage system. According to the hydrodynamic network calculations (Arcadis, 2003) the water pressure rises more than 2 m above the top of the sewer pipe. In a topographic depression 300 m downstream this leads to sewer overflow into the public garden area which also includes a playground. The problem has been stemmed with the use of water tight manhole covers in this area.

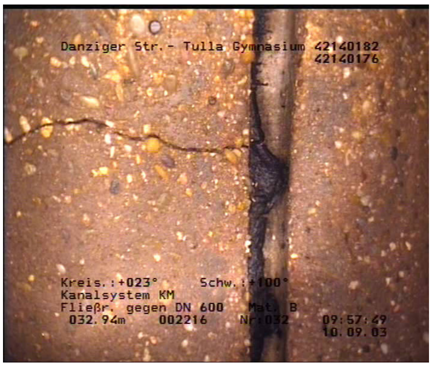




Fig. 3-19: Defects in the concrete sewer 42140182 as noticed during the CCTV inspection. Left: Improperly built joint with ageing and uneven distributed sealing tar. Right: Longitudinal crack.

### 3.5.3 Geological / Hydrogeological Setting

The geological setup as encountered during the drilling of the observation wells in the test area Danziger Strasse is depicted in Fig. 3-20. The sediments are of Pleistocene age and the result of a fluviatile depositional period. A common element in all four drillings is a layer of gravel and sand below ca. 112 m a.s.l. which denominates the aquifer in this setting. Above, a fine grained layer of silt and fine sand is encountered in the drillings D1, D2 and D4. The section between 2.3 m b.g.l. and 5 m b.g.l. in drilling D3 consists of fine to medium sand mixed with gravel and is believed to be a refill of the sewer construction process. The most common construction practice used to be the refilling of the trench with the material previously dug out which results here in a mixture of sand, silt and gravel. Not observed in a drilling but inevitably existent is the anthropogenic filling around sewer B.

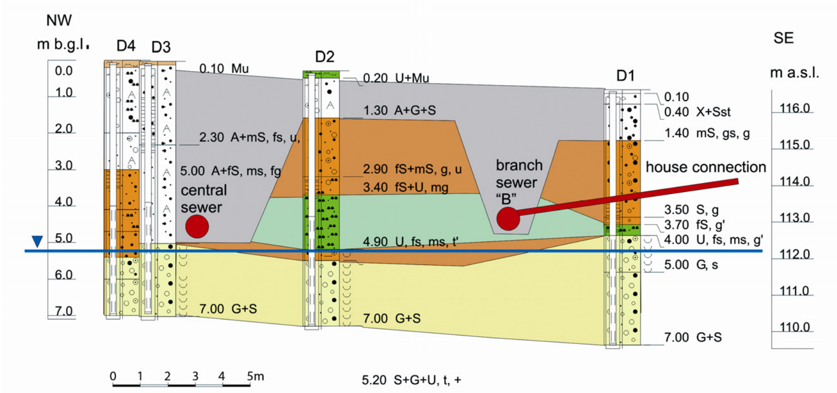



Fig. 3-20: Geological cross section at the test site Danziger Strasse.

From a hydrogeological viewpoint it has to be noticed that the central drainage sewer close to D3 is characterised by a very short distance to the water table (sometimes the water level even rises above the sewer bottom) and by a very good contact to the coarse sand and gravel deposits which form the uppermost aquifer section. Whether the construction trench around the branch sewer B is penetrating the silt layer observed in drillings D1 and D2 can not be judged from the available information.

The general groundwater flow direction was determined from groundwater level isoline maps as well as with the use of numerical groundwater models

(Fig. 3-18) The four wells of the test site are situated along the same flow path. However, the local flow field at the scale of just a few meters might also deviate from this general flow field. Deviations of at least 5° are expected and could be responsible for a lack of coherent chemical results in the four observation wells.

### 3.5.4 Online Measurements

Multiparameter probes measuring electrical conductivity, temperature, pH and water level have been installed in various observation wells since 2002. The observations include the well D3 in the Danziger Strasse which exhibits daily variations in electrical conductivity. Following the construction of the focus observation wells D1, D2 and D4, all available multiparameter probes from the city wide monitoring were shifted to the test site Danziger Strasse. The longest time series is available for well D3 (June 2003-March 2005, with some interruptions). From October 2004 onwards an online probe was also continuously installed in the central sewer Danziger Strasse. Various technical problems have caused data gaps and some of the measurements had to be discarded due to malfunctioning sensors which created erroneous results. Following October 2004, the probes were operating rather stable (Fig. 3-21). During this period the electrical conductivities in the two observation wells downstream of the defect sewer Danziger Strasse (D3 and D4) were subject to strong variations between 800 µS/cm and 1250 µS/cm. On the contrary, the upstream wells D1 and D2 and Elf P1 show a more balanced time series of the electrical conductivity. This demonstrates that the observation well D2 is not affected by the leaking sewer Danziger Strasse. The observation well D1, ca. 12 meter upstream of D3 and D4, shows only very minor oscillations in electrical conductivity. The small oscillations in D1 are mostly synchronous with the large changes in D3 and D4, indicating that a small influence of sewer water levels is present. Well D2 might not show this oscillations because (a) it is hydraulically separated or (b) a leaking house connection is present near well D1.

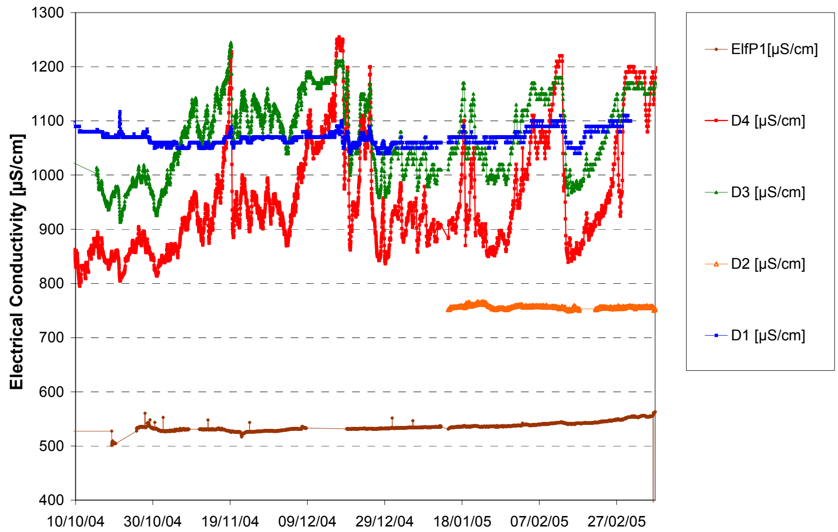



Fig. 3-21: Electrical conductivity of groundwater measured at the test site Danziger Strasse. Observation well Elf P1 is provided as a typical urban background.

Fig. 3-22 shows a period of three weeks in February 2005 in more detail and also in relation to the water levels measured inside the sewer. Following heavy rainfall on 12th February, the electrical conductivities in the observation wells D3 and D4 decline by 350 µS/cm. During the next two weeks electrical conductivities rise again as the wastewater composition returns from 100 µS/cm during rain events to electrical conductivities around 1500-2000 µS/cm. The next decline on the 28th of February is more difficult to explain as no major changes in sewer fill level were noticed in the sewer. In this case, changed wastewater composition or altered microbiological conditions in the clogging layer could be responsible for this signature. One could also think that the infiltration at the surrounding green spaces during the rain event on the 12th of February produced a wetting front that reaches the water table at the 28th of February and leads to a dilution of the groundwater. However, this effect should be noticeable also in the remaining three wells (D1, D2, ElfP1)

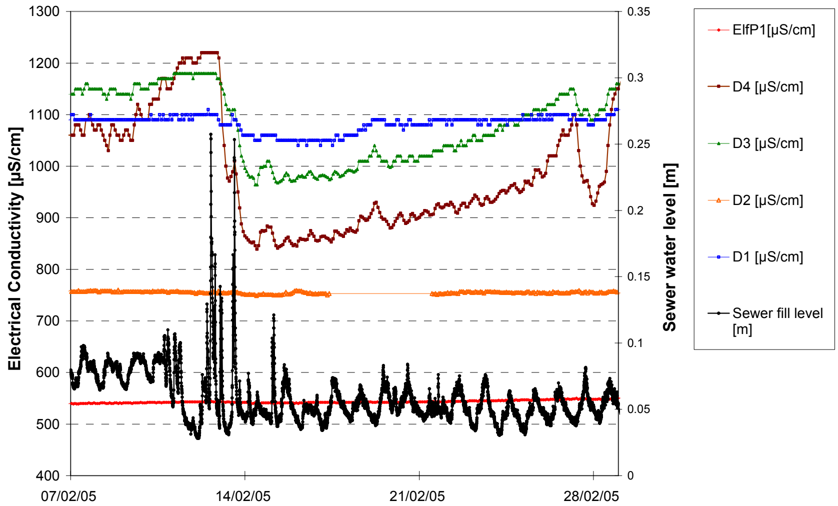



Fig. 3-22: Detailed observations for three weeks in February 2005 and influence of water levels in the sewer on groundwater quality. A sharp decrease in electrical conductivity of groundwater can be observed in wells D3 and D4 following strong rainfalls on 12.2.2005.

Fig. 3-23 provides an even finer time resolution with a display period of 6 days. Here it is possible to take a more detailed look on the response times of the aquifer. After the first major storm event (12.2.05) it takes about six hours until a reaction in groundwater chemistry is visible. Following a second storm water event at the 13.2.05, the response in the groundwater is already visible after three hours. A third small storm water event only contributes to a pre-existing downward trend and no response time can be estimated.

Fig. 3-25 displays the available time series of groundwater levels recorded with the online probes at the wells D3, UWA1 and Elf P1 during the year 2004. The three wells, which are located within a distance of 250 m show a very similar trend. Corresponding to the regional flow pattern, Elf P1 exhibits a higher piezometric level than D3 which itself is slightly above the water table recorded at UWA 1. Clearly visible is the very rapid rise in groundwater during January 2004, where almost 50 cm increase were recorded within 20 days. The strong increase is caused by extensive rainfall and snowmelting beginning at the end of 2003, combined with very low groundwater levels still present from the very dry year 2003 ("Jahrhundertsommer").

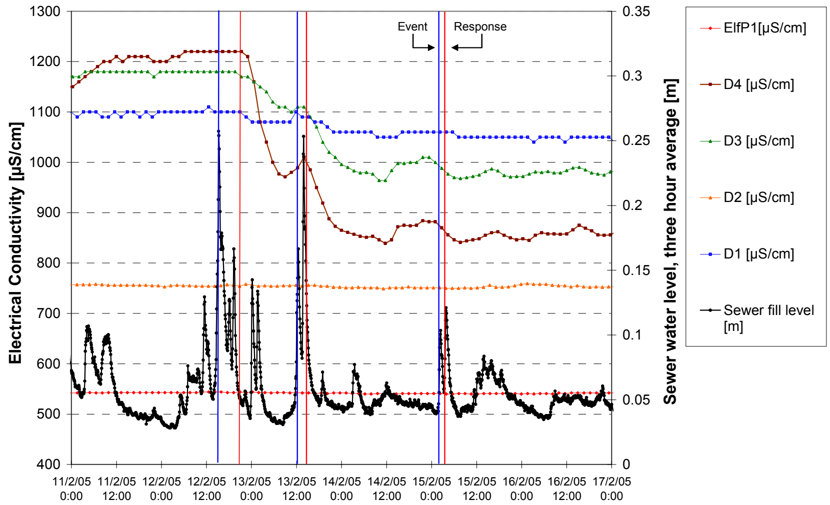



Fig. 3-23: Detailed observations for six days in February 2005 and influence of water levels in the sewer on groundwater quality. Blue vertical lines mark the arrival of the stormwater in the sewer, red vertical lines the first reaction of the groundwater.

From the online measurements at the sewer test site Danziger Strasse it can be concluded that:

- Sewers are in close contact with the aquifer
- Significant water and solute volumes reach the groundwater table
- Travel times through the unsaturated zone are short (< 1 day)
- Upstream and background observation wells are not affected by the fluctuations
- Wastewater composition is extremely variable
- Large volumes of wastewater exfiltrate during storm events but result in a drop in electrical conductivity due to the strong impact of the clean rain water on the wastewater chemistry.

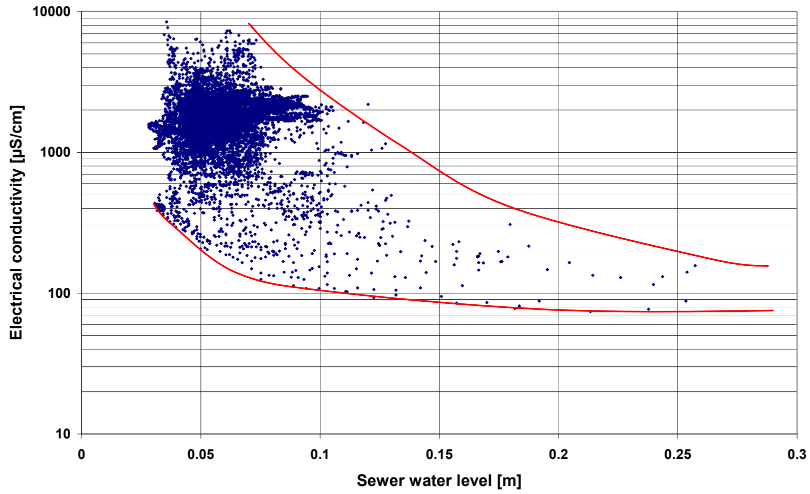



Fig. 3-24: Relation between sewer water level and electrical conductivity of the wastewater based on 19200 measurements between 26.11.04 and 9.3.2005.

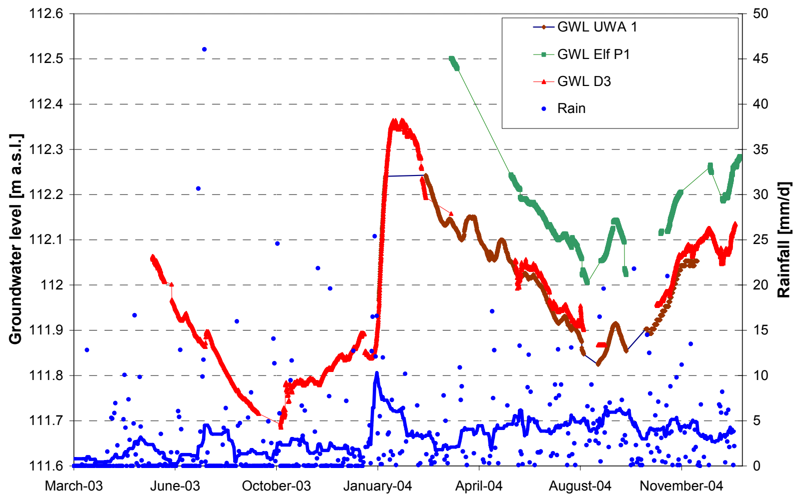



Fig. 3-25: Time series of groundwater levels at selected wells recorded with the online probes during 2004.

### 3.5.5 Water sampling and chemical analysis

Starting in January 2004, a set of 10 focus observation wells was sampled in a monthly/bi-monthly interval (see chapter 3.6), including the observation well D3 of the test site Danziger Strasse. From July 2004 onwards, also observation wells D1, D2 and D4 were sampled. Fig. 3-26 demonstrates the elevated ammonium concentrations downstream of the sewer. Fig. 3-27 documents the mean concentrations (July 2004 – March 2005) of major cations in the four observation wells arranged according to their position along the flow path.

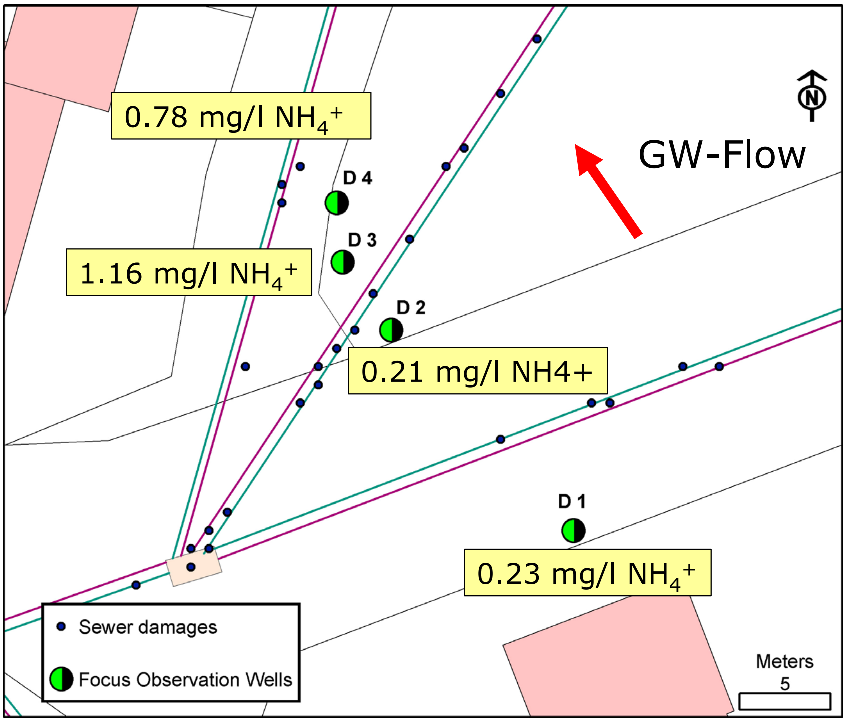
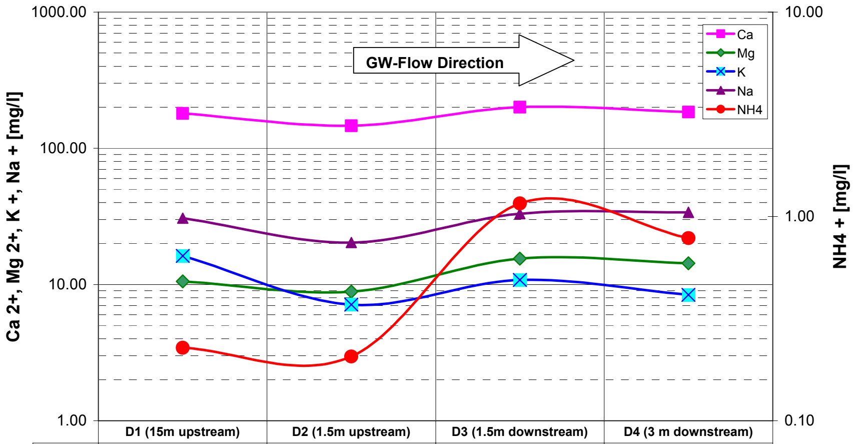



Fig. 3-26: Mean concentrations of ammonium in the groundwater observation wells D1-D4.

Obviously, ammonium concentrations are significantly higher in the downstream wells. Also for calcium, magnesium, sodium and potassium, concentrations in the downstream wells are higher than in well D2, which is directly upstream of the centrals sewer. However observation well D1, which is about

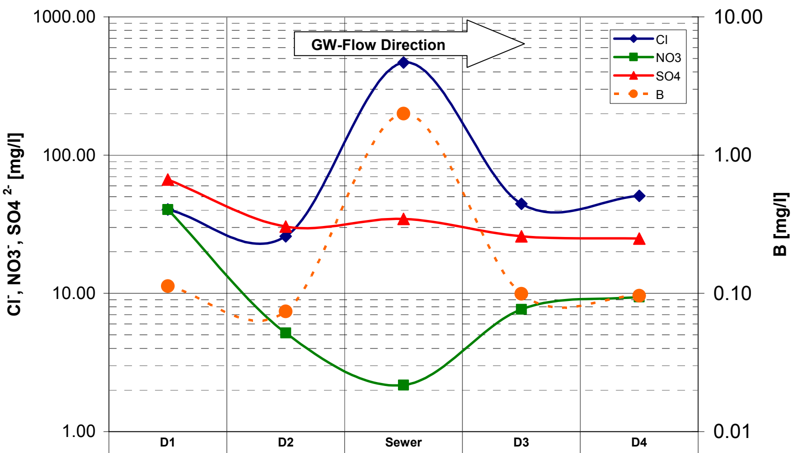
15 m upstream of the central drainage sewer Danziger Strasse, shows the same high concentrations of major cations as the wells D3 and D4. Especially potassium, which was successfully used as a sewage tracer in other places, is found at a mean concentration of 16.24 mg/l in the upstream observation well D1 whereas the downstream wells only show 10.79 mg/l (D3) and 8.4 mg/l (D4).




| | D1 (15m upstream) | D2 (1.5m upstream) | D3 (1.5m downstream) | D4 (3 m downstream) |
|---|---|---|---|---|
| Ca | 180.34 | 146.64 | 200.26 | 184.30 |
| Mg | 10.54 | 8.86 | 15.50 | 14.34 |
| K | 16.24 | 7.11 | 10.79 | 8.40 |
| Na | 30.66 | 20.30 | 33.13 | 33.89 |
| NH4 | 0.23 | 0.21 | 1.16 | 0.78 |

Fig. 3-27: Mean concentrations of major cations in the groundwater observation wells D1-D4. Based on 5 samples per well in the period July 2004 – March 2005.

Fig. 3-28 shows the concentrations of anions along the flowpath of the site Danziger Strasse. Well D1 exhibits the highest concentrations of nitrate, sulphate and borate while the downstream wells D3 and D4 show significantly higher levels of chloride. This underscores the different sources of solutes in the 4 observation wells. Well D1 is likely to be influenced by a leaky private house connection. The chloride (467 mg/l) and boron (2 mg/l) concentrations in the sewage exceed the background concentrations in wells D1 and D2 by more than one magnitude and are therefore likely to be noticed in the downstream wells. On the other hand, sulphate concentrations in the sewage water are low (34 mg/l) compared to the groundwater in well D1 (66 mg/l).




| | D1 | D2 | Sewer | D3 | D4 |
|---|---|---|---|---|---|
| Cl | 40.46 | 25.89 | 467.41 | 44.47 | 50.70 |
| NO3 | 40.42 | 5.17 | 2.17 | 7.67 | 9.42 |
| SO4 | 66.55 | 30.43 | 34.53 | 25.86 | 24.90 |
| B | 0.11 | 0.07 | 2.00 | 0.10 | 0.10 |

Fig. 3-28: Mean concentrations of anions in the groundwater observation wells D1-D4. Based on Tab. 3-15 and on wastewater samples taken on the 24.11.2004.

Tab. 3-15: Overview of parameters monitored at the wells D1-D4. n= number of samples.

| | | D1 | | D2 | | D3 | | D4 | |
|---|---|---|---|---|---|---|---|---|---|
| Parameter | Unit | Mean | n | Mean | n | Mean | n | Mean | n |
| Water level | m b.g.l. | 4.37 | 6 | 4.93 | 6 | 5.17 | 13 | 5.21 | 6 |
| E.C. after 5 min pumping | [µS/cm] | 953 | 6 | 702 | 6 | 1053 | 13 | 998 | 6 |
| E.C. after sampling | [µS/cm] | 954 | 6 | 713 | 6 | 1052 | 12 | 1004 | 6 |
| Temperature | [°C] | 15.22 | 6 | 15.37 | 6 | 14.52 | 13 | 15.22 | 6 |
| pH | [-] | 6.52 | 4 | 6.80 | 4 | 6.67 | 10 | 6.64 | 4 |
| Diss. Oxygen | [mg/l] | 0.78 | 5 | 0.39 | 5 | 0.42 | 12 | 0.26 | 5 |
| Redox [mV] | [mV] | 122.00 | 1 | 11.00 | 1 | 112.0 | 1 | 92.00 | 1 |
| $Na^+$ | [mg/l] | 30.66 | 5 | 20.30 | 5 | 37.56 | 18 | 33.89 | 5 |
| $K^+$ | [mg/l] | 16.24 | 5 | 7.11 | 5 | 10.28 | 18 | 8.40 | 5 |
| $Ca^{2+}$ | [mg/l] | 180.34 | 5 | 146.6 | 5 | 194.4 | 18 | 184.30 | 5 |
| $Mg^{2+}$ | [mg/l] | 10.54 | 5 | 8.86 | 5 | 15.27 | 18 | 14.34 | 5 |
| $NH4^+$ | [mg/l] | 0.27 | 6 | 0.23 | 6 | 1.09 | 17 | 0.85 | 6 |
| $Cl^-$ | [mg/l] | 40.46 | 3 | 25.89 | 3 | 44.47 | 5 | 50.70 | 3 |
| $NO_3^-$ | [mg/l] | 40.42 | 2 | 5.17 | 3 | 7.67 | 5 | 9.42 | 3 |
| $SO_4^{2-}$ | [mg/l] | 66.55 | 3 | 30.43 | 3 | 25.86 | 5 | 24.90 | 2 |
| $BO_3^-$ | [mg/l] | 0.11 | 1 | 0.07 | 1 | 0.10 | 8 | 0.10 | 1 |

Pharmaceutical residues and iodated x-ray contrast media analyses confirm the wastewater exfiltration Fig. 3-29. However, many substances occur at very low concentrations, indicating that the wastewater from in this mixed industrial/commercial/residential catchment are does not contain high pharmaceutical loadings.

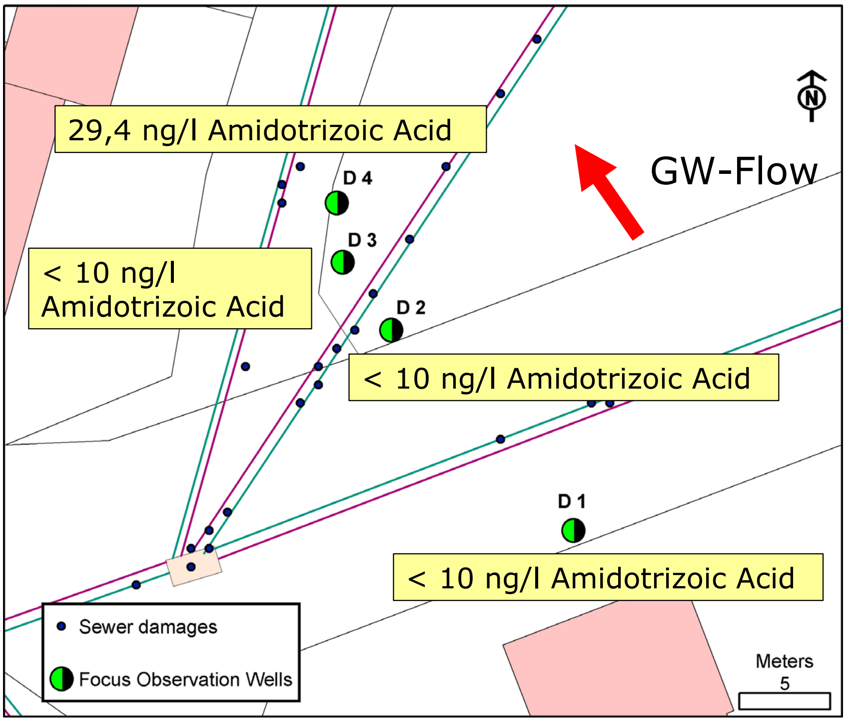



Fig. 3-29: Distribution of amidotrizoic acid at the monitoring wells D1-D4.

From the chemical sampling programme in wells D1-D4 it can be concluded that:

- Ammonium, sodium, magnesium and chloride are released in significant volumes from the sewer.
- Other factors affect the upstream observation well D1 and are responsible for high sulphate and potassium concentrations. An influence of a defect private house connection is possible. Private house connections were mentioned in other works to contribute considerably to the overall leakage (Thoma, 2005).

## 3.6 Observations at additional focus groundwater observation wells

### 3.6.1 Location

Besides the four wells at the Danziger Strasse, seven additional wells were drilled close to known sewer defects in Rastatt (Fig. 3-30). All are situated within the Rastatt city centre. The observation well Hasenwäldchen was not monitored in detail after 2004 as the corresponding sewer was filled with concrete and replaced with a parallel line.

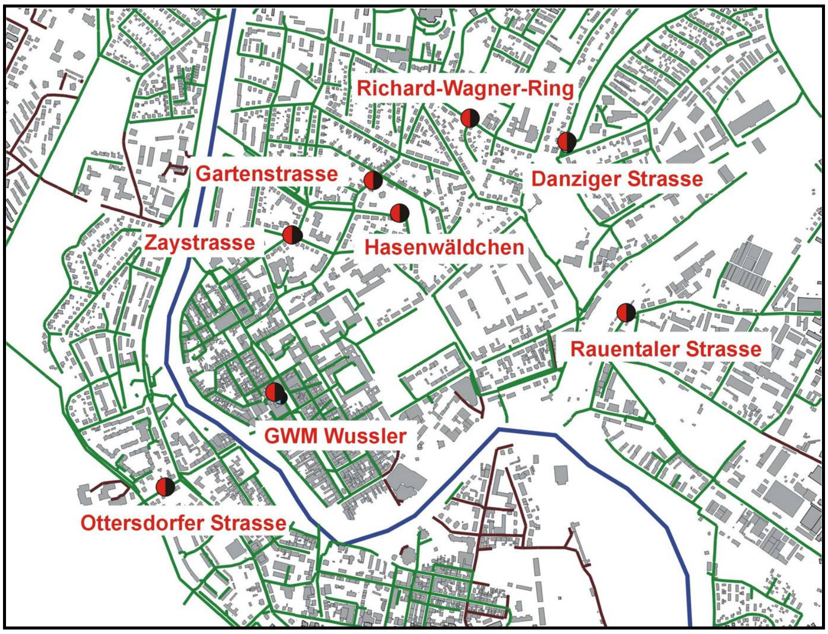



Fig. 3-30: Groundwater monitoring wells built in Feb/March 2003 (Green: combined sewer, Brown: separated sewer, Blue: River Murg). At the site "Danziger Strasse", 3 additional wells were constructed in April 2004, forming the test site Danziger Strasse (Eiswirth et al., 2004)

### 3.6.2 Seasonal variations

A comparison between the hydrochemical analyses of March 2003 and October 2003 showed a significant increase in boron concentrations in 21 observation points, which was not accompanied by an increase in major anion contents (Wolf *et al.*, 2004). This suggested that the percentual fraction of wastewater in groundwater recharge is increasing during the summer months. A decrease of chloride content in the summer is likely due to absence of road salting.

In order to analyse the seasonal variations of the hydrochemical parameters in more detail, monthly samplings were performed at 10 (later 13) selected wells throughout Rastatt. The sampled wells were screened for major anions and cations as well as boron and ammonium.

Fig. 3-31 depicts the temporal evolution of sodium concentrations in the observation wells. The focus observation well D3 shows the highest concentrations of sodium, with a strong decrease after the first sampling in January 2004. The lowest concentrations, but also the lowest variations of sodium content have been observed for the reference well 0021/211-6 on agricultural land outside the city area. It can be observed that wells which are close to known sewer defects, exhibit larger variations in sodium content than the reference wells (e.g. BK 1/102; Elf P1). However, no clear seasonality could be detected in the samples. A strong influence of road salting, as indicated by the high sodium concentrations at January 2003 was not noticed in 2004. Consequently it was checked, whether there had been any changes in the application pattern. However, this is not the case. The city of Rastatt applies around 5 g/m² halite to its road surfaces during each service drive. In total, less than 200 t/a are applied in the city area including the suburbs. This practice has remained unchanged in the last years.

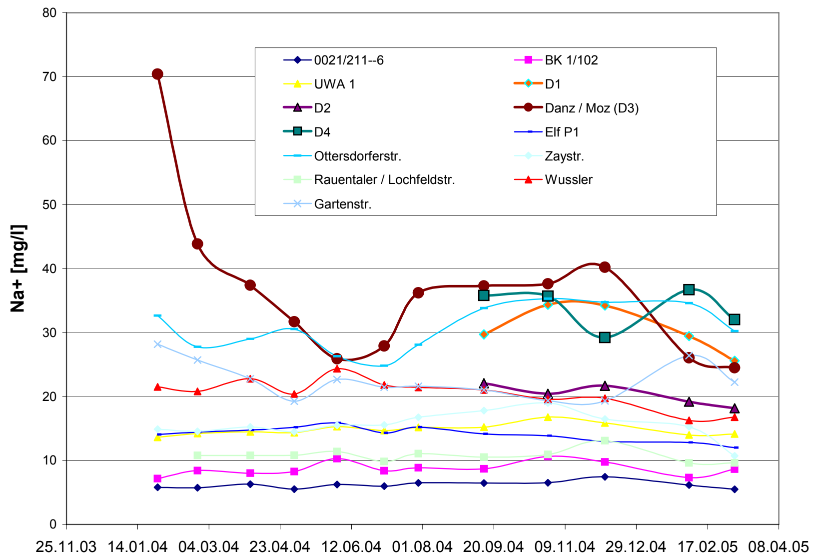



Fig. 3-31: Temporal evolution of sodium concentrations in selected wells in Rastatt.

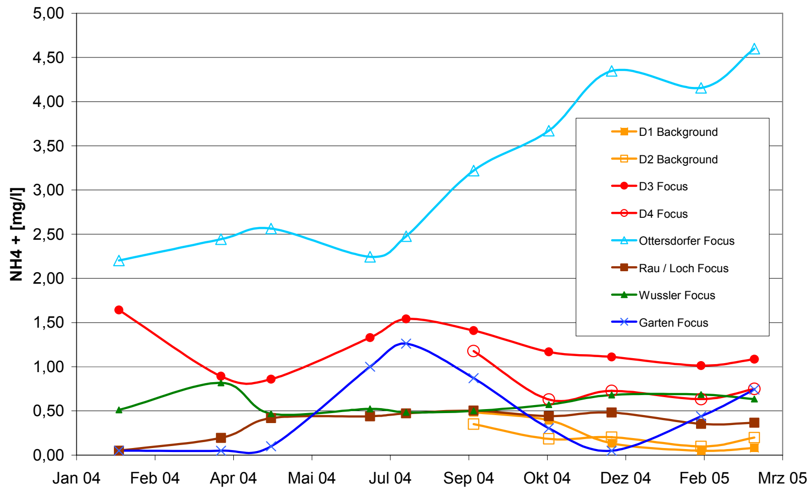



Fig. 3-32: Temporal evolution of ammonium concentrations in focus observation wells in Rastatt.

Only eight different stations are displayed in Fig. 3-32 with the temporal variations of ammonium concentrations as the other stations remained below the detection limit of 0.02 mg/l. The highest concentrations of ammonium were measured at the focus observation well Ottersdorfer Strasse which is located close to a defect branch connection. The well at the Ottersdorfer Strasse also exhibits very high concentrations of the iodated contrast media iomeprol (max. 1655 ng/l) as well as very high boron concentrations (max. 0.2 mg/l). The increasing ammonium concentrations indicate, that the environment around the source has not yet reached stable conditions or that the defects are increasing.

From the limited number of analyses, no pronounced seasonality can be seen for the observation wells except the focus observation well Gartenstrasse which seems to have low ammonium concentrations in winter. The peak in June 2004 correlates with the yearly low in groundwater levels in 2004. Further sampling activities are recommended to investigate this phenomena.

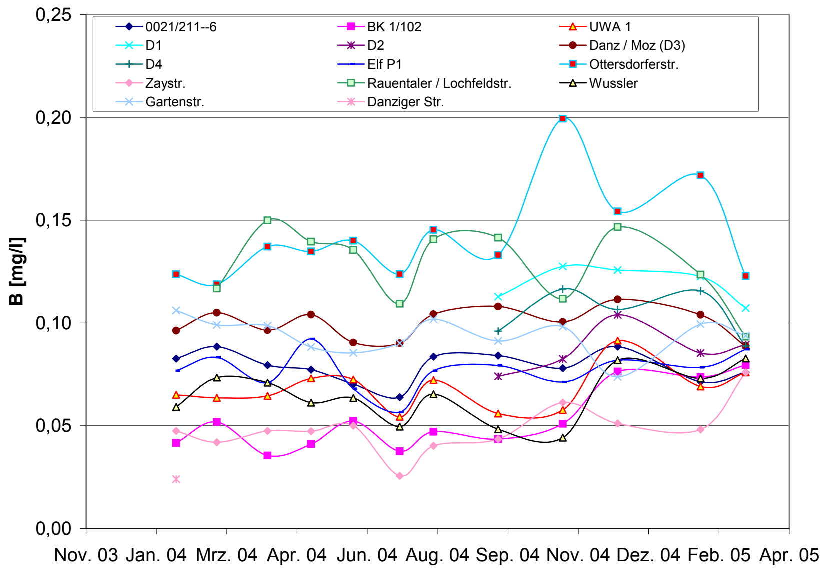



Fig. 3-33: Temporal evolution of boron concentrations in focus observation wells in Rastatt.

Pronounced variations have been observed in boron concentrations (Fig. 3-33) but again no common seasonal trends could be identified. Corresponding to the increasing ammonium concentrations, the focus observation well Ottersdorfer Strasse shows increasing boron concentrations with time. This is corroborating the assumption on a significant defect still in development. Significantly elevated concentrations have also been measured in the focus well Rauentalerstr/Lochfeldstrasse which is located close to a sewer draining the industrial area. However, this well does not show any other hints for sewage influence.

### 3.6.3 Hourly and daily variations

The observation wells Zaystrasse, Ottersdorfer Strasse and Danziger Strasse were chosen for detailed monitoring with multi-parameter probes. Groundwater level, pH value, temperature and specific electrical conductivity (SEC) are permanently monitored with multi-parameter on-line probes.

The observation wells Ottersdorfer Strasse and Danziger Strasse show elevated SEC compared to unaffected groundwater in Rastatt (300-600 µS/cm). In addition to this generally increased SEC, diurnal variations could be observed with a maximum range of 100 µS/cm in the Danziger Strasse. No such pattern could be observed for pH or temperature.

### 3.6.4 Conclusions

Conclusions from the time series monitoring at focus observation wells include:

- Seasonal patterns are not pronounced in the urban groundwater wells.
- Short term increase of sodium and chloride concentrations due to road salting were not confirmed.
- Daily variations in hydrochemical parameters in the most affected wells.
- The parameters ammonium and boron show a parallel evolution, clearly indicating sewer leakage.

## 3.7 Long term hydrochemical time series

In the archives of the environmental protection agency of Baden-Würrtemnberg (LfU), hydrochemical information dating back to 1980 could be found for several wells. The data was converted into time series and analysed for trends indicating an urban impact. Fig. 3-34 is depicting the longest available time series for the well of the local brewery, "Hofbrauhaus Hatz". The well is screened in 39 – 46 m below ground level.

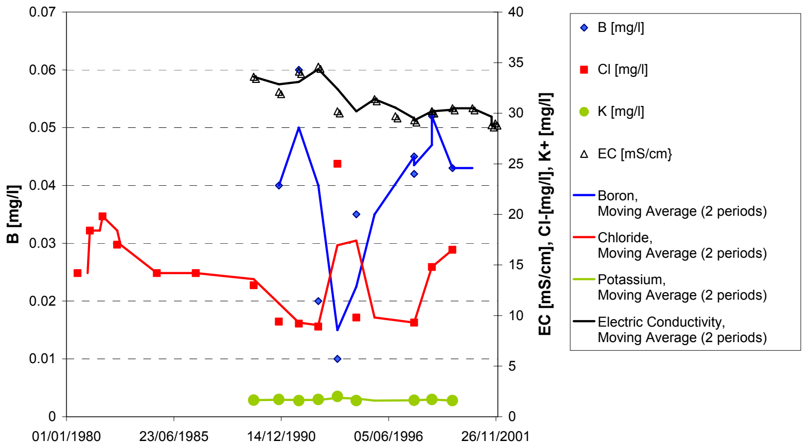



Fig. 3-34: Time series of selected hydrochemical parameters at the supply well of the local brewery, Hofbrauhaus Hatz. Lines represent moving averages over two periods

For none of the selected parameters, an obvious trend can be identified. Especially boron concentrations show a large variation. This might also be attributed to the difficulties in laboratory analysis of boron. Especially the accidental use of the standard brown glass bottles which are made of borosilicate glass might be responsible for some elevated concentrations. Furthermore, the responsible laboratory has changed several times. In the years following 1994, the measured boron concentrations seem to be more stable.

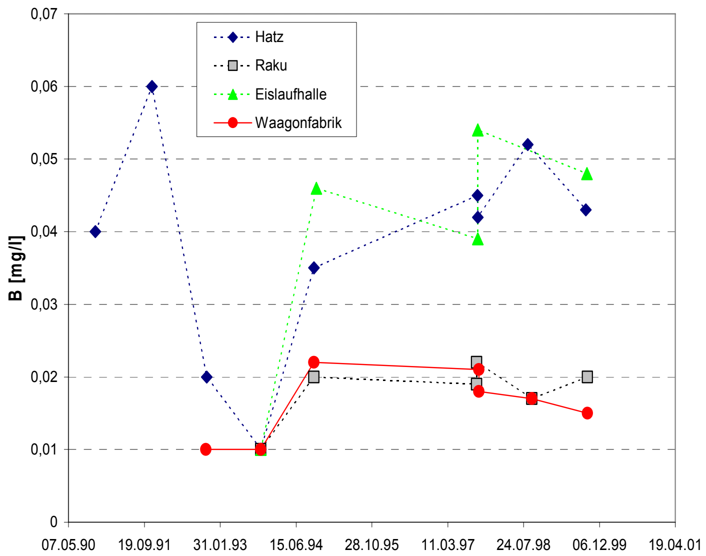



Fig. 3-35: Time series of boron concentrations in four wells in Rastatt (Data from LfU).

The time series of boron concentrations in four wells in Rastatt during a period of 10 years (see Fig. 3-35) also do not show any significant common trend. However, the wells "Hofbrauhaus Hatz" and "Eiskunstlaufhalle" clearly exhibit elevated boron concentrations around 0.04 -0.05 mg/l, which is typically only observed inside the city area.

The existing database of marker species concentrations does not reveal a distinct historical trend. However, the time series are too short to allow for the comparison between the pre-urbanisation period and the actual state.

# 4 Describing exfiltration at a single leak

## 4.1 Documented experimental work on quantities of wastewater exfiltration

Rauch & Stegner (1994) set up a test site at a wastewater treatment plant and measured exfiltration rates by using various pipes, different leak sizes, and different materials surrounding the pipes. They observed a strong decrease in exfiltration rates after the experiment started and all experiments were stopped after a maximum duration of 1 hour. They applied Darcy´s law to their set up and calculated leakage factors between 86.4 and 864 l $d^{-1}$ (i.e. 0.01 and 0.001 l $s^{-1}$). A leakage factor is also the ratio between thickness of the colmation layer and the hydraulic conductivity of the layer (Eq. 4-1).

$$L = \frac{h_{colmation}}{K} \qquad \text{Eq. 4-1}$$

With

$L$: Leakage factor [1/T]
$h_{colmation}$: Thickness of the colmation layer [L]
$K$: Hydraulic conductivity of the clogging layer [L/T]

Assuming a thickness of 0.01 m for the colmation layer this corresponds to hydraulic conductivities between 8.64 m $d^{-1}$ and 0.864 m $d^{-1}$ (i.e. 1.0 e-4 m $s^{-1}$ and 1.0 e-5 m $s^{-1}$). From depth-specific sampling of organic matter content in the bedding material they concluded that the thickness of the colmation layer is between 0.01 and 0.05 m.

Dohmann *et al.* (1999) also set up a test site at a wastewater treatment plant and used a concrete pipe with a 4 mm wide longitudinal crack in the bottom section. The surrounding material was medium to coarse sand. After 10 days of operation they measured exfiltration rates of 1.2 l $d^{-1}$ per m length of the crack. This would correspond to an extremely low hydraulic conductivity of the clogging layer of $2.68 \times 10^{-3}$ m $d^{-1}$ (i. e. $3.1 \times 10^{-8}$ m $s^{-1}$). Based on a large number of additional experiments, Dohmann (1999) calculated a best case (31 million m³ wastewater exfiltration/year) and a worst case scenario (445 million m³ wastewater exfiltration/year) for the western part of the German Federal Republic with a total length of 193.156 km combined and 87.221 km of separate wastewater sewers. The best case calculation is based on the assumption of a dry weather exfiltration rate of 4.08 l $d^{-1}$ per crack, 19.92 l $d^{-1}$ per crack during rain events and 6697.2 l $d^{-1}$ per crack for storm events (the author assumes a hydraulic pressure of 1.5 m during these events; only applied for 26 h in one year). The worst case calculation is based on the assumption of a dry weather exfiltration rate of 7.68 l $d^{-1}$ per crack, 37.68 l $d^{-1}$ per crack during rain events and 24744 l $d^{-1}$ per crack for storm events.

In a similar experimental set up, Dohmann *et al.* (1999) tested the influence of different soil types on the exfiltration behavior and found significant lower flow rates for silty soils as discussed in the following chapter.

Rott & Zacher (1999) described experiments in which they also set up a sewer test site and operated it for a total of 405 days. Both longitudinal and transversal cracks were simulated. Wastewater was applied to the system each day, but only for 10 hours. Typical exfiltration rates ranged between 0.012 l $d^{-1}$ $cm^{-2}$ and 0.68 l $d^{-1}$ $cm^{-2}$. However, the discontinuous application prevents comparison with the other experiments.

Vollertsen & Hvitved-Jacobsen (2003) described pilot scale experiments conducted with raw sewage water in Denmark. They simulated the effect of storm events, flushing of pipes, and alternating infiltration/exfiltration conditions. Furthermore, they studied different leak types and the influence of different soils. Based on their experimental data, the researchers calculated leakage factors and hydraulic conductivities for different set ups. For holes and cracks they measured typical exfiltration rates of 0.06 l $d^{-1}$ $cm^{-2}$. Assuming a thickness of the colmation layer of 0.01-0.02 m they estimated the hydraulic conductivity of the clogging layer to be 0.17 m $d^{-1}$ (i.e. $2x10^{-6}$ m $s^{-1}$). For open joints they measured significantly lower exfiltration rates of 0.02 l $d^{-1}$ $cm^{-2}$. During the simulation of storm events, 20 to up to 56 times higher exfiltration rates were measured.

As part of a multidisciplinary research project at the University of Karlsruhe, a long-term experiment on sewer leakage was conducted at the local wastewater treatment plant (Forschergruppe Kanalleckagen, 2002). A leak (0.4 cm x 14 cm) was cut into a 0.2 m diameter sewer running through a 1.5 m x 1.5 m x 3 m box filled with medium sand. Hydraulic and chemical processes were monitored using TDR probes, tensiometers and suction cups. All effluent was collected at the bottom. Typical exfiltration rates of the first 100 days were around 2 l $d^{-1}$. However, 10 times higher exfiltration rates were observed following a minor manual damage to the colmation layer. After one year of operation, exfiltration rates amounted to only 0.6 l $d^{-1}$ - 1 l $d^{-1}$.

Held *et al.* (in print) describe the results of a new sewer test site which was constructed beneath a real sewer at the city of Rastatt. Differing from the other cited experimental set ups, this test site is subject to the natural variations of water level and sewage composition induced by rain events. The daily summation of the outflow of the collecting tank underneath one leak shows large differences of base outflow (~$6\,l\,d^{-1}$ - $10\,l\,d^{-1}$) and extreme events (max. $230\,l\,d^{-1}$). The presence of a small void between the leak and the surrounding soil might be possible from the set up of the experiment. This situation also frequently occurs at non-artificial leaks where intermittent infiltration has washed away the surrounding soil. After a settling period of about six months of operation, the exfiltration rates seem to reach a quasi-stable state with approx. 2 l/d exfiltration. The very high amounts of sewage exfiltrated during the initial period of the measurements contribute strongly to the total volume which leaves the sewer through the crack. So, the average exfiltration over the entire test site running time (July 04 - June 05) amounts to 8.87 l/d. In contrast to the total amount, only 1,56 l/d leave the sewer on average during the period of February to June 05. However, even after 12 months of operation, the hourly exfiltration rate still changes by more than one order of magnitude connected to rain events.

## 4.2 Colmation

### 4.2.1 Generalities

All documented experimental set ups either at the lab scale or at the field scale show that the colmation layer (or clogging layer), which is building up in the area of the defects, has a dominating influence on the exfiltrating water volumes.

Colmation is defined as the process of decreasing pore space caused by small particles which settle down in the pore space of the soil (Busch & Luckner, 1974). Colmation phenomens may be classed in physical, chemical or biological colmation (Baveye *et al.*, 1998; Schwarz, 2004). In most cases all three processes are important (Rinck-Pfeiffer *et al.*, 2000).

The key factors influencing colmation are:

a) characteristics of the soil or filter body
b) characteristics of the loading fluid
c) characteristics of the hydraulic processes

Tab. 4-1: Parameters influencing colmation (based on Schwarz, 2003 and Schächli, 1993).

| | |
|---|---|
| **Matrix**<br>(soil or filter) | • Grain size distribution<br>• Porosity<br>• Structure of the grain surface<br>• Biology (microorganisms)<br>• Temperature, Sun |
| **Loading fluid**<br>(wastewater) | • Particle size distribution<br>• Concentration of suspended or dissolved substances<br>• Cohesion and adhesion ability<br>• Surface phenomena<br>• Biology (microorganisms)<br>• Temperature |
| **Hydraulics**<br>**(Hydrodynamics)** | • Soil water saturation<br>• Flow velocity<br>• Shear stress<br>• Air pressure (e.g. entrapped gas)<br>• Steadiness of the flow field |

Colmation is an important factor in many industrial processes and a significant amount of literature is available. However, the governing parameters of the established equations applied in technical colmation theory are difficult to estimate in natural settings due to their strong temporal and spatial variability. The most important factor with regards to sewer leakage are the different soil compositions in the immediate surrounding of the sewer. The dominating matrix grain size might vary between 20 mm (gravel) and 0.0063 mm (silt). Also the wastewater composition is subject to strong variations, especially in the case of combined sewer systems which exhibit very high water levels, flow velocities and shear stress during storm events.

#### 4.2.1.1 Laboratory experiments with soil filters

Investigations concerning the development of colmation layers on soil filter systems in water saturated conditions are documented in (Schwarz, 2004) and depicted in Fig. 4-1. The hydraulic conductivity of the clogging layer did not decrease constantly but exhibits several phases.

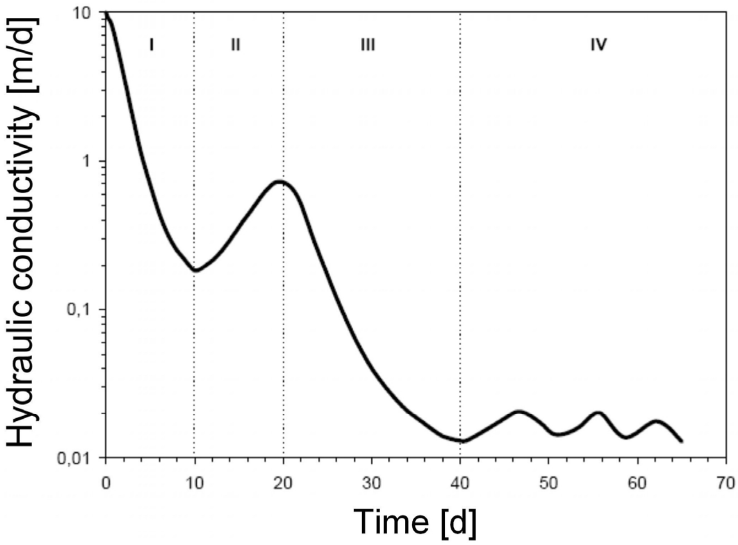



Fig. 4-1: Temporal evolution of the hydraulic conductivity of the first 4 cm soil during the colmation process. Modified after (Schwarz, 2004) using data from (Baveye *et al.*, 1998; Okubo & Matsumoto, 1979; Okubo & Matsumoto, 1983).

In most cases, the different authors distinguish four separate phases which are governed by different colmation mechanisms (Schwarz, 2004):

I. Strong biomass growth leads to a fast and usually exponential decrease of the hydraulic conductivity (Okubo & Matsumoto, 1979; Okubo & Matsumoto, 1983).

II. The biomass growth and the associated microbial transformations lead to a fast decrease in oxygen content. The oxygen content becomes the limiting factor. The lack of sufficient oxygen causes decay of biomass. In turn also the hydraulic conductivity increases as the availability of intergranular space increases.

In some experiments, air was entrapped in the soil pores. If this air is removed by the passing fluid, a temporal increase of hydraulic conductivity can result (Allison, 1947).

III. The aerobic decomposition of the dead biomass finally leads to anaerobic conditions. The ongoing growth of anaerobic organisms and the trapping of microbiologically produced gas in the soil pores leads to a pronounced decrease of hydraulic conductivity (Okubo & Matsumoto, 1983).

IV. Finally a balance between the processes described in phases I – III is achieved. The hydraulic conductivity staggers around a certain value and exhibits only minor variations. During the tests cited in Fig. 4-1 the final conductivity amounted to approx. 0.02 m/d or $2.3x10^{-7}$ m/s.

In more recent experiments, a close correlation between the decrease of hydraulic conductivity and the increase of microbial biomass in the soil filters was observed (Schwarz, 2004). The highest biomass concentration is always found in the topmost soil layer and defines the movement of water in the filter system. The biomass is the main factor controlling the hydraulic conductivity of the colmation layer.

#### 4.2.1.2 Special issues for the colmation of sewer leaks

Model concepts and quantitative descriptions of the colmation process are available for soil filters and other technical applications of filter theory. However, the application of these concepts to the colmation processes in sewer leaks is complicated by several factors which are difficult to retrieve:

- Variation of the sewage composition caused by human behaviour patterns and irregular industrial process discharges
- Strong variations of sewage composition connected to rain events
- Increased hydraulic loading and shear stress connected to rain events.

The changing sewage composition imposes different nutrient and oxygen supply to the micro-organisms in the biofilm andd leads to changes between aerobic and anaerobic conditions. This causes decomposition of biomass and an increase of hydraulic conductivity. For these reasons, measurable variations in the exfiltration rates have to be expected under every operating sewer even without the influence of rain events.

## 4.3 Mathematical description of the exfiltration process

The results of laboratory experiments on sewage exfiltration performed in recent years under various conditions and exfiltration rates have been described in the previous section. Most of the experiments revealed no clear correlation between leak size and exfiltration rates (esp. Dohmann *et al.,* 1999). This leads to a broad band of uncertainties in defining hydraulic conductivities for the clogging layer and consequently only few attempts were undertaken by the previous authors to derive such values. However, without calculating the coefficients, comparison between the different exfiltration rates measured in different experiments is not possible. As different results are available, this thesis tries to compare the individual findings and to analyse the variability in a second step. The concept of calculating hydraulic conductivities (K-values) in this case is given in Fig. 4-2.

Unlike the situation at a leak with free drainage conditions which could be calculated using Torricelli´s law, the exfiltration process is predominantly influenced by the surrounding sediments, which calls for Darcy´s law.

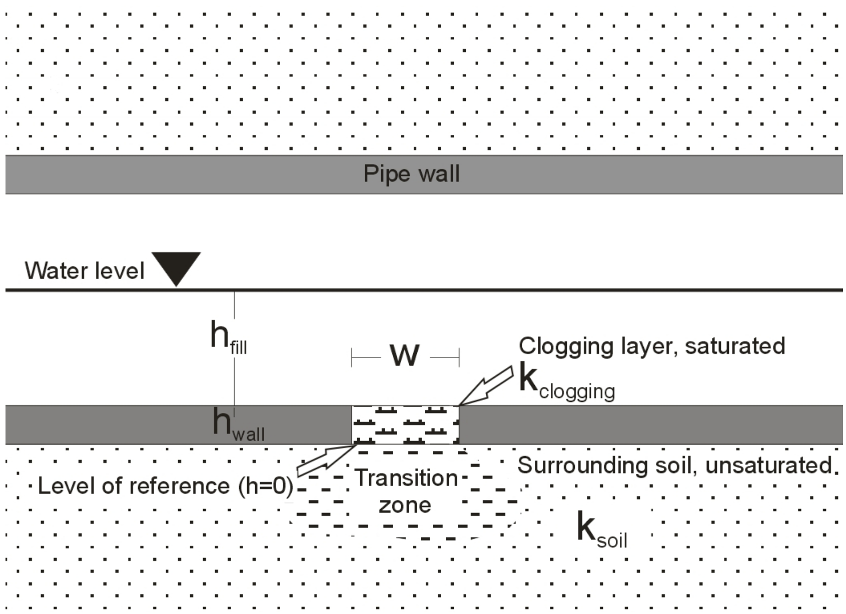



Fig. 4-2: Schematic representation of the clogged leak.

For the following calculations it is assumed that the exfiltration behaviour is mainly controlled by the filling of the crack with a mixture of sediment and bio-film (i.e. colmation layer) and that the pipes are bedded in clean medium sand. Medium sand is both the standard test environment for most of the available experiments and the most commonly used fill material for the utility trenches around the pipes.

Calculating K-values for the clogging layer following Darcy´s law (Eq. 4-2, Eq. 4-3):

$$Q = K \cdot I \cdot A \qquad \text{Eq. 4-2}$$

equivalent to

$$K = \frac{Q}{I \cdot A} \qquad \text{Eq. 4-3}$$

with:

$$I = \frac{h_{fill} + h_{wall}}{h_{wall}} \qquad \text{Eq. 4-4}$$

| | |
|---|---|
| $A$: | Size of the leak [$L^2$] |
| $Q$: | Exfiltrating flow [$L^3/T$] |
| $K$: | Hydraulic conductivity of the clogging layer [L/T] |
| $I$: | Hydraulic gradient [-] |
| $h_{fill}$: | Water level inside the sewer [L] |
| $h_{wall}$: | Pipe wall thickness [L] |

Equation Eq. 4-4 defines the gradient which is the driving force for the water movement. It is assumed that all exfiltrating water has to pass the slot (i.e. leak) which is filled with low hydraulic conductivity sediments. The level of reference for the calculation of this height difference is set at the outer border of the pipe. This includes the assumption that only a section equivalent to the pipe wall thickness is responsible for the exfiltration process. The unsaturated soil outside the pipe is assumed to ensure free drainage.

The given equation is only valid if the hydraulic conductivity of the clogging layer is indeed the control factor for wastewater exfiltration.

Under field conditions, two cases can be distinguished:

- $K_{soil} > K_{clogging\ layer}$ : Clogging layer is determining the exfiltration rate

- $K_{soil} < K_{clogging\ layer}$ : Hydraulic conductivity of the soil is the limiting factor.

The influence of the bedding material on the exfiltration rate was observed (Dohmann *et al.*, 1999) and is demonstrated in Fig. 4-3. A silt content of 18 % leads to a significantly lower flow rate compared to sand with just 7 % silt.

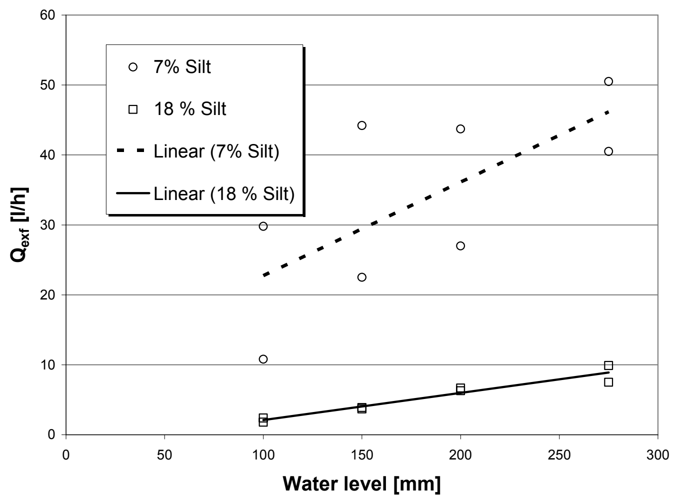



Fig. 4-3: Exfiltration from a sewer leak in relation to water level and soil type (redrawn from (Dohmann *et al.*, 1999).

## 4.4 Comparing hydraulic conductivities of the colmation layer

The K-values were calculated from the laboratory data from the various sources. They range between $6.2 \times 10^{-5}$ m s$^{-1}$ and $3.1 \times 10^{-8}$ m s$^{-1}$. This not only reflects the variability of different leak geometries but also the different set ups which lead to different ages and conditions of the clogging layer. For example, the Dohmann experiments were performed with a very young or almost non-existing colmation layer and consequently show high K-values. As the clogging layer had not yet built up to its full extent, Dohmann *et al.*

(1999) also noted a strong influence of the grain size of the surrounding soil (Figure 2). However, given the information gathered through the Karlsruhe Forschergruppe Kanalleckagen (2002) and Aalborg (Vollertsen & Hvitved-Jacobsen, 2003) experiments, it is likely that the exfiltration rates would have also continued to decrease in the (Dohmann *et al.*, 1999) experiments because the conductivity of the clogging layer would have fallen below the conductivity of the surrounding soil.

Emphasizing the importance of the age of the undisturbed clogging layers is Fig. 4-4, which compares only hydraulic conductivities of clogging layers built up on medium sand.

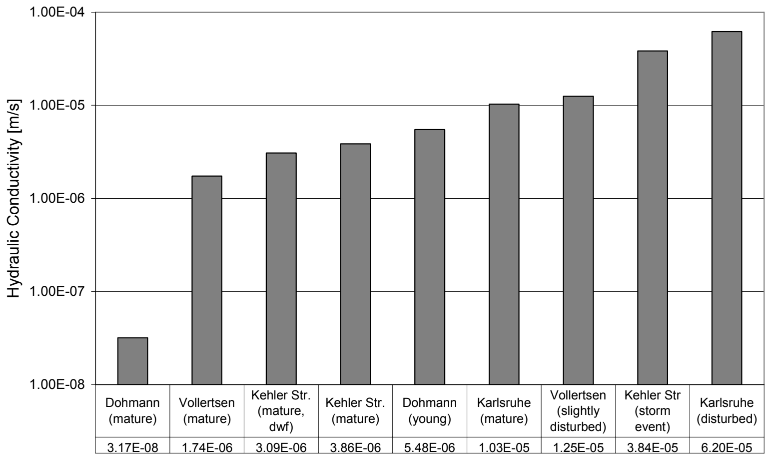



Fig. 4-4: Hydraulic conductivities in comparison with the condition of the clogging layer derived from experiments in Aalborg (Vollertsen & Hvitved-Jacobsen, 2003), Karlsruhe (Forschergruppe Kanalleckagen, 2002), and Rastatt – Kehler Strasse (Held *et al.*, 2005; Klinger *et al.*, 2005; Wolf *et al.*, 2005a).

# 5 Assessing Exfiltration at the City Scale

## 5.1 Sewer condition monitoring techniques

### 5.1.1 Generalities

As a result of legal enforcements in the frame of the „Eigenkontrollverordnung“, more and more sewers have been subject to condition monitoring in recent years. A widespread survey in the year 2004 revealed that about 89 % of the public German sewer network was already surveyed (DWA, 2005). The sewer condition assessment is performed according to advisory leaflet ATV M 143-1. The length of the public sewer network in Germany is 486 159 km, of which 48 % are combined sewers, 32 % foul water sewers and 20 % rain water sewers. In addition to this, the length of private sewers and house connections is estimated to be at least 900,000 km which are usually not included in the condition monitoring programmes.

### 5.1.2 CCTV-Inspection

Almost 1.3 million kilometres of sewers in the Federal Republic of Germany are not suited for direct human inspections. Their condition assessment relies on indirect techniques. The standard procedure for these indirect inspections is the so called “closed circuit television” (CCTV). The CCTV inspection is usually performed with encapsulated camera systems mounted on remotely controlled robot which is either pulled through or automatically moving through the sewer. Even though systems are equipped with colour television, only a visual sewer condition assessment is possible (ATV, 1999; Bölke, 1996; Stein, 1999), Modern systems include measurements of defect size, angle, deformation and temperature. All results are stored on analogue video tape or digital media. The results of the CCTV inspection are classified according to the technical guidance documents ATV A 149 & ATV M 149 in Germany.

However, the common method of CCTV inspection is associated with several problems (Mesch, 2003):

- Defective sealings of the pipe joints only in exceptional cases visible and remain usually undetected.

- Voids and open spaces in the bedding material of the pipe are not visible. The voids usually originate from the erosion of the bedding material during infiltration or are caused by wrong construction practice. The voids and cavities can lead to ground settlements which are associated with pipe bursts, street collapses or building damages.
- The result of the visual inspection is subject to the individual knowledge and judgement of the operator. The comparability of the assessment is therefore limited.

### 5.1.3 Innovative sensor systems

In an attempt to overcome the limitations of traditional CCTV techniques research was conducted on alternative monitoring systems since the 1990s (Eiswirth, 2001). In the course of a DFG research group 3d-optical triangulation with laser beams (Deutscher *et al.*, 2003), geoelectrical probes (Wolf, 2003), hydrochemical sensors (Held, 2003), microwave backscattering techniques (Munser & Hartrumpf, 2003), acoustic systems (Herbst, 2003) or radioactive sensors (Heske, 2003) were investigated. As each of the sensors described only selected aspects of the sewer condition, Frey & Kuntze (2003) developed a software which allows a neuro-fuzzy based fusion of the sensor signals. Fig. 5-1 displays the results obtained with the geoelectrical probe at the sewer test site Rastatt-Kastanienweg in comparison to the observations with the CCTV camera. While the geoeletrical probe produces a quantified signal which can easily be used in further data processing, the CCTV data delivers more detailed information (e.g. closed branch connections) In addition, due to the higher inspection cost compared to the well established CCTV systems, no widespread application of any of these sensors has started yet. Likewise the informations acquired with the new sensor systems can not be transformed into leak rates as there are no data available on the correlation between sensor signal and exfiltration rate.

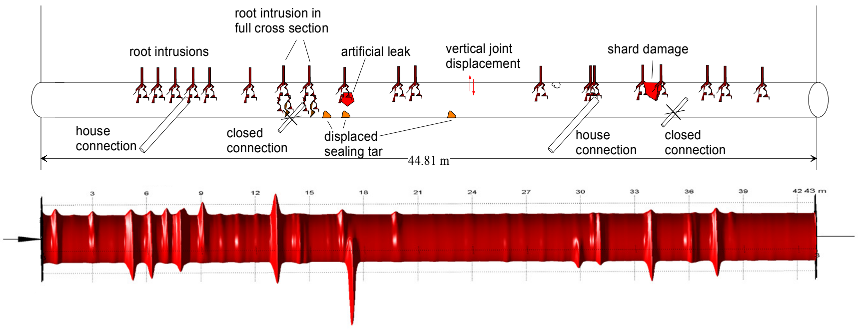



Fig. 5-1: Signals recorded with the geoelectrical probe AMS-4 at a DN300 sewer in Rastatt-Rheinau compared to results from CCTV inspections (Wolf, 2003).

### 5.1.4 Twin packer system

For the direct measurements of exfiltration rates at single leaks, a specially constructed double packer system was used in Rastatt (Eiswirth et al., 2004). The original double packer system was built at the RWTH Aachen and subsequently upgraded in Karlsruhe. The double packer system consists of two inflateble discs which effectively seal an 80 cm long pipe section (Fig. 5-2). Fresh water can be fed into the interspace via a separate hose. The water level inside the packer system is monitored by a pressure transducer which is connected via an analogue/digital interface to a laptop computer. In laboratory tests, the relation between water level and water volume in the interspace has been determined. For visual control of the packer position, a video camera system has been installed in the interspace. The double packer is moved through the sewer by two steel cables, several deflection rollers and two mechanical winches. Results of packer tests conducted between 2003 and 2004 were first documented in Eiswirth et al. (2004) with the interpretation expanded later in Wolf et al. (2005a, b).

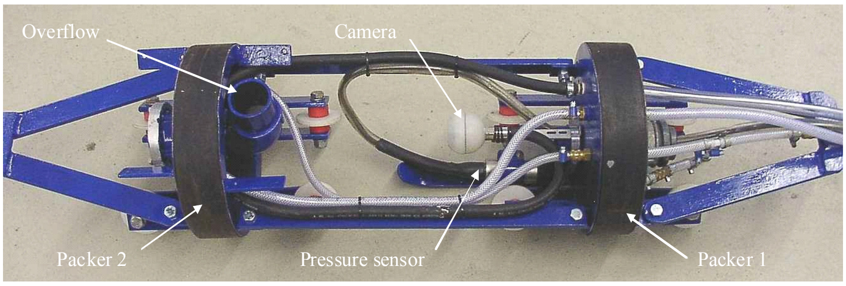



Fig. 5-2: Double-packer system (DN300) employed in Rastatt (Picture: D. DeSilva).

With the help of the double-packer system it was possible to demonstrate the effect of high pressure cleaning on the colmation layer as shown in Fig. 5-3.

From the packer tests conducted in Rastatt the following conclusions can be drawn:

- High exfiltration rates have been measured compared to the laboratory studies, international literature or the experiments at the test site Kehler Strasse.
- Not every damage noticed with CCTV also leads to measurable exfiltration.
- The exfiltration rate is strongly dependant on the sewer fill level (exponential function).
- Double packer tests with fresh water are not representative for real conditions in the sewer network. No K-value calculations have been performed.
- The packer systems are prone to malfunctioning and unhandy to use in real sewers with an uneven surface.
- Lower exfiltration rates are measured if the measurements are repeated.

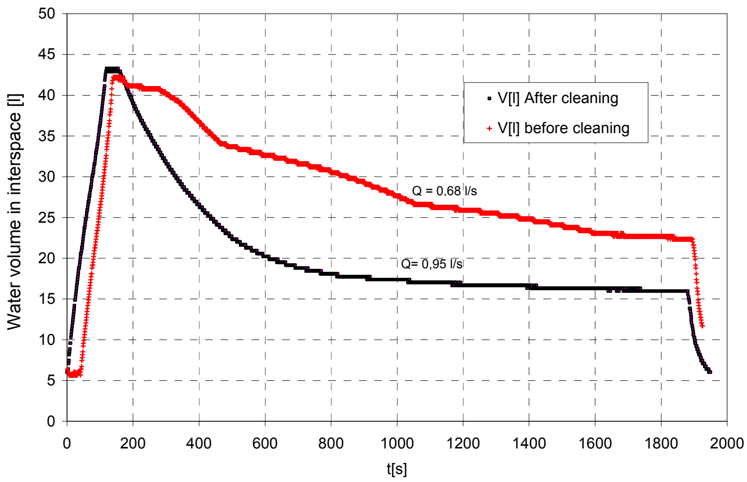



Fig. 5-3: Repeated exfiltration measurements with increase of exfiltration rate due to macceration and destruction of colmation layer (modified after Eiswirth et al., 2004).

### 5.1.5 Exfiltration test on assets

An exfiltration test on an entire asset has been performed by the AGK group and the first results documented in Eiswirth et al. (2004). The 51 m long asset (DN600 combined sewer) close to the groundwater monitoring well Danziger Strasse was filled up with water between two manholes for an exfiltration test. The decrease of water level was monitored with pressure probes and the water level and SEC in the groundwater monitoring well were observed. From 15 $m^3$ water used for the filling, approx. 600 litres were lost in half an hour (Fig. 5-4). The CCTV inspection of the sewer showed longitudinal cracks several meters long as well as leaky joints and root intrusions.

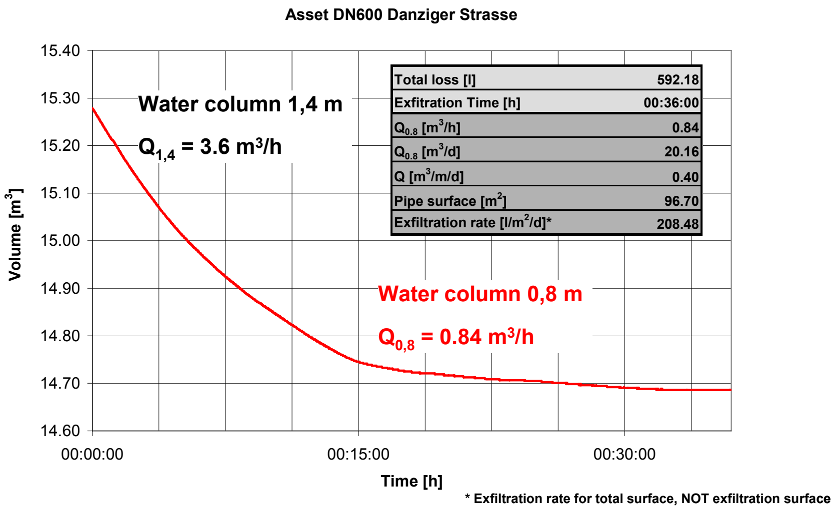



Fig. 5-4: Exfiltration behaviour in the DN600 asset Danziger Strasse during an exfiltration measurement. Exfiltration rates in the grey box are calculated with the data for an 80 cm water column (Eiswirth et al., 2004).

As the pipe could not be blocked too long, the test had to be stopped when the water level in the manholes was around 0.8 m. With the exfiltration rate at this stage (0.84 $m^3$/h), a total amount of 20 $m^3$ of wastewater per day (7358 $m^3$ per year) could be exfiltrating out of this asset into the aquifer (Eiswirth et al., 2004).

Within this thesis a further analysis of this packer test was performed considering the total area of the defects in the 51 m long sewer pipe. Based on the information stored in the sewer defect database and the calculation methods for defect areas detailed in chapter 5.2, a total defect size of 1,048 m² has been estimated. This corresponds to approx. 1 % of the total pipe wall area. However, 90 % of the defect area derives from defects at the joints. Based on a later reviewing of the video tapes and also manual inspection, it was postulated that every joint exhibits an opening of 1 mm on average.

Using the defect area detailed in Tab. 5-1 and the theoretical background detailed in chapter 4.3, the Darcy equation has been applied to the experiment. Only fill levels above 0.8 m were considered in the analysis as only these guarantee that the complete pipe diameter was filled and that uniform conditions were present. Variation of the parameter K describing the hydraulic conductivity resulted in a best fit of $K=1,14x10^{-3}$ m/s (Fig. 5-5). This is a typical value for a sand aquifer. It can therefore be concluded that no colmation effects were present under the tested conditions. Several observation support this:

- The pipe was cleaned before the installation of the packer system
- Clean water was used for the experiment
- Most damages are located above the usual water level in the sewer. No colmation is present at these defects.

Tab. 5-1: Estimating total defect area for the sewer Danziger Strasse based on CCTV inspections.

| Type | Position | Opening width | Quantity | Defect area [m²] |
|---|---|---|---|---|
| Longitudinal cracks | Upper section | 0,2 cm | 3 | 0,050 |
| | | 0,3 cm | 1 | 0,041 |
| | Left or rigth pipe wall | 0,1 cm | 2 | 0,006 |
| Root intrusions | | | 8 | 0,008 |
| Defects assumed based on own observations | | | | |
| Joint defect | | 0,1 cm | 50 | 0,942 |
| Total area of defects in pipe Danziger Strasse | | | | 1,048 |

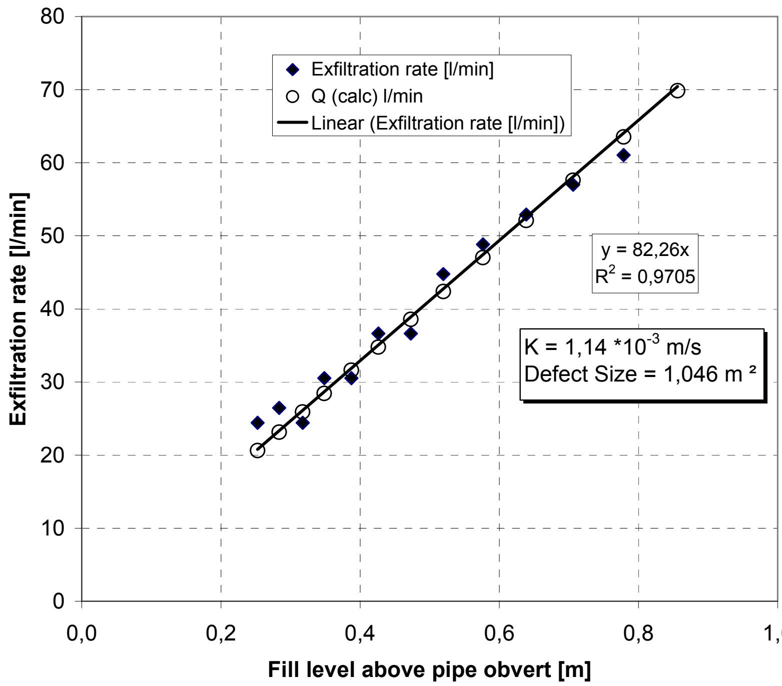



Fig. 5-5: Hydraulic conductivity calculated from the asset packer test.

## 5.2 Structural condition of the sewer network in Rastatt

### 5.2.1 Public sewer system

Obeying recent modifications in German law, the City of Rastatt has surveyed more than 90 % of its sewer network using CCTV inspections during the last ten years. Each observed damage was classified according to ATV advisory leaflet M143 (1999) and stored in a database. This sewer defect database was given spatial reference in a GIS. One has to consider that there is a considerable underestimation of defects on the sewer bottom, as during the inspection the latter was frequently obscured by the remaining water. This is of special importance as the bottom section of the sewer is usually filled with base flow sewage and therefore prone to constant exfiltration.

In order to exploit the vast amount of available CCTV inspection recording and the corresponding sewer defect database, it was attempted to estimate the resulting leak sizes. The assumptions for the calculation of leak sizes from the CCTV protocols are listed in Tab. 5-2. These assumptions are currently based on expert opinion and therefore a considerable factor of uncertainty. Empirical research on the relations between leak size and the CCTV short codes is needed. Attempts to employ neural networks for the defect classification have been documented (Moselhi & Shehab-Eldeen, 2000).

Tab. 5-2: Assumptions for calculation of leak sizes from CCTV information.

| Defect | Code | Calculation of defect size |
|---|---|---|
| Crack, transversal, total circumference | RQ | Circumference * crack width |
| Crack, transversal, one sector | RQ-L, RQ-U, RQ-R, RQ-O | 0,5* Circumference * crack width |
| Crack, longitudinal | RL | Standard length (6 m) * crack width |
| Cracks at a connection | AR | 20 cm * crack width |
| Cracks at a branch connection | SR | 20 cm * crack width |
| Joint displacement | LV, LH, LK | Circumference * displacement |
| Leaky joint | UC-- | 0,1 cm * perimeter |
| Sealing damaged | WD | 0,5 cm * perimeter |
| Root intrusion | HP | 10 cm² (wholesale) |
| Shard cracks | RX, RS | 50 cm length * crack width |
| Missing shard | BS | 10cm*10cm=100cm² (wholesale) |
| Missing shard at a branch connection | BC-- | 5cm*5cm=25cm² (wholesale) |
| Missing piece of wall | BW | 10cm*10cm=100cm² (wholesale) |
| Collapse, total | BT, BT-G | 5cm * perimeter |
| Collapse, partly | BT-L, BT-R | 5cm * perimeter *0,5 |

For the longitudinal cracks in the network, a typical length of four meters per crack was assumed. This is based on the analysis of 490 longitudinal cracks known from CCTV protocols which showed a median crack length of 4 meters. The average crack length determined from this analysis would be 10.38 m and demonstrates the involved uncertainty. The median opening width of the cracks was determined as 0.2 mm while the average opening width amounted to 0.39 mm.

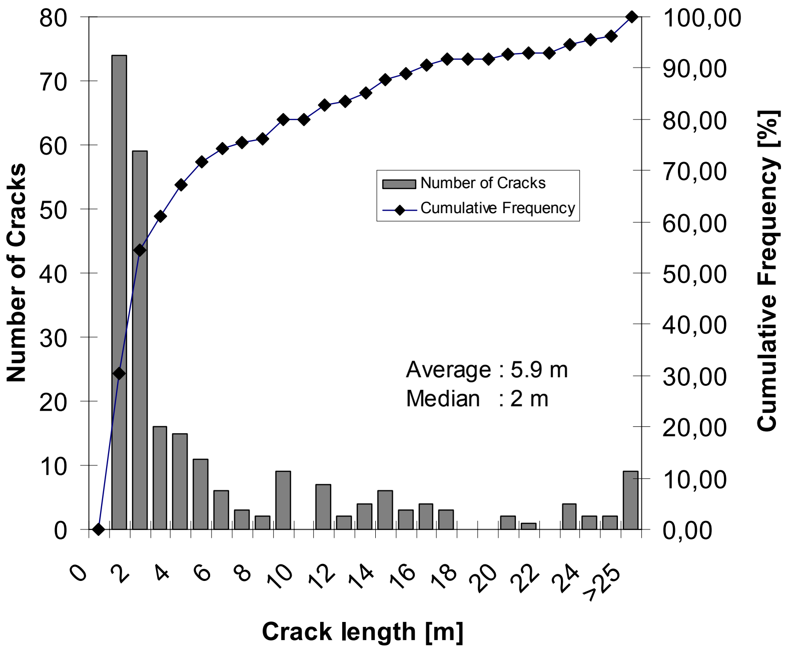



Fig. 5-6: Histogram of crack lengths determined from the CCTV database.

Tab. 5-3: Leak sizes calculated from CCTV-data in Rastatt.

| Parameter | Value | Unit |
|---|---|---|
| Area considered | 10420704 | [m²] |
| Length of inspected network | 164736 | [m] |
| Number of leaks considered | 5295 | [-] |
| Total leak size | 49.23 | [m²] |
| Average leak size | 0.0093 | [m²] |
| Leak area per m sewer | 2.99 | [cm²] |
| Leak frequency | 31.11 | [1/m] |

All three sewer types (wastewater, rainwater, combined) are considered for the following calculations. Even though rainwater sewers are assumed to convey uncontaminated water, they are included in the analysis as they are significantly contributing to the total groundwater recharge from sewers. Furthermore rainwater is affected by runoff from polluted urban surfaces (from atmospheric deposition and abrasion from tyres) as well as by accidental spillages of chemicals in the urban area. The total length of the inspected network is 165 km.

Using the assumptions stated above, a total leak size of 49.23 m² was calculated for the entire city area excluding the suburb of Rauental. A total number of 5295 leaks was considered. While more events have been noted down during the CCTV inspection, only the more serious damages listed in Tab. 5-2 have been considered. Fig. 5-7 gives an overview about the spatial distribution of the defect

Tab. 5-4: Statistics of defects considered in further calculations.

| Defect | Entire Network<br>Number of events [%] | Entire Network<br>Total defect area [%] |
|---|---|---|
| Crack, transversal, total circumference | 6.50 | 4.17 |
| Crack, transversal, one sector | 3.74 | 1.84 |
| Crack, longitudinal | 16.52 | 25.00 |
| Cracks at a connection | 0.85 | 0.02 |
| Cracks at a branch connection | 4.34 | 0.38 |
| Joint displacement | 15.70 | 47.06 |
| Leaky joint | 2.86 | 2.42 |
| Sealing damaged | 1.93 | 1.70 |
| Root intrusion | 21.90 | 2.70 |
| Shard cracks | 16.42 | 4.76 |
| Missing shard | 4.68 | 5.77 |
| Missing shard at a branch connection | 3.03 | 0.94 |
| Missing piece of wall | 1.18 | 1.45 |
| Collapse, total | 0.33 | 1.67 |
| Collapse, partly | 0.03 | 0.12 |
| Sum | 100.00 | 100.00 |

Tab. 5-4 shows that root intrusions, longitudinal cracks, shard cracks and joint displacements are the most frequent damages considered. However, in terms of defect area the joint displacements clearly dominate with 47 % of the total defect area, followed by longitudinal cracks contributing 25 % of the total defect area.

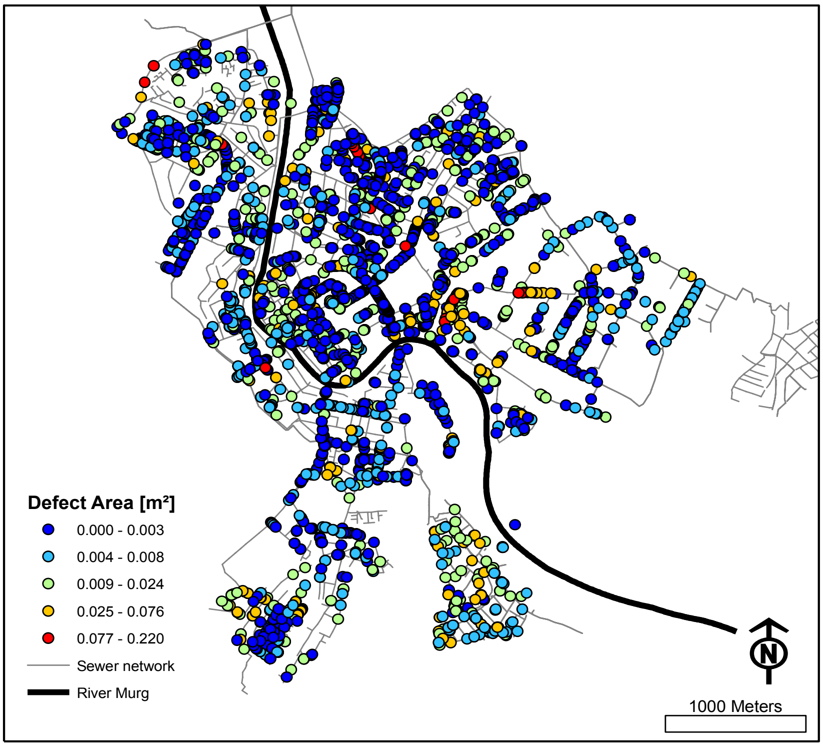



Fig. 5-7: Defect sizes estimated from the CCTV inspection data.

### 5.2.2 House connections and private sewers

No figures and databases are available in Rastatt to describe the length and structural condition of the house connections and private sewers. Investigations in other cities point to a more or less consistent picture of 90 % leaky sewers on private properties (Eisener, 2005). Special concerns were raised for private sewers connected to chemical laundry facilities (Dornbusch, 2001). The total length of the private networks is estimated as two to four times the length of the public network. Information on the condition of these sewers is usually not available. According to new regulations in Germany, all house connections with domestic sewage should be inspected before the year 2015 (DIN 1986-30). However, given the limited number of service providers and resources, it is estimated that the inspections will need at least 30 years and the corresponding rehabilitation will need 20–40 years (Thoma, 2005).

The inspection of 163 private properties in Unterfranken and Göttingen showed an average pipe density of 0.08–0.12 m per m² property area. The average length of pipes per building was 47 m. A more detailed survey of 4000 m house connection pipes on 88 properties showed average defect frequencies of 250 defects / km (Thoma, 2005).

The urban area of Rastatt as outlined in Fig. 5-7 contains 10272 buildings (according to GIS analysis). Applying the average pipe length observed in Unterfranken and Göttingen, the length of the private sewer network in Rastatt would be approx. 482 km, which is about 3 times the length of the public sewer system. Considering a similar structural condition, this would correspond to 102500 defects.

It is obvious that the CCTV-based assessment of sewer leakage for the urban area of Rastatt can never be accurate if such a large network of unknown condition is not considered in the calculations. However due to the non-availability of more detailed information, this thesis focuses on the public sewer network. Even though the length of the public network is smaller, exfiltration volumes are expected to be higher due to the continuous hydraulic load.

## 5.3 Hydrogeological boundary conditions

### 5.3.1 Groundwater levels

The key factor determining whether a leaky sewer will be subject to infiltration or exfiltration is the groundwater level. If the groundwater level rises above the sewer water level, water can only enter the leaky sewer and no exfiltration of waste water into the groundwater is possible. In order to analyse the extent of sewers beneath the groundwater table, available groundwater table information was compared to the depth of sewer bottom via a GIS analysis (Fig. 5-8)

It is evident that the largest part of the Rastatt sewer network is above the groundwater table but that a significant part is also situated within a distance of only 0.5 m to the groundwater table. As the analysis of long term time series of groundwater levels (cf. chapter 2.3.6) shows that groundwater levels in the area of Rastatt may vary up to 1.8 m, the surcharging situation has been analysed for different hydrological situations.

Based on the groundwater level measurements performed during 2001-2005 and the long term data series available from the LfU, groundwater level contour maps have been produced for 5 characteristic dates. Also the characteristic dates which were defined by the environmental agency of the state of Baden-Württemberg (LfU) for low, medium and high groundwater level situations in the Upper Rhine Graben were taken into account:

- 2.2.2002: Highest groundwater levels recorded by AGK during 2001-2005
- 4.8.2004: Lowest groundwater levels recorded by AGK during 2001-2005
- 9.9.1991: Characteristic situation for low groundwater levels as specified by LfU (1996)
- 20.10.1986: Characteristic situation for low groundwater levels as specified by LfU (1996)
- 11.4.1988: Characteristic situation for low groundwater levels as specified by LfU (1996)

As shown in Fig. 5-9, 31.27 % of the Rastatt sewer network is situated below the groundwater table during typical high groundwater levels as specified for the 11.4.1988. At typical low groundwater levels as specified for the 9.9.1991, only 5.43 % of the sewer network is situated below the groundwater table. At the beginning of February 2002, high groundwater levels led to infiltration conditions for 37.35% of the sewers.

Altogether in can be concluded, that approximately 30 % of the sewer network are subject to changing infiltration / exfiltration conditions. This is of special importance as an intermittent infiltration process removes not only the colmation layer but can also transport sediment material from the sewer surrounding into the sewer. The surrounding soil material is carried away in the sewer network and leaves behind an empty void around the leak. In a subsequent exfiltration period, this void will fill up again with sewage sludge and biofilm.

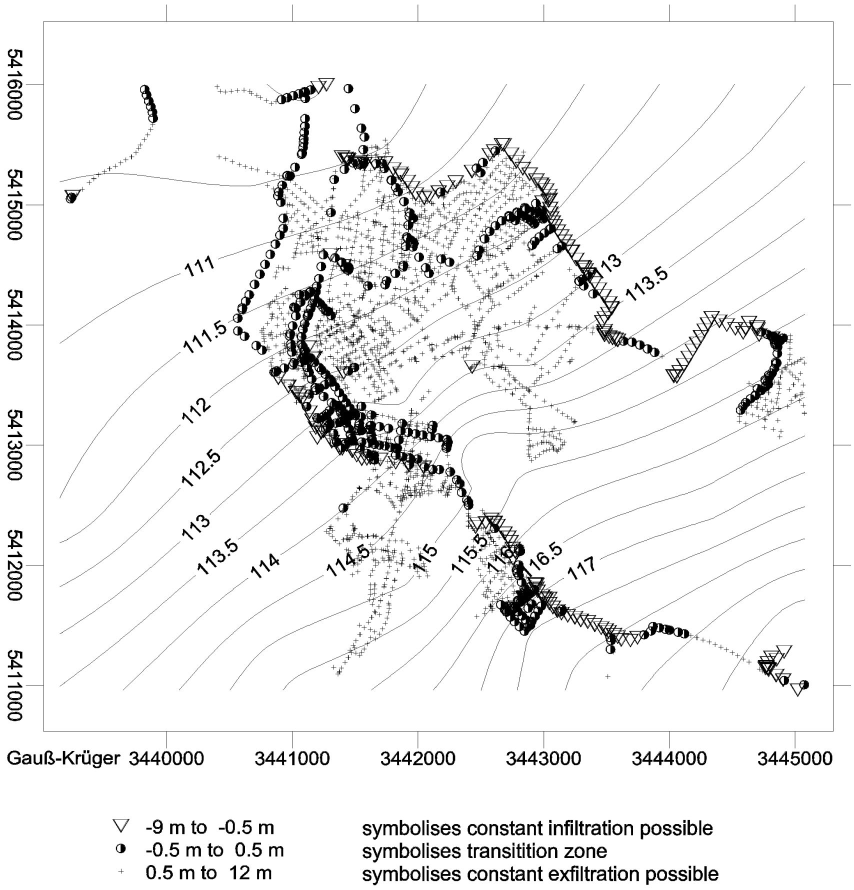



Fig. 5-8: Position of sewers in Rastatt in relation to groundwater level (m a.m.s.l. measured at 7.9.2001).

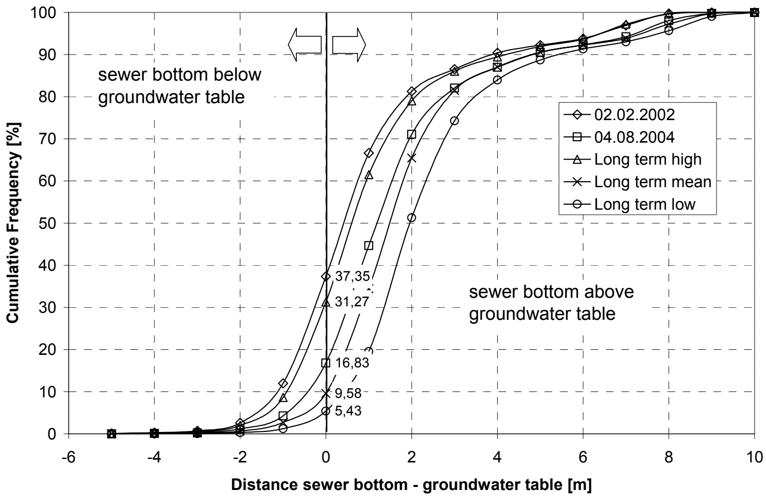



Fig. 5-9: Position of sewers in Rastatt in relation to groundwater level at different hydrological situations.

### 5.3.2 Protective function of the unsaturated zone

The distance between sewer bottom and groundwater level also describes the thickness of the unsaturated soil zone which is responsible for the main part of contaminant attenuation and degradation. Beneath sewers with only a very thin unsaturated zone, the contaminants are directly released into the groundwater. The thickness of the unsaturated zone is therefore a key factor in the risk assessment of leaky sewers.

Based on the information shown in Fig. 5-8, a risk map was produced for the Rastatt sewer network. The map differentiates between four risk classes associated with the unsaturated zone characteristics:

- Infiltration: Sewer is always beneath the water table (even during the low water table situation at the 9.11.1991). No exfiltration possible. Low risk
- High: Sewer is subject to intermittent infiltration and exfiltration. Direct contact to the aquifer + possible voids due to sediment transports. Sewer is located below the high water table (9.11.1992) and above the characteristic long term low (11.8.1988). Highest risk class.

- Medium: Constant exfiltration but less than 1 m unsaturated zone. Medium risk.
- Low: Constant exfiltration and always more than 1 m unsaturated zone. Low risk.

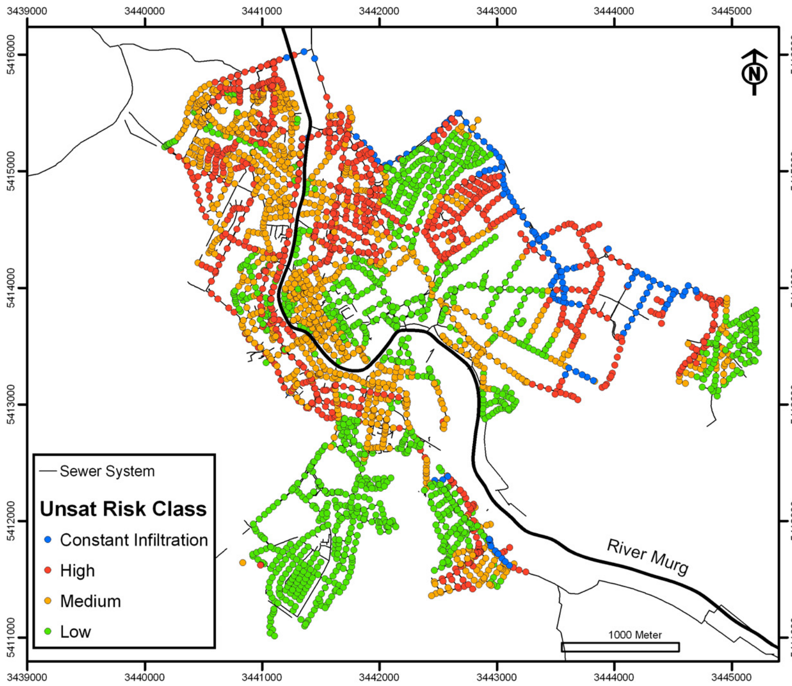



Fig. 5-10: Risk indicators based on unsaturated zone thickness.

The distribution of risk classes shown in Fig. 5-10 does not consider any information about the lithological composition of the unsaturated zone sediments. For instance, if clay is present beneath the sewer, the risk of groundwater contamination is reduced significantly. The map in Fig. 5-10 assumes that sand is underlying the complete sewer system. However the upper section of sediments present in the urban area of Rastatt is quite a heterogeneous mix between fine and coarse grained material as it is typical for a fluviatile sedimentary environment. As it is not possible to map the detailed distribution of clay and silt lenses, it is advisable to use a risk map which may overestimate risk at individual points rather than to rely on a non-existing clay barrier during the planning process. Furthermore, it must be considered that at least the bottom section of the utility trench around the sewer must be filled with sand according to German construction standards.

## 5.4 Monte Carlo based quantification of exfiltration at network scale

### 5.4.1 Principles

Uncertainty is usually associated with risk, where risk includes the possibility of an undesirable event coupled with severity. As uncertainty and risk increase, decisionmaking becomes more difficult (CB Manual). The challenge within the quantification of sewer leakage is both the significant variability but also the remaining uncertainty in the input parameters of any upscaling procedure. Traditionally, best and worse case scenarios are set up to produce a feel for the uncertainty in the predictions. A more advanced method is to use Monte Carlo approaches to demonstrate the range of possible solutions and the probability attached to a result class. First applications of Monte Carlo techniques to the quantification of sewage exfiltration based on CCTV information were described in Wolf & Hötzl (in print) for the subcatchment Rastatt-Danziger Strasse.

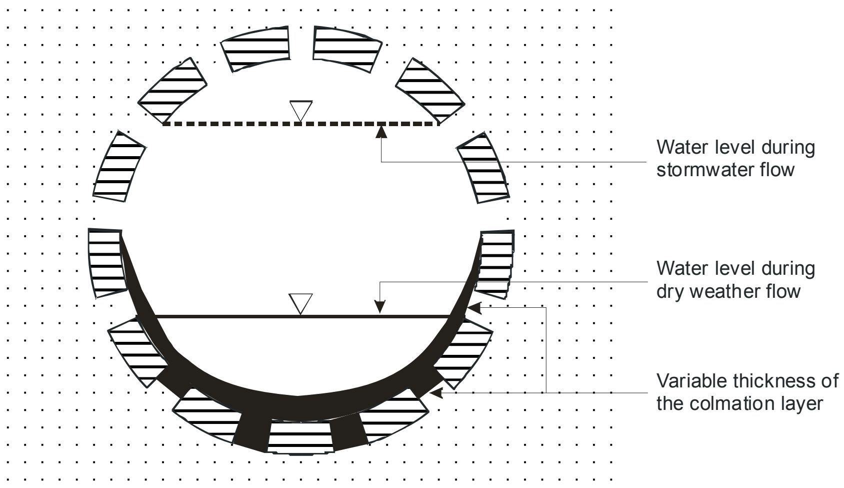



Fig. 5-11: Variable thickness of the colmation layer shown in a cross section of a sewer.

In the following chapter, the approach described in Wolf & Hötzl (in print) is applied to the entire city area of Rastatt. While the Monte-Carlo analysis for the sub-catchment Danziger Strasse was performed using an own MS-Excel based model, the analysis for the entire city area is now performed with the help of Crystal Ball$^{TM}$ Ver. 7.2.0. Crystal Ball$^{TM}$ Ver. 7.2.0 is commony applied to support economic decision making under uncertainty and allows the specification of various probability density functions for each input parameter.

The basic equation for the description of the exfiltration volumes for a leak has been derived from the model concepts detailed in chapter 4.3. As all laboratory experiments described a strong variation between high flow and low flow conditions, separate calculations are done for these two cases. Different hydraulic conductivities are assumed for the colmation layer as well as different fill levels in the sewer, leading to different hydraulic gradients.

Fig. 5-11 shows schematically the distribution of the colmation layer in the cross section of a sewer. Cracks in the sewer bottom are constantly submerged and receive a permanent input of particulate matter and nutrients. On the contrary the upper sewer sections are seldom covered by wastewater and even then receive a very low material input as mainly stormwater flows through the system.

The total annual exfiltration for a single leak may be defined as:

$$q_{Leak} = k_{LowFlow} \cdot A_{Leak} \cdot I_{LowFlow} \cdot \alpha + k_{HighFlow} \cdot A_{Leak} \cdot I_{HighFlow} \cdot (1-\alpha) \qquad \text{Eq. 5-1}$$

With:

$\alpha$ [-] : Ratio between dry weather flow and stormwater flow.
$k_{LowFlow}$ [m/s]: Hydraulic conductivity of the clogging layer during dry weather flow.
$k_{HighFlow}$ [m/s]: Hydraulic conductivity of the clogging layer during stormwater events.
$A_{Leak}$ [m²]: Total area of open sewer defect/soil interface in the catchment.
$I_{LowFlow}$ [-]: Hydraulic gradient across the colmation layer during dry weather conditions.
$I_{HighFlow}$ [-]: Hydraulic gradient across the colmation layer during dry weather conditions.

The precision of equation Eq. 5-1 can be increased if only the wet proportion of the leak area is considered for the calculation of exfiltration. In a first order approximation it is assumed that the defects noted by the CCTV inspection are more or less evenly distributed over the entire pipe section. This implies that only a small proportion of the total leak size is available for exifltration during dry weather flow conditions. To address this problem, equation Eq. 5-2 was derived using trigonometry to connect fill level and wetted pipe perimeter:

$$WP = \frac{1}{\pi} \cdot \frac{2 \cdot \arccos(1-h)}{180°} \qquad \text{Eq. 5-2}$$

With:
WP [m]: wetted perimeter.
h [m]: Sewer fill level

For a more general use, the wetted perimeter is converted to the wetted perimeter ratio (WPR) in Eq. 5-3:

$$WPR = \frac{WP}{TP} \qquad \text{Eq. 5-3}$$

With:

| | |
|---|---|
| WPR: | Wetted perimeter ratio |
| TP: | Total perimeter |

Fig. 5-12 shows the wetted perimeter ratio as a function of the sewer fill level. While the function is linear in the central part, there is an increased steepness for low fill levels. As these low fill levels are prevailing in the sewer system for most of the year, it is important to apply the transformation explained above.

For the entire city area the hydrogeological boundary condition of the percentage of sewers below groundwater table must be considered:

$$q_{Leak} = (k_{LowFlow} \cdot A_{Leak} \cdot WPR_{LowFlow} \cdot I_{LowFlow} \cdot \alpha + k_{HighFlow} \cdot A_{Leak} \cdot WPR_{HighFlow} \cdot I_{HighFlow} \cdot (1-\alpha)) \cdot AGW$$

Eq. 5-4

With:

$AGW$ [-] : Proportion of sewers above the groundwater table.

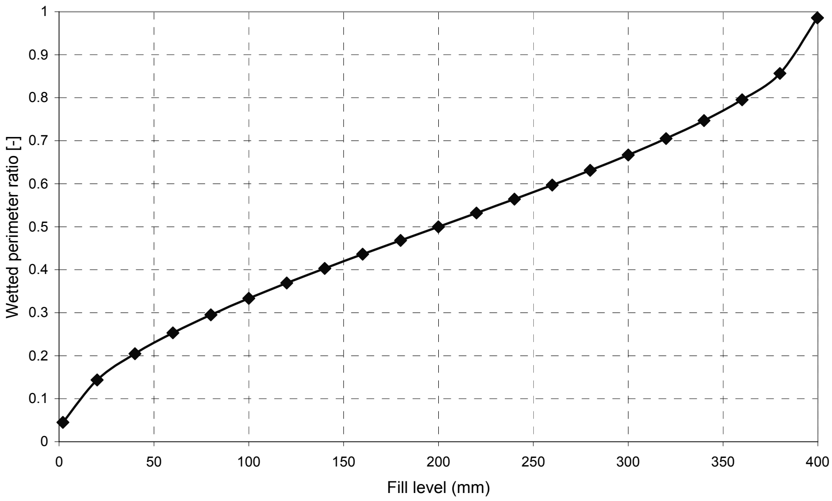



Fig. 5-12: Wetted perimeter ratio as a function of sewer fill level.

### 5.4.2 Defining uncertainty ranges for input parameters

#### 5.4.2.1 Types of probability distributions

For each uncertain variable in the simulation, the possible values are defined with a probability distribution. The type of probability distribution depends on the available knowledge and the conditions surrounding the variable. Common types of probability distributions are shown in Fig. 5-13.

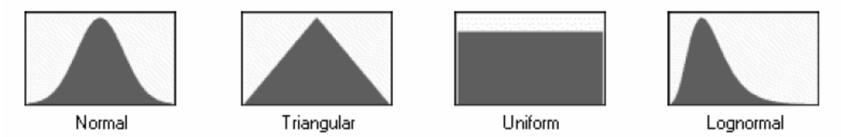



Fig. 5-13: Common types of probability distributions (Decisioneering, 2006).

Based on the different results from experiments described in the literature as well as the measurements performed directly in Rastatt, probability distributions were determined for each of the seven input parameters.

#### 5.4.2.2 Hydraulic conductivity of the colmation layer

The hydraulic conductivities of the colmation layer are subject to: a) the range of values from different laboratory experiments, and b) varying soil types in the area. For this reasons the whole range of values measured in different experiments has been applied for this calculation.

Two different hydraulic conductivities need to be described according to equation Eq. 5-4 for the conditions of dry weather flow and for storm water flow. For this the data summarized in chapter 4.4 as well as additional data from (Wolf *et al.*, 2005a) are used. The hydraulic conductivity during dry weather conditions is described with a triangular distribution with a mean of $1\times10^{-6}$ m/s, a minimum of $1 \times 10^{-7}$ m/s and a maximum of $1 \times 10^{-5}$ m/s (Fig. 5-14).

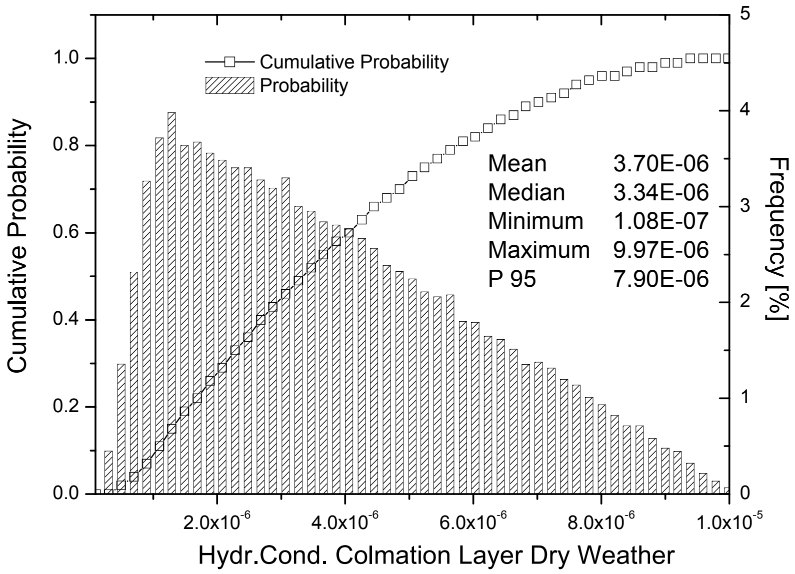



Fig. 5-14: Probability distribution assigned to the hydraulic conductivity of the colmation layer during dry weather flow.

The hydraulic conductivity of the colmation layer during storm events is considerably higher. This is caused on one hand by the partial removal of the clogging layer under high hydraulic stress. On the other hand the defects which are activated by the high fill levels during storm events exhibit a much thinner colmation layer (see also Fig. 5-11). Coming to the sewer top, the colmation layer may even be missing and the conductivity of the surrounding material (in our case sand) becomes the limit. As less data is available on the hydraulic conductivity during storm events, a triangular probability distribution is chosen. This probability distribution is less centered on the precise knowledge of the mean. The hydraulic conductivity during storm water flows is described with a mean of $1 \times 10^{-5}$ m/s, a minimum of $1 \times 10^{-6}$ m/s and a maximum of $1 \times 10^{-4}$ m/s.

#### 5.4.2.3 Total defect area

The total defect area in the catchment is the most uncertain parameter besides the hydraulic conductivity. The estimation of the total defect size is influenced by:

- Undetected defects at the pipe joints (detection not possible with CCTV).
- Backwater which remains in the sewer and obscures the sewer bottom during the CCTV inspection. Defects at the sewer bottom are underestimated.
- No CCTV-data on private house connections is available. Private sewer networks may constitute twice the length of the public network. Furthermore, the private sewer networks are likely to be in a worse condition due to the absence of regular inspections.
- CCTV inspection only observes pipe surface but defects may not penetrate the pipe wall.
- Subjectivity of the inspector (crack diameters are frequently only estimates).
- Wrong assumptions in defect area calculations (e.g. 2 m length for a typical longitudinal crack).

The total defect area is described with a log-normal distribution (Fig. 5-15) with a mean of 49.23 m², a minimum of 24.61 m² and a maximum of 73.84 m². This reflects a range of +/- 50 % as also specified in (DeSilva *et al.*, in print). The standard deviation is set to a tenth of the mean, corresponding to 4.92 m².

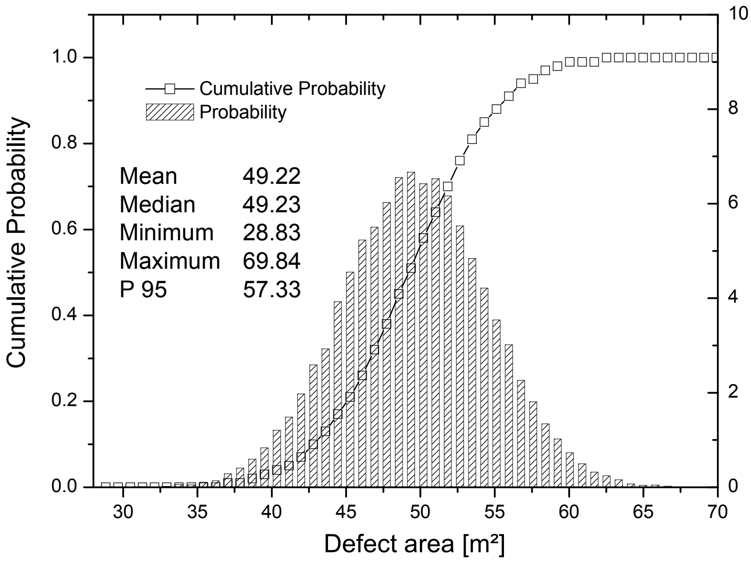



Fig. 5-15: Probability distribution assigned to the sewer defect area in the Rastatt sewer network.

#### 5.4.2.4 Percentage of sewers below the groundwater table

As specified in chapter 5.3.1, the Rastatt sewer network is partly submerged by groundwater and no exfiltration can occur. To reflect this circumstance, the total defect area is reduced according to the percentage of submerged sewers. The percentage of sewers below the groundwater table is described with a triangular distribution with a mean of 9.58%, a minimum of 5.43% and a maximum of 37.50%.

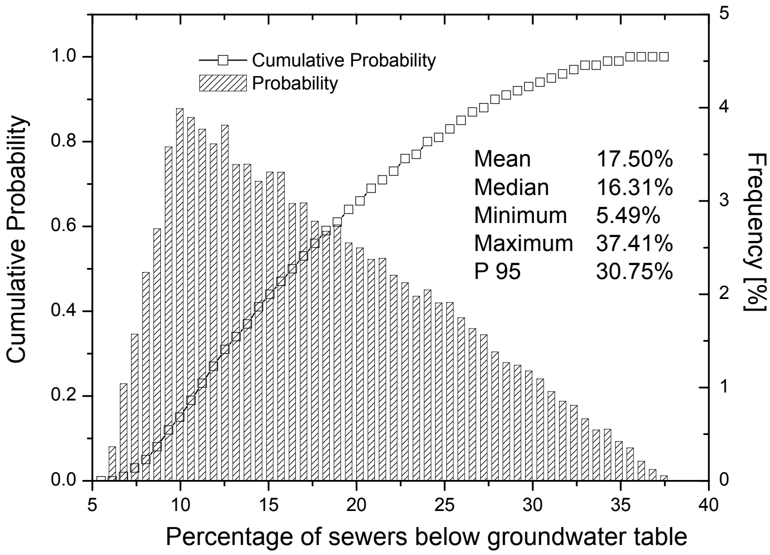



Fig. 5-16: Probability distribution assigned to the percentage of sewers below groundwater table.

#### 5.4.2.5 Water level in the sewer

The water level inside the sewer is also denoted as fill level in the following. To characterise typical water levels in the sewer, 19200 measurements taken between 26.11.04 and 9.3.05 at the focus monitoring test site Danziger Strasse (see also chapter 3.5) were analysed. The mean water level recorded was 6 cm. With a probability of 95 % the water level was below 8.7 cm. The maximum fill level recorded during the period was 30 cm. Higher water levels are also possible and were observed by local inhabitants (back flushing of sewers into the house cellars is a known problem in Rastatt) but not recorded during the four months of observation.

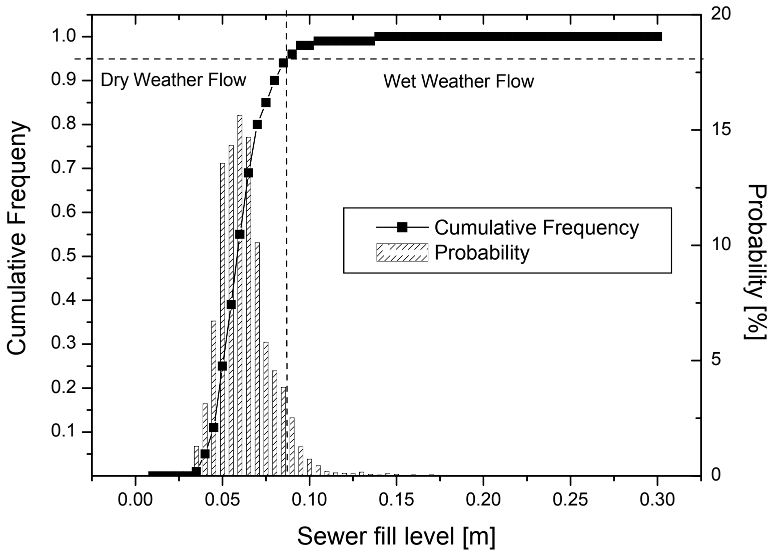



Fig. 5-17: Histogram of sewer fill level at RA-Danziger Str.

For the purposes of the further modelling exercises, a separation between dry weather flow conditions and storm weather flow conditions was introduced at the point of 95 % cumulative probability (in this case 8.7 cm fill level). It is assumed that above this fill level, the supply of sewage to the leak area is too infrequent to allow a strong development of the colmation layer. In addition, the wastewater during these stom events does not carry significant amounts of particulate matter and nutrients, which further inhibits the development of a colmation layer. The expert guess of the separation between dry and wet weather flow conditions is however an additional factor of uncertainty.

The fill level is described with a log-normal probability distribution, with a mean of 0.06 m, a minimum of 0.01 m and a maximum of 1 m. The standard deviation is set to 0.02 m. Values less than 0.087 m are assigned to the dry (or normal) weather flow conditions and values above 0.087 m to wet (storm) weather flow conditions.

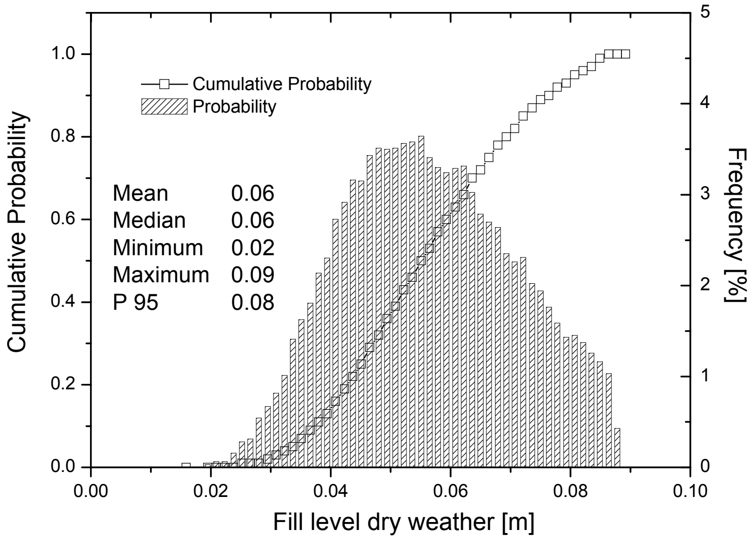



Fig. 5-18: Probability distribution assigned to the fill level at dry weather flow conditions.

#### 5.4.2.6 Frequency of dry weather flow

The ratio between low and high water levels in the sewer was assessed based on the evaluation of rainfall data and on direct water level measurements in the sewer. However, no reliable data exist yet on the sewer water level required for a substantial destabilisation of the clogging layer and on the switching between high and low conductivity values of the clogging layer in the leak. In other words: it is unknown, how often and at which water levels the colmation layer is removed or damaged. Also, the distribution of the colmation layer within the pipe (as depicted in Fig. 5-11) is not known. Based on the water level measurements in the sewer Danziger Strasse and the corresponding histogram (Fig. 5-17), it was assumed that storm events occur only during 5 % of the year. However, this definition is somewhat arbitrary and an uncertainty range between 1 % and 10 % of the year is entered into the Monte Carlo simulations.

The percentage of dry weather conditions throughout the year is described with a log-normal probability distribution, with a mean of 94.54 %, a minimum of 90% and a maximum of 99% (Fig. 5-19). The standard deviation is set to 9.26%.

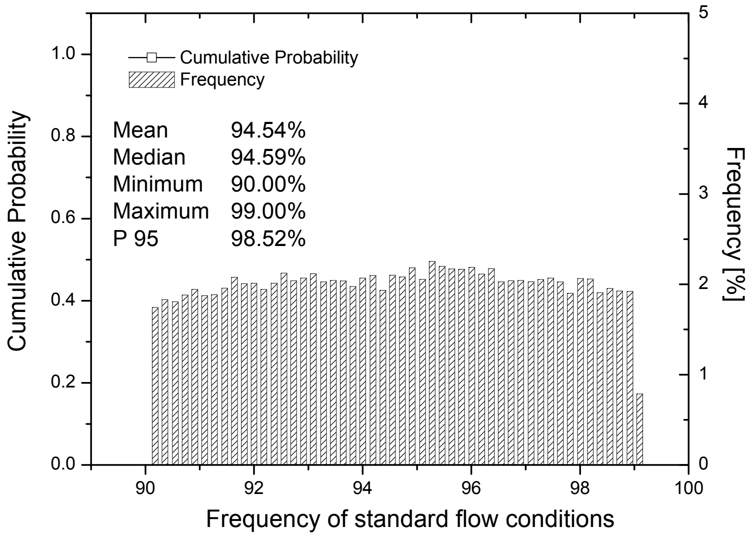



Fig. 5-19: Probability distribution assigned to the frequency of dry weather flow conditions.

#### 5.4.2.7 Thickness of colmation layer

The thickness of the colmation layer is dominantly assumed to be 2 cm in the literature. Similar observations were also made within the DFG research group on sewer leakage in Karlsruhe. The thickness of the colmation layer is described with a log-normal probability distribution, with a mean of 0.02 m, a minimum of 0.01 m and a maximum of 0.03 m. The standard deviation is set to 0.01 m.

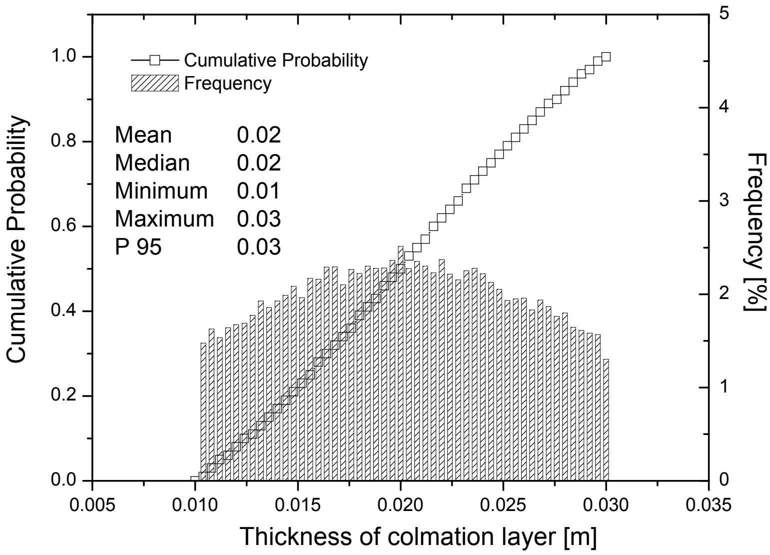



Fig. 5-20: Probability distribution assigned to the thickness of the colmation layer.

### 5.4.2.8 Boron concentration in the sewage

Typical boron concentrations in the Rastatt sewage were assessed based on the results of an intensive sampling campaign which was performed between 13.10.2004 and 25.10.2004 (Wolf et al., 2005) for three sewer location in Rastatt. The results were revisited with regard to a correlation between sewer fill level and boron concentrations (Fig. 5-21). Two different sections were used to define boron concentrations for dry weather flow and during rain events.

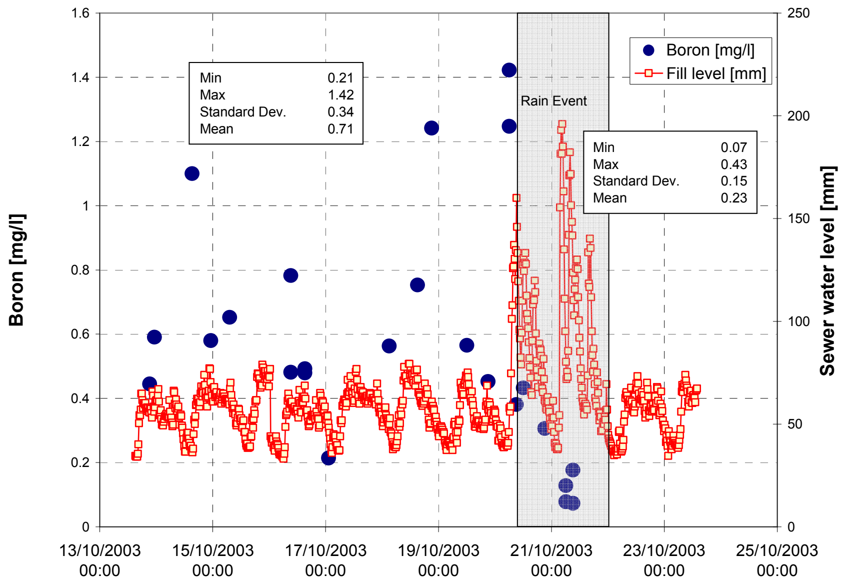



Fig. 5-21: Boron concentration in the sewage measured in at the sewer Danziger Strasse in comparison with fill levels.

Within the Monte Carlo analysis, the concentration of boron in sewage during dry weather flow is described with a log-normal probability distribution, with a mean of 0.71 mg/l, a minimum of 0.21 mg/l and a maximum of 1.42 mg/l. The standard deviation is set to 0.34 mg/l.

The concentration of boron in sewage during wet weather flow is described with a log-normal probability distribution, with a mean of 0.23 mg/l, a minimum of 0.07 mg/l and a maximum of 0.43 mg/l. The standard deviation is set to 0.15 mg/l.

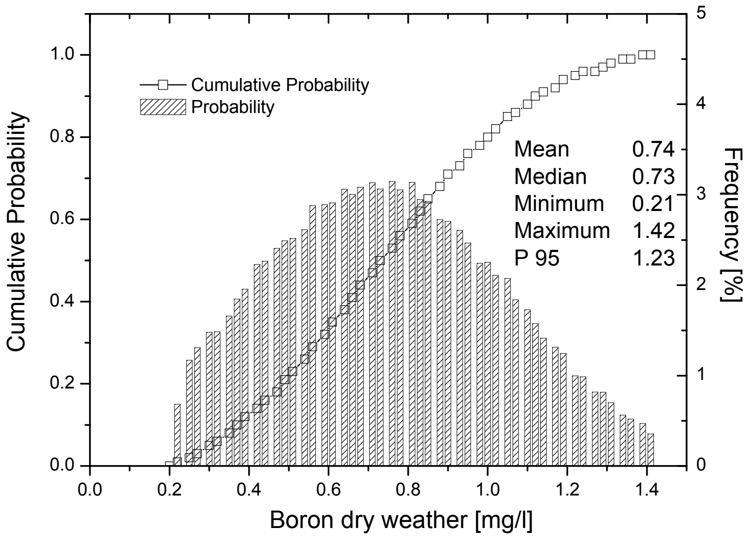



Fig. 5-22: Probability distribution assigned to the boron concentration in the sewage during dry weather flow.

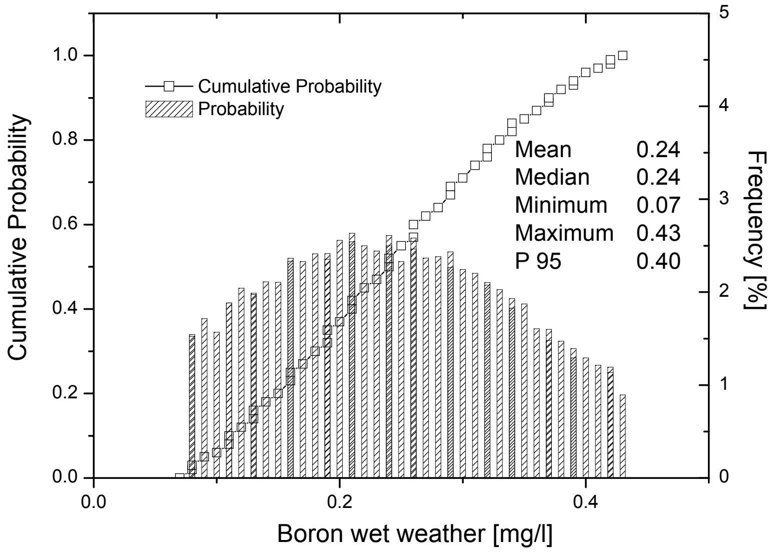



Fig. 5-23: Probability distribution assigned to the boron concentration in the sewage during wet weather flow.

#### 5.4.2.9 Chloride concentrations in the sewage

From the analysis of chloride concentrations measured at different flow conditions in the sewer Danziger Strasse, a dilution with factor 5 was observed. As the concentrations of chloride measured at the Danziger Strasse are rather high compared to other measurement points, the average of 111 mg/l chloride from different sewage measurement in Rastatt is used for further considerations.

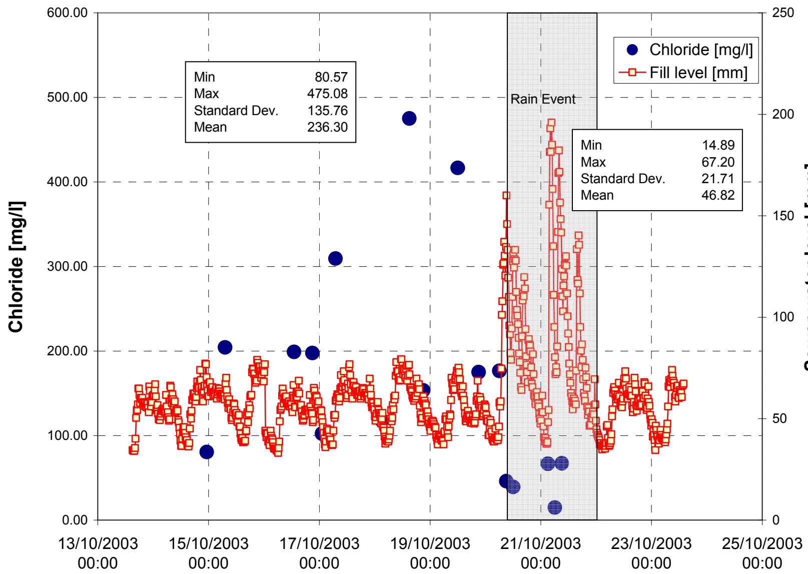



Fig. 5-24: Chloride concentration in the sewage measured in at the sewer Danziger Strasse in comparison with fill levels.

## 5.5 Results

### 5.5.1 Probability

#### 5.5.1.1 Volumes

50000 runs were performed with each input variable randomly chosen within the specified boundaries. The forecast returned a mean leakage rate rate of 0.99 mm/a related to the entire city area (Fig. 5-25). The forecast is characterised by a rather high standard deviation of 0.66 mm/a and a total variance of 0.43 mm/a. The minimum is at 0.02 mm/a and the maximum at 10.38 mm/a.

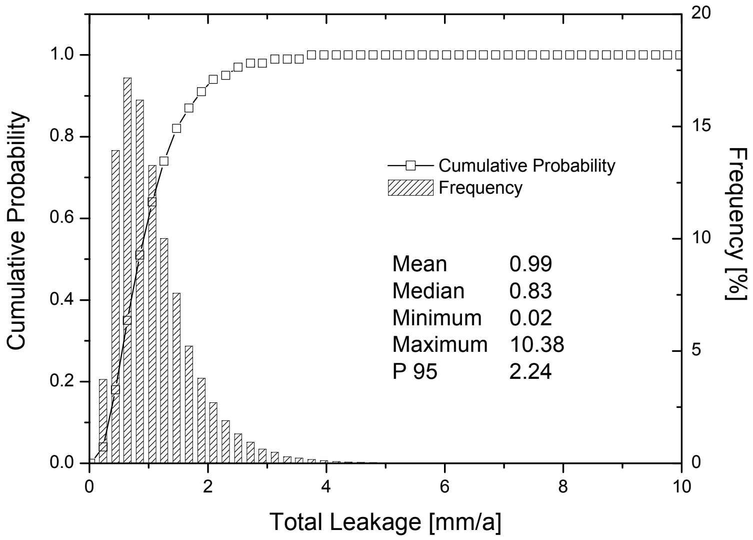



Fig. 5-25: Annual leakage rates computed for the entire city area.

With a probability of 95 % the leakage rate will be below 2.24 mm/a. For the mean, this equals leakage rates of 5.34 l/d at a single leak which corresponds reasonably well with the measurements at the test site Kehler Strasse where an average rate of ca. 8 l/d was measured for the entire test duration and ca. 3 l/d after stabilisation of the exfiltrating conditions.

Tab. 5-5: Results of the Monte Carlo simulation for total leakage rates.

| Scenario | City area | Exfiltration rate | | | | related to defect area | At a single leak |
|---|---|---|---|---|---|---|---|
| | mm/a | m³/a/km | l/d/m | l/s/km | % dwf | l/d/cm² | l/d/Leak |
| MC Mean | **0.99** | 62.6 | 0.172 | 0.002 | 0.20 | 0.057 | 5.338 |
| MC 95 % Cumulative probability | **2.24** | 141.7 | 0.388 | 0.004 | 0.46 | 0.13 | 12.078 |
| MC Max | **10.38** | 656.6 | 1.799 | 0.021 | 2.15 | 0.60 | 55.967 |

#### 5.5.1.2 Boron loads

Besides the exfiltration volumes also the load of boron to the soil-aquifer system from sewer leakage was investigated. The results reflect a similar pattern as the exfiltration volumes but are not linear dependent on them. The specification of low boron concentrations during periods of high exfiltration rates gives more weight to the exfiltration process during dry weather flow. The forecast returned a mean boron load of 0.45 mg/m²/a related to the entire city area (Fig. 5-26). The forecast is characterised by a standard deviation of 0.34 mg/m²/a and a total variance of 0.12 mg/m²/a. The minimum is at 0.01 mg/m²/a and the maximum at 4.26 mg/m²/a. With a probability of 95 % the boron load will be below 1.1 mg/m²/a.

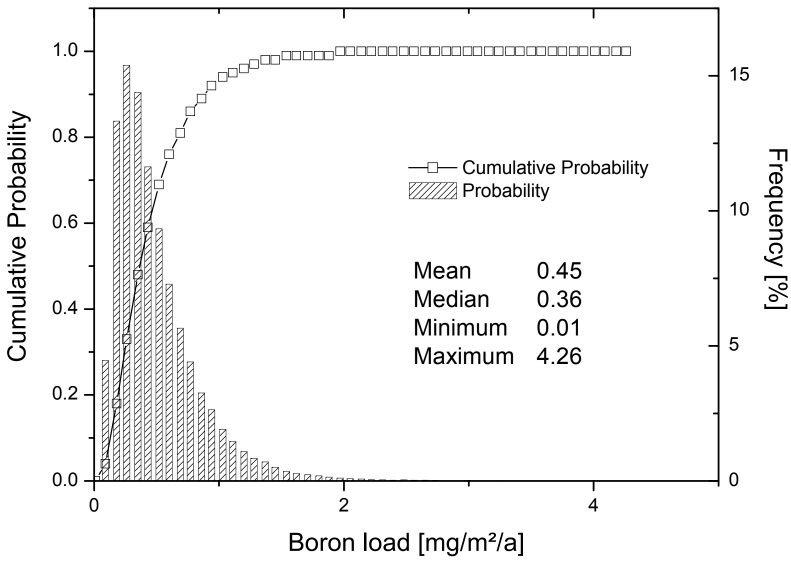



Fig. 5-26: Annual boron loads computed for the entire city area.

5.5.1.3 Chloride loads

To demonstrate the flexibility of the chosen approach, chloride was also calculated as a second marker species. The forecast returned a mean chloride load of 71.14 mg/m²/a related to the entire city area (Fig. 5-26). The forecast is characterised by a standard deviation of 59.79 mg/m²/a. The minimum is at 2.34 mg/m²/a and the maximum at 879.21 mg/m²/a. With a probability of 95 % the chloride load will be below 186 mg/m²/a.

### 5.5.2 Sensitivity

Sensitivity is defined as the amount of uncertainty in a forecast that is caused by both the uncertainty and model sensitivity of an assumption. The model sensitivity itself is defined as the overall effect that a change in an assumption produces in a forecast. This effect is solely determined by the formulas in the spreadsheet model.

The relative importance to gather additional knowledge on an input parameter is therefore defined not only by the uncertainty regarding his determination (e.g. will the defect area be equal 10 m² or 100 m²) but also by his weigth in the formula to calculate the total leakage.

Fig. 5-27 shows the sensitivity of each input parameter as the contribution to the variance of the annual leakage forecast. The most sensitive parameter is the hydraulic conductivity during wet weather flow, even if those conditions are only present during 5 % of the total time. The knowledge hydraulic conductivity of the colmation layer during dry weather has a similar large influence on the result. The result is also sensitive to the frequency of storm events (18 % contribution to the variance). The thickness of the colmation layer poses a comparatively small contribution to the variance as it was specified within rather narrow boundaries (1-3 cm thickness), corresponding to a common understanding in the current literature. Compared to the other factors, the uncertain knowledge of the defect area has a rather limited influence on the results.

In order to achieve a more reliable estimation of the exfiltrating sewage volumes, it is therefore of prime importance to investigate the hydraulic conductivities of the colmation layer, especially with regard to the storm events in the sewer. This directly refers to lacking measurements of the thickness of the colmation layer in different sectors of the pipe.

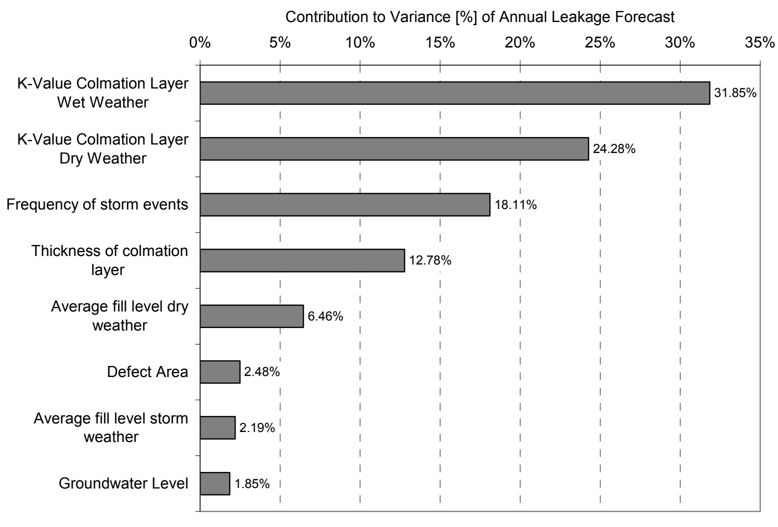



Fig. 5-27: Contribution of individual parameters to the variance in the calculation of total leakage volumes

A completely different picture evolves from the sensitivity analysis of the boron load forecast (Fig. 5-28). The specification of low boron concentrations during periods of high exfiltration rates gives more weight to the exfiltration process during dry weather flow. Consequently, the hydraulic conductivity of the colmation layer during dry weather flow becomes the most influential input parameter, with 44 % contribution to the variance. Secondly, the correct specification of boron concentrations in the sewage has a major impact on the result (14.7 % of the variance). The sensitivity analysis for the chloride forecast shows similar results to the boron load forecast.

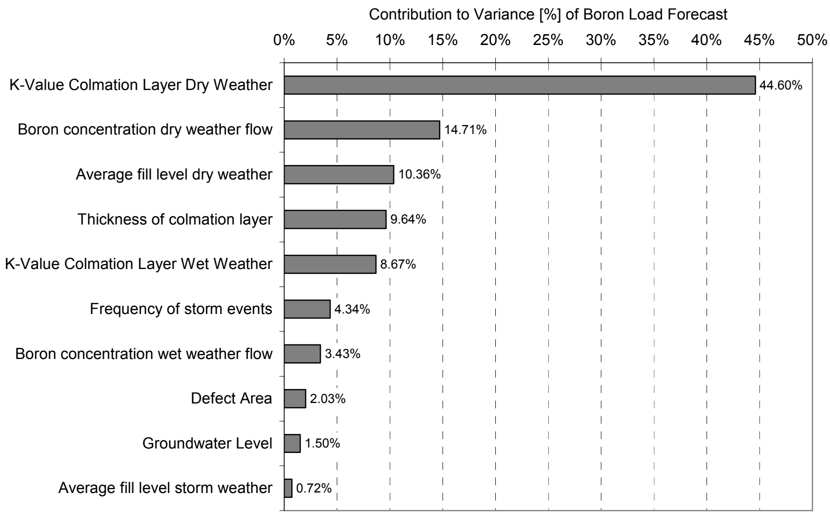



Fig. 5-28: Contribution of individual parameters to the variance in the calculation of boron load to the soil-aquifer system.

This result demonstrates that the exfiltration during dry weather flow conditions is of prime importance with regards to the overall contaminant input to the soil water system. Nevertheless exfiltration during rain events may still pose the larger threat to human health as the large exfiltration volumes lead to short travel times through the unsaturated zone and may effectively rinse down contaminants trapped in the unsaturated zone in periods of dry weather flow.

# 6 Numerical Groundwater Flow and Transport Modelling

## 6.1 Background & Theory

### 6.1.1 Purpose

For the assessment of the impact of leaky sewers on urban groundwater it is necessary to quantify the flow and transport processes inside the groundwater body. This allows the determination of the dilution of the leaking wastewater within the groundwater flow. It is important to note in this respect that groundwater flow velocities may vary over several orders of magnitude in hydrogeological systems, thus forming a very sensitive parameter in the environmental impact assessment. Today, the most accurate technique to describe groundwater flow processes are numerical groundwater models and consequently it was attempted within this thesis to apply numerical groundwater models to the City of Rastatt. In general, groundwater models provide answers to a range of typical questions in urban areas:

- Predicting the impact of groundwater abstraction resulting from construction-stage dewatering, public or private water supply or geothermal groundwater circulation schemes
- Delineation of catchment zones
- Prediction of groundwater quality deterioration as a result of accidental spillages or ongoing urban activities
- Planning of remediation measures for contaminated sites
- Prediction of groundwater quality deterioration from defective sewer systems

Numerical modelling of groundwater flow and transport processes has nowadays become a favourite tool to describe hydrogeological and hydrological problems and especially numerical flow modelling can be considered as a well established and widely applied technique. The concepts have been described extensively in the literature (Anderson & Woessner, 1992; Diersch, 2005; Harbaugh, 2005; Kinzelbach & Rausch, 1995; Kovar, 2003; McDonald & Harbaugh, 1988; Nützmann *et al.*, 1998; Rausch, 2005; Rushton, 1979) and only the governing equations of groundwater modelling are listed within this thesis. With the growing abilities of the numerical solvers and user interfaces, users tend to construct very complex groundwater models which often introduce additional uncertainties rather than reducing them (Diersch, 2001).

### 6.1.2 Basic equations of groundwater flow

The basic formula to describe groundwater flow in porous media is derived from the combination of the Darcy-Equation and the principle of mass conservation. The Darcy-Equation is a phenomologically derived constituative equation that can be transformed to relate the pore velocity with the hydraulic gradient and a constant of proportionality. For the general condition of a three dimensional groundwater flow field it may be written as:

$$\vec{v} = K_f \vec{\nabla} h \qquad \text{Eq. 6-1}$$

with:

$$\vec{\nabla} h = \frac{\partial h}{\partial x} + \frac{\partial h}{\partial y} + \frac{\partial h}{\partial z} \qquad \text{Eq. 6-2}$$

| | | |
|---|---|---|
| using: | $v$ | groundwater velocity |
| | $K_f$, | hydraulic conductivity |
| | $\nabla$ | three dimensional nabla-operator |
| | $\partial$ | partial differential |

In the case of flow through an isotropic porous medium the constant of proportionality becomes a scalar and is named as the coefficient of hydraulic conductivity $k_f$. In the case of an anisotropic medium, the symmetric coefficient of hydraulic conductivity must be used.

Now a mass balance must be performed to arrive at the transient groundwater flow equation together with Darcy´s law. This balance is analogous to the energy balance used in heat transfer calculations to arrive at the heat equation. It is simply a statement of accounting, that for a given control volume, aside from sources or sinks, mass cannot be created or destroyed. The conservation of mass states that for a given increment of time ($\Delta$t) the difference between the mass flowing in across the boundaries, the mass flowing out across the boundaries, and the sources within the volume, is equal to the change in storage. Considering the relative incompressibility of water and aquifer matrix, this mass balance may also be applied to volumes, enabling it to be merged with Darcy´s law. Further assuming that the density of the fluid is constant, the following partial differential equation can be derived:

$$\vec{\nabla}\vec{v} = -S_s \frac{\partial h}{\partial t} + w \qquad \text{Eq. 6-3}$$

With

$$\vec{\nabla}\vec{v} = \frac{\partial v_x}{\partial x} + \frac{\partial v_y}{\partial y} \frac{\partial v_z}{\partial z} \qquad \text{Eq. 6-4}$$

using:
- $v$ — groundwater velocity
- $\nabla$ — three dimensional nabla-operator
- $v_x$, $v_y$, $v_z$ — vector components of the pore velocity
- $\partial h/\partial t$ — transient change of piezometric height
- $w$ — source or sink term, related to the representative volume
- $S_s$ — specific storage coefficient
- $\partial$ — partial differential

The left part of the Eq. 6-3 describes flow in and out of the representative volume via outer borders. The right part of the equation contains a term for the storage of water and a second term to describe sources or sinks (e.g. wells).

Now the general groundwater flow equation can be derived by combining the mass conservation equation with Darcy´s law. The pore velocity can be replaced by the piezometric height, resulting in the Laplace-Equation for a homogeneous isotropic medium (Kinzelbach & Rausch, 1995):

$$\frac{\partial^2 h}{\partial x^2} + \frac{\partial^2 h}{\partial y^2} + \frac{\partial^2 h}{\partial z^2} = 0 \qquad \text{Eq. 6-5}$$

This equation describes the distribution of piezometric heights depending on the position within the flow field. This basic equation of groundwater flow is likewise applicable to flow processes in electrical or heat fields. The transient flow of groundwater is described by a form of the diffusion equation, similar to that used in heat transfer to describe the flow of heat in a solid (heat conduction).

Analytical solutions to the above mentioned partial differential equation are only available for very simple geometries, homogeneous media and coarse boundary conditions.

These partial differential equations describe the physical phenomena in an unspecific and general way. They do not contain any information about a specific study area and have an unlimited amount of possible solutions. Only with the definition of appropriate boundary conditions it is possible to arrive at a unique solution. This necessity puts a strong weight on the proper selection of the conceptual hydrogeological model as well as on the correct description of the boundary conditions. For instance, wrongly specified boundary conditions may result in a model which is not sensitive to otherwise relevant changes in the input parameters.

Within the software package FEFLOW used in this thesis, the following kinds of boundaries may be specified:

- Dirichlet boundary (Type 1): A fixed piezometric head is described. It will not change during the simulation. Consequently care must be taken that the processes within the area of interest (e.g. the wells) have no influence on the model boundary. This requires either sufficient distance to the boundary or a relatively infinite possibility for water in- and outflow, as usually constituted by a river with unconstrained contact to the aquifer.
- Neumann boundary (Type 2): This boundary condition specifies a prescribed flux perpendiculare to the boundary plane (which is defined by two model slices). The impervious boundary is a special case of the Neumann boundary ($\partial h/\partial n = 0$).
- Cauchy boundaries (Type 3): Also called transfer boundaries. Cauchy boundary conditions allow the prescription of a hydraulic head combined with a transfer coefficient. It is assumed that a layer of lower hydraulic conductivity is separating an upper water body from a lower groundwater body. The transfer coefficient is defined by the ratio of hydraulic conductivity and the thickness of this layer. A typical application is a river with a slightly colmated bottom layer.
- Well boundaries (Type 4): These are a special form of the Neumann boundary in which the inflow or outflow from the model domain is specified for a point or line source inside the groundwater model.

### 6.1.3 Predicting solute transport with numerical models

The solute transport modelling exercises employed in this thesis use the flow field calculated by the numerical groundwater flow model and attach mass concentrations to the transported water volumes. It is assumed that the concentration of solutes (or contaminants) never reaches values large enough to induce significant variation in the density or viscosity of the groundwater, thereby altering the flow field.

The main physical and chemical processes describing the transport of dissolved solutes are (Rausch, 2005):

- Advection
- Molecular diffusion
- Dispersion
- Chemical, biochemical and physico-chemical reactions, including radioactive decay.

Advection, or advective transport is the migration of dissolved species resulting from groundwater flow. Pure advective transport, which would occur if all solute molecules migrate at the same velocity and at the absence of diffusion would produce a solute plume that does not spread or mix. However, not all solute molecules travel at the same velocity and real tracers undergo molecular diffusion, grain-scale dispersion and macrodispersion. As a consequence the plume is further deformed and diluted.

The governing equation for mass transport is also based on the principle of mass conservation:

$$S = \frac{\partial (cn)}{\partial t} = -\nabla \cdot j_{adv} + \nabla \cdot j_{diff} + \nabla \cdot j_{disp} + \sigma \cdot n \qquad \text{Eq. 6-6}$$

using:

| | | |
|---|---|---|
| using: | $\nabla$ | three dimensional nabla-operator |
| | S | mass storage in the control volume, per unit time |
| | $j_{adv}$ | advective mass flux |
| | $j_{diff}$ | diffusive mass flux |
| | $j_{disp}$ | dispersive mass flux |
| | c | solute concentration |
| | $\sigma$ | external source of sink |
| | n | effective porosity |

The expressions for the advective, diffusive and dispersive mass fluxes may be replaced by their individual equations. After the application of the chain rule of derivation and using the assumption that the effective porosity n is constant, the general equation to describe mass transport in the aquifer is:

$$\frac{\partial c}{\partial t} = -u \cdot \nabla c + \nabla \cdot (D \nabla c) + \sigma - \frac{q_s}{n}(c - c_s) \qquad \text{Eq. 6-7}$$

using: ∂c/∂t transient change of solute concentration
u groundwater velocity
D Dispersion tensor

The mass transport equation may be solved analyrically for simple cases or using a finite difference or finite element approach. If a numerical solution is performed, the influence of spatial and temporal discretisation needs to be considered. For instance, the geometry of the mesh has a distorting influence on the prediction of mass transport if the mesh elements are too large. The result is a overestimated lateral spreading of the contaminant plume (Ohlenbusch, 2001). This effect is called numerical dispersion. Numerical dispersion may also be the product of too large time steps in transient simulations.

In order to keep the numerical dispersion below the natural dispersion, three criteria need to be fulfilled (Kolditz, 1997):

COURANT-Criterium:

$$\left| \frac{v_a \cdot \Delta t}{\Delta l} \right| \leq 1 \qquad \text{Eq. 6-8}$$

NEUMANN-Criterium:

$$\frac{D}{\Delta l^2} \cdot \Delta t \leq \frac{1}{2} \qquad \text{Eq. 6-9}$$

Grid-PECLET-Number:

$$\left| \frac{v_a \cdot \Delta l}{D} \right| \leq 2 \qquad \text{Eq. 6-10}$$

With $v_a$ groundwater velovity [m/s]

| | |
|---|---|
| $\Delta l$ | length of an element in flow direction [m] |
| $\Delta t$ | time step [s] |
| D | dispersion coefficient [$m^2/s$] |

The dispersion coefficient may be calculated in first order approximation based on the dispersivity and the groundwater velocity:

$$D = \alpha \, v_a \qquad \text{Eq. 6-11}$$

With $\alpha$ dispersivity [m]

As general rule, mesh elements should be chosen as fine as possible in order to avoid numerical dispersion and oscillations.

## 6.2 Existing numerical groundwater models around Rastatt

Two local numerical groundwater flow models (water works Rheinwald, water works Sandweier) were built before the onset of this thesis for areas north and south of Rastatt area. They are partly overlapping with the study area and provided important information for the Rastatt models developed in the AISUWRS project. The first one is a 2-dimensional horizontal transient groundwater flow model developed for the water work Karlsruhe-Rheinwald by the municipal waterworks Karlsruhe (Kühlers & Hoffmann, 2002). This FEFLOW finite-element model (44,960 elements) was calibrated using a time series from January 1965 to December 1994 and is predicting the groundwater flow regime north of Rastatt very well. Also for the water works in Sandweier, approx. 15 km south of Rastatt, a two-dimensional steady state groundwater model was set up by the Department of Applied Geology using FEFLOW. In addition to these local models, a regional flow model for the Upper Rhine Valley has been created in the course of the LIFE-project using the finite difference code MODFLOW. This regional groundwater flow model of the Upper Rhine Valley was created jointly by French, German and Swiss researchers using the MODFLOW code. The spatial resolution of the finite difference grid is 500 m x 500 m. The steady state, 2-dimensional model was calibrated for representative high, medium, and low piezometric levels.

Within a diploma thesis supervised by the department of Applied Geology, Klinger (2003) set up a two dimensional, steady state model specific to Rastatt and its immediate surroundings using FEFLOW. This model was used as the basis for the expansion to a three-dimensional mass transport model developed within the AISUWRS project (Schrage *et al.*, 2005; Wolf et al., in print).

## 6.3 Close-Up models

### 6.3.1 Box model

As a pre-study to the general modelling exercises, a conceptual box model for a single leak was set up with a border length of 100 m. The model contains 57,320 elements distributed to 10 layers with a thickness of 2 m each. Fixed head boundary conditions were applied at the upstream (100.4 m a.s.l.) and downstream borders (100.0 m a.s.l.), introducing a hydraulic gradient of 0.4 m / 100 m typical for the flow field in Rastatt. The leak was represented as a injecting well. A fixed concentration of 0.03 mg/l boron was specified at the upstream boundary.

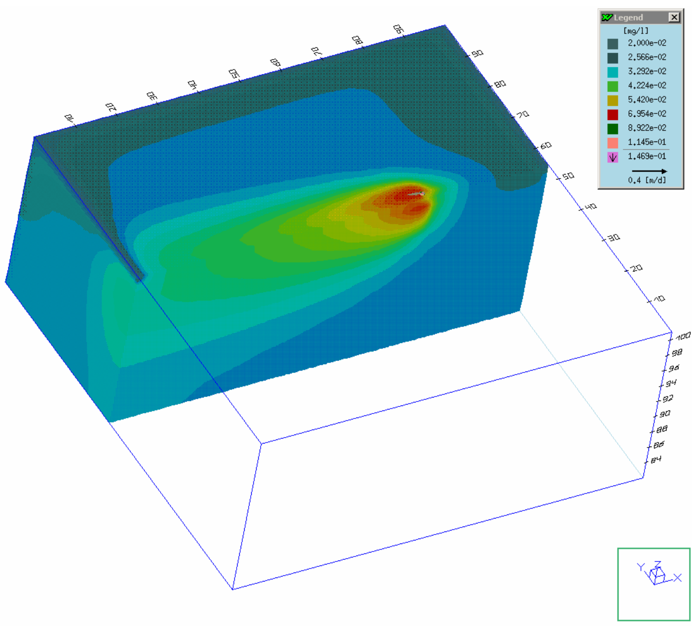



Fig. 6-1: 3-dimensional modelling of a single leaking sewer.

The results of the numerical modelling showed that the quantitative effect of a single sewer leakage with an high exfiltration rate of 1 m³/day is quite limited in a 20 m thick aquifer with hydraulic conductivities of $1x10^{-3}$ m/s. The corresponding rise in groundwater level is less than 1 cm in a distance of 10 m from the leak. However, the qualitative impact of a conservative species extends much further and elevated concentrations (e.g. boron) should be measurable in 70 m distance from the leak. As a conventional leak will have exfiltration rates much less than 1 m³/d from the leak, also the affected volume will be much smaller.

### 6.3.2 Catchment Danziger Strasse

As the next step in the upscaling process, a three-dimensional model was set up for the catchment Danziger Strasse. The sewer catchment Danziger Strasse has also been the pilot study for the application of the UVQ model in the course of the AISUWRS project (Klinger & Wolf, 2004).

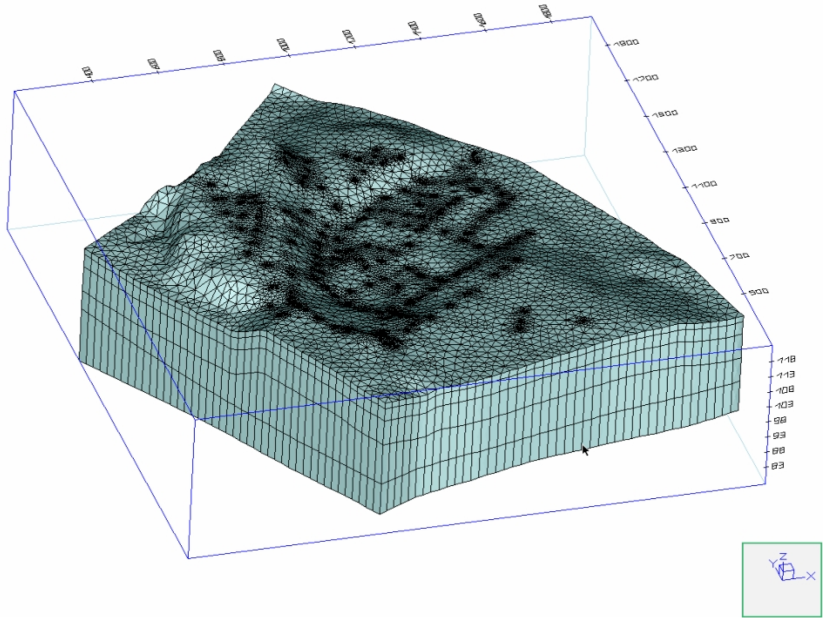


Fig. 6-2: Model geometry and model mesh for the sub-model Danziger Strasse.

The model consists of 6 layers to include Upper and Middle Gravel Layer from a maximum of 123 m a.s.l. down to 77 m a.s.l. It covers an area of 1.87 km² and a volume of 0.08 km³.

Water enters the model via a prescribed flux at the southeastern model boundary, by the specified leaks and by means of groundwater recharge. The model is drained by a fixed head boundary in the north west. The values for the respective boundary conditions were extracted from the existing large scale models (Klinger, 2003; Kühlers & Hoffmann, 2002). Based on this model, several different investigations carried out. Fig 6-2 shows the calculations to describe the dilution of a conservative tracer introduced at a single leak.

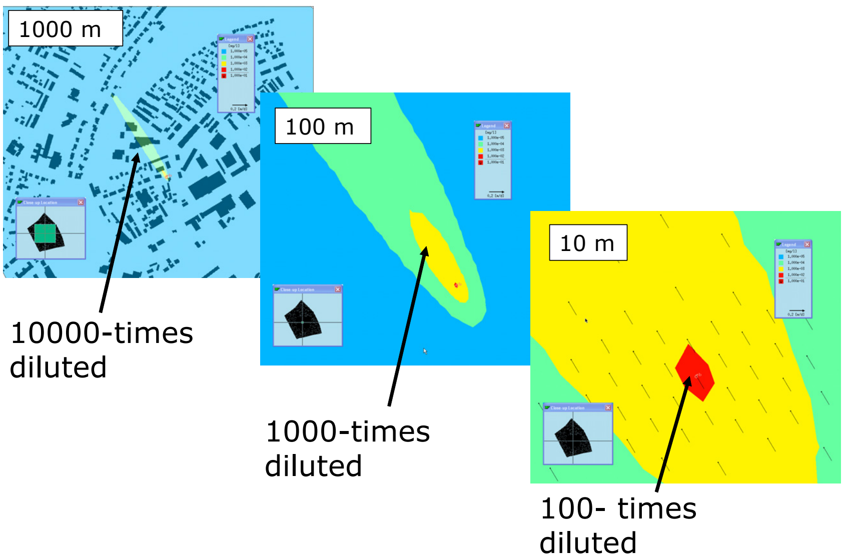



Fig. 6-3: Concentration plumes calculated for a single leak at the test site Danziger Strasse. The scale bars describe the width of the entire window.

The maximum detection distance is listed for different leakage rates as well as for different tracer properties in Tab. 6-1. With a leakage rate of 20 litres per day, a substance with a concentration in the sewage of 1000 times the background concentration, could be still be measured in a distance of 25 m from the leak.

Tab. 6-1: Suitability of tracers to detect a single leak in the Upper Gravel Layer in Rastatt.

| Exiltrationrate | Max. detection distance [m] for a tracer | | |
|---|---|---|---|
| [m³/d] | 100-times | 1000-times | 10000-times |
| 0.02 | 2 | 25 | 300 |
| 0.04 | 5 | 50 | 600 |
| 0.23 | 38 | 350 | >800 |

### 6.3.3 Modelling exercises at the monitoring site Danziger Strasse

Different numerical transport models were set up in order to analyze the observed effects at the monitoring site Danziger Strasse.

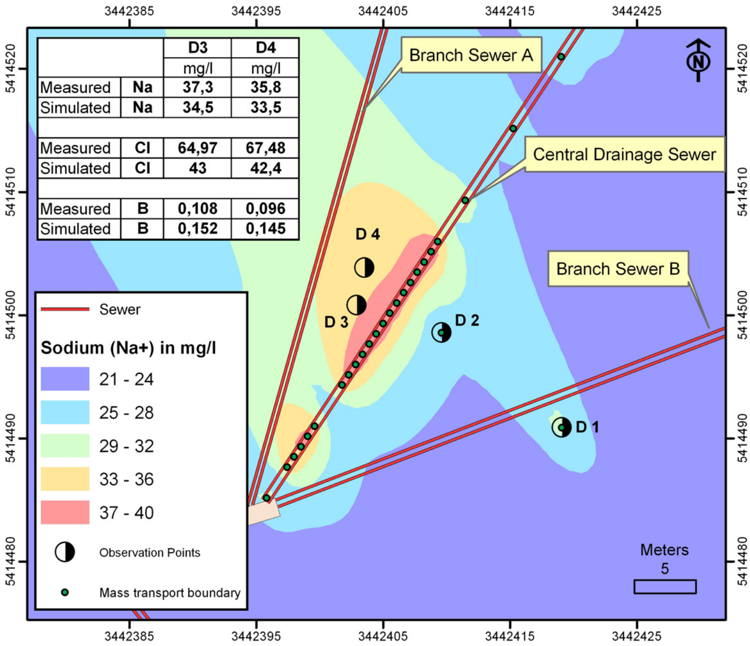



Fig. 6-4: Steady state modelling performed for the sewer monitoring site Danziger Strasse.

The basic model is defined by 47,500 elements and 6 slices. With the inverse calibration it was possible to reach a good agreement between measured and modelled concentrations at the downstream wells D3 and D4 if an exfiltration of 60 l/d with a concentration of 1 mg/l boron was applied at 15 leaks. However, due to the uncertain influence of the suspected leaky house connection close to the observation well D1 the steady state model results for the monitoring site are not detailed within this thesis. The same accounts for the transient model calculations which were performed to describe the impact of a rain event in the sewer on the electrical conductivity of groundwater in the close observation wells.

## 6.4 City-scale groundwater flow and transport model

### 6.4.1 Generalities and model geometry

Within the scope of the AISUWRS project and the research group for sewer leakage at the University Karlsruhe, a three dimensional groundwater model was set up by a group of people (L. Wolf, C. Schrage, J. Klinger) during the years 2002-2005. It is described in its final form in Wolf et al. (in print). For the purposes of this thesis the basic model has been revised in various details and expanded to include sewer leaks as point sources.

A pre-study had demonstrated that simulated time periods of less than one year are sufficient to attain a quasi steady state. Consequently, in order to save computational time only steady state calculations were performed thereafter.

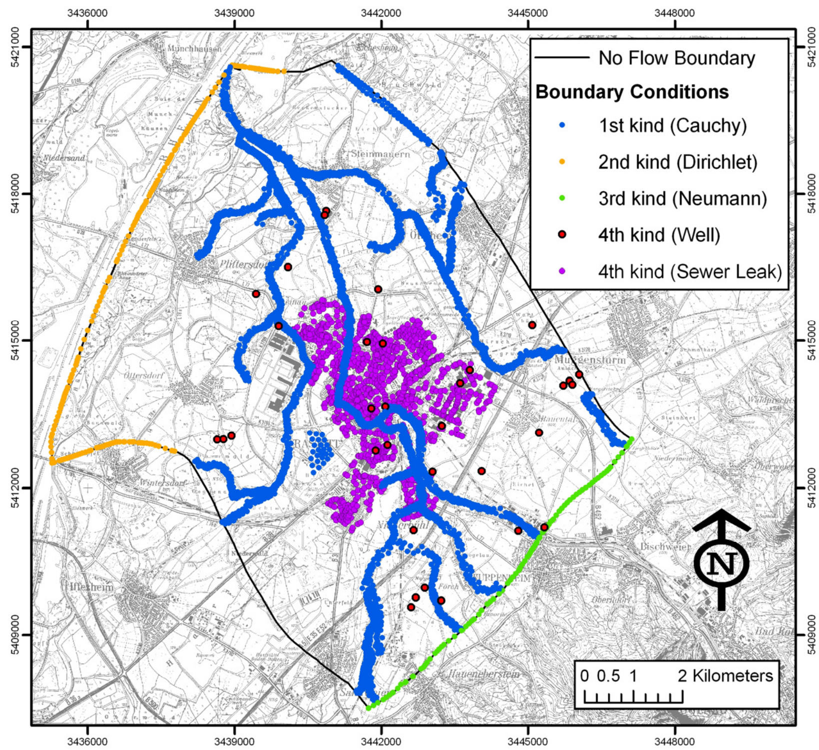



Fig. 6-5: Outline of the model area and boundary conditions for flow.

In order to avoid any constraints exerted by boundary conditions on the model results inside the city area, the model domain was chosen as depicted in Fig. 6-5. The model domain covers an area of about 88 km². Within the model domain lies the urban focus area comprising a total of 11 km². Following the mesh refinements at the individual leaks, the model is defined by 324,180 elements and 195906 nodes.

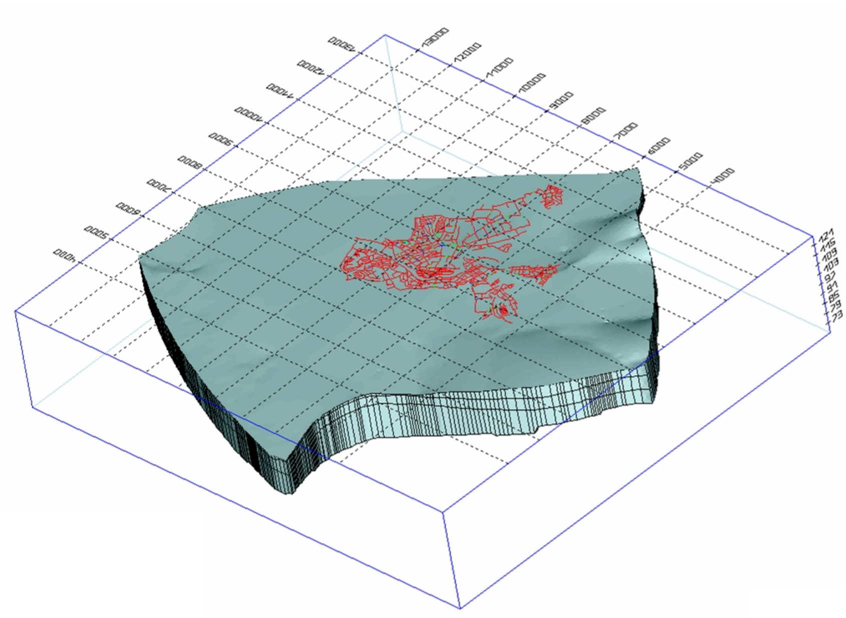


Fig. 6-6: 3-D view of the model body. Red lines denote the Rastatt sewer system, Numbers on grid axes are given in meters.

The model comprises 7 layers defined by 8 slices. Considering the hydrogeological system, only 3 different units need to be included in the model. However, to achieve a better vertical characterisation of the mass transport induced from the top slice, the Upper Gravel Layer is now represented by four layers.

The top layer is required only for numerical purposes and has a fixed thickness of 0.1 meter. It is assigned a low hydraulic conductivity of $1x10^{-6}$ m/s. As a result, the groundwater recharge specified at the top passes through the first layer and acquires the mass concentration specified with the fixed concentration boundary conditions at slice one. This procedure is an effective means to assign a certain concentration to an areal flux in FEFLOW. The low conductivity of the top layer ensures that the fixed concentrations at the top do not influence the groundwater flow in layers beneath by dispersion processes. The topmost slice was specified as “free & movable” according to the BASD technique described in Diersch (2005).

Tab. 6-2: Basic characteristics of the employed model layers. OKL = Upper Gravel Layer, OZH = Upper Interlayer, MKL = Middle Gravel Layer.

| Layer | Strat. Unit | Hydro-geological Unit | Top Elevation [m a.s.l.] | | Thickness [m] | | | Hydr. Conductivity Kx [m/s] | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | Min | Max | Min | Max | Mean | Min | Max | |
| 1 | Younger Quaternary | OKL | 108.8 | 128.1 | 0.10 | 0.10 | 0.10 | 1.0E-06 | 1.0E-06 | |
| 2 | | OKL | 108.7 | 128.0 | 0.10 | 0.10 | 0.10 | 2.3E-05 | 6.0E-03 | * |
| 3 | | OKL | 107.7 | 127.0 | 1.00 | 1.00 | 1.00 | 2.3E-05 | 6.0E-03 | * |
| 4 | | OKL | 106.7 | 126.0 | 1.00 | 1.00 | 1.00 | 2.3E-05 | 6.0E-03 | * |
| 5 | | OKL | 107.6 | 126.9 | 0.10 | 23.92 | 10.18 | 2.3E-05 | 6.0E-03 | * |
| 6 | | OZH | 91.6 | 122.6 | 0.10 | 8.98 | 4.95 | 2.0E-06 | 6.0E-03 | |
| 7 | | MKL | 87.2 | 116.3 | 0.59 | 25.18 | 11.39 | 2.3E-05 | 5.0E-03 | |

* = deeply excavated gravel pits penetrate the Upper Gravel Layer locally. A conductivity of 1 m/s was assigned in this case

Also the second layer possesses a thickness of only a few decimetres. It was implemented to describe the concentrations in the upper part of the aquifer before a mixing with the entire aquifer thickness has taken place. This is thought to represent the conditions during the groundwater sampling exercises, where the pump is placed one meter below the groundwater table in the well.

### 6.4.2 Boundary conditions for flow

#### 6.4.2.1 Fixed Head

The northern-western boundary is occupied by a constant head boundary on the first four slices. This boundary represents the River Rhine as well as the flow field distorted by a series of gravel pits and old meanders close to the river Rhine. The water levels and flow directions in this area are changing quickly and prevent from assigning a no-flow boundary at the edges. While this is depicting the observed flow field rather well, it leads to high rates of transfer between fixed head nodes in close proximity. The height of the fixed head ranges from 111 m a.s.l. to 108 m a.s.l. The River Rhine drains about 53,000 m³/d over this boundary.

#### 6.4.2.2 Flux

In the southwest an integral flux is assigned to all slices. This boundary represents the inflow from the Black Forest Mountains. The value was taken from the model by (Kühlers & Hoffmann, 2002) and adjusted according to the cross-sectional area. It accounts for an inflow of ca. 14,000 m³/d.

#### 6.4.2.3 Transfer

Compared to the model developed within the AISUWRS project, the coefficients describing the transfer between surface water bodies and the aquifer were changed. This was necessary as the former model overestimated the transferred amounts due to discretisation problems. As a result, rivers and small creeks were represented in the model with up to 10-times larger bed width. To correct this misrepresentation without changing the entire mesh geometry, the transfer coefficients were reduced accordingly.

Rivers with permanent water flow were assigned with both transfer in and transfer out functionalities whereas very small creeks acting only as drainage features were only assigned with a transfer out functionality.

#### 6.4.2.4 Wells and Sewer Leaks

A database is assigned to the model containing pumping rates for 33 wells which deliver roughly 19,000 m³/d. The pumping rates represent the situation of October 1986, the date showing a long term average water level. The entire flow model is calibrated to the water levels at this date.

The individual leaks which were identified as relevant from the sewer defect database are described as injecting wells within the groundwater model. For this task the finite element mesh was transformed using a mesh nesting algorithm. The mesh nesting algorithm ensured that a model node was created at the exact position of the leak. However, some defects which either occurr at the same location (e.g. a defect joint combined with a shard crack) or very close to each other (e.g. less than 5 m distance) could not be assigned properly withtin the employed Feflow software. To overcome this problem all leaks within one asset were summed up to a single leak. An asset is defined as the sewer pipe connection between two manholes. The typical length of an asset is 50 m. With this procedure the number of leaks was reduced from 5295 to 1300 without loosing too much detail. From the 1300 defined leaks, 1216 could be implemented using the mesh nesting procedure. The leakage rates of the remaining 84 leaks were summed up and distributed to the 1216 implemented leaks in proportion to their defect size.

### 6.4.2.5 Groundwater recharge

The groundwater recharge was implemented according to the quantifications achieved within the AISUWRS project (Wolf et al., in print) for the complete sewer rehabilitation scenario. It is assembled from three different information sources.

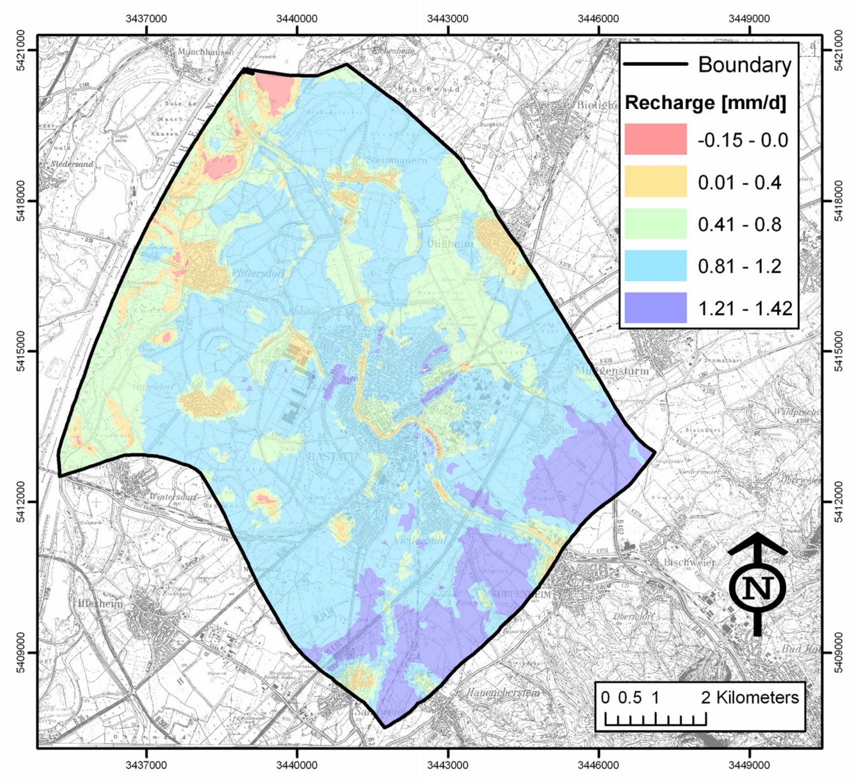



Fig. 6-7: Groundwater recharge specified at top slice. Values derived from Pfützner (1996), Kühlers (2000), Klinger (2003), Wolf et al. (in print).

For most of the area the estimation relies on the unsaturated zone model results calculated with a 30 years time series for the water works in Karlsruhe (Pfützner, 1996). This data was implemented into the transient groundwater model described by (Kühlers & Hoffmann, 2002). Taking into account the basic landuse classes the estimates from Pfützner were transferred to the remaining 20 % of the model area at the Department of Applied Geology (Klinger, 2003). Within the AISUWRS project the recharge for the urban area of Rastatt was completely revised. Based on the combined results from the application of the UVQ models, which took into account urban sealing, diversion of runoff to open spaces, irrigation, drinking water network losses etc., the groundwater recharge within the city area was raised from ca. 90 mm/a to ca. 300 mm/a on average.

### 6.4.3 Boundary conditions for transport

Boundary conditions for transport were set at the top slice and at the south eastern model boundary. Using the workaround mentioned in chapter 6.4.1 it was possible to use first kind boundary conditions for mass throughout the model.

Fig. 6-8 shows the fixed mass concentrations at the top slice of the model. The respective top slice has a fixed thickness of 0.1 m and a low hydraulic conductivity of 1 $\times 10^{-6}$ m/s. All groundwater recharge enters the model via this layer and attains the specified concentration. Due to the low conductivity of the top layer the fixed mass concentrations at the top do only affect the vertical recharge but not the water passing in lateral direction through the major model layers below.

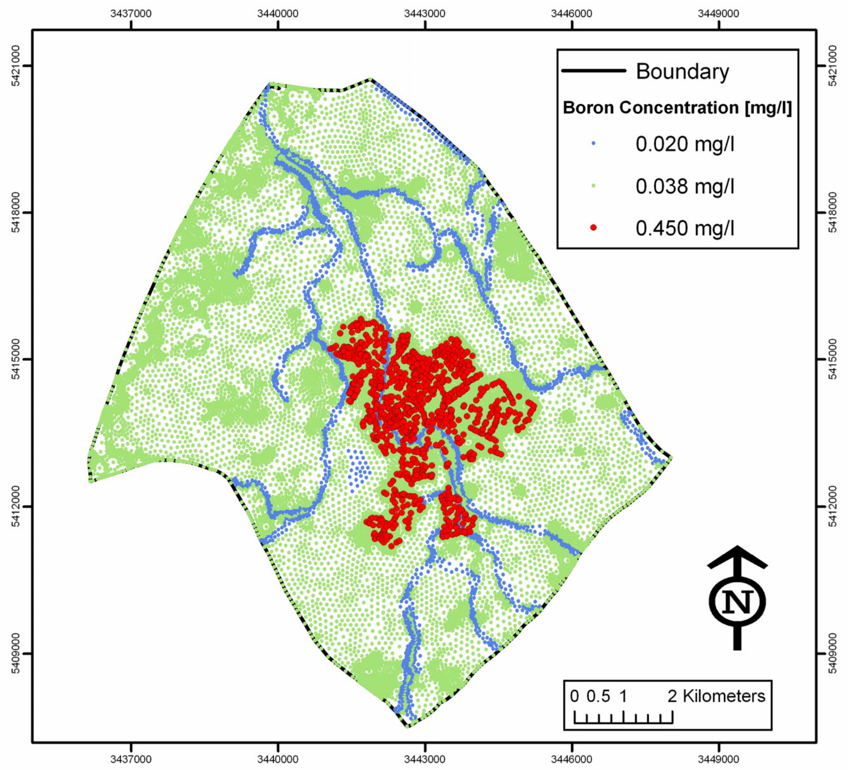



Fig. 6-8: Fixed mass concentrations specified at the top slice of the model.

The mass boundary conditions at sewer leaks were specified according to the results of the Monte Carlo analysis described in chapter 5.5. The boundary conditions describe a sewer leakage rate of 1 mm/a, equivalent to 28 m³/d related to the city area. This water flow is associated with a boron load of 0.45 mg/m²/a, equivalent to an average concentration of 0.45 mg/l boron in the exfiltrating sewage. A concentration of 0.45 mg/l boron in the exfiltrating sewage seems low compared to the typical concentrations of 1 mg/l boron in the sewage inside the pipe. However, the majority of sewage exfiltration occurs during storm events with boron concentrations in the sewage as low as 0.1 mg/l. For the numerical solution of the transport equation a streamline upwinding method is applied. The longitudinal dispersion was set to the default of 5 m with the transversal dispersion at 0.5 m. These default dispersion lengths are rather small and are suited only for a problem dimension of roughly 10 m, which is the key area of dilution around a sewer leak. However, an appropriate grid refinement is only given close to the leaks. With increasing distance to the leak and growing element sizes, the dispersivity is underestimated. The model therefore tends to oversestimate the lateral spreading of an introduced pollutant. However, as the results of the first runs already indicated that the model is governed by other limitations, no further effort was made to derive at a refined implementation of dispersion coefficients. The numerical transport model calculations manily serve to demonstrate the involved mechanisms.

### 6.4.4 Model calibration

A model calibration exercise had already been performed in the course of the AISUWRS project. The AISUWRS model was calibrated using an inverse parameter estimation procedure (PEST). For this reason the model domain was divided into 20 patches of homogeneous hydraulic conductivity (denoted as calibration grid in Fig. 6-9). Within the PEST procedure, the hydraulic conductivities in the respective patches was automatically changed until a minimum difference between measured and modelled hydraulic heads was obtained at the 12 observation wells (Schrage *et al.*, 2005; Wolf et al., in print). However, the PEST calibration produced hydraulic conductivities of $8 \times 10^{-3}$ m/s in large parts of the model domain. These high hydraulic conductivities were not in accordance with the measurements obtained during pumping taverage hydraulic conductivities of ca. 2-3 $\times 10^{-3}$ m/s, with up to $5 \times 10^{-3}$ m/s locally (Eiswirth, 2002; Watzel & Ohnemus, 1997).

Within the model calibration exercise for this thesis it was decided to decrease the maximum hydraulic conductivities and to conduct a posterior manual calibration exercise. The maximum hydraulic conductivity was now set to $5 \times 10^{-3}$ m/s in the Middle Gravel Layer and $6 \times 10^{-3}$ m/s in the Upper Gravel Layer. Also the distribution within the patched grid was rearranged. Still, the hydraulic conductivities required for a good representation of hydraulic heads differ from the pattern known from the pumping tests. However, the hydraulic conductivities derived from the pumping tests change over short distances and the validity of the interpolation to larger areas is questionable.

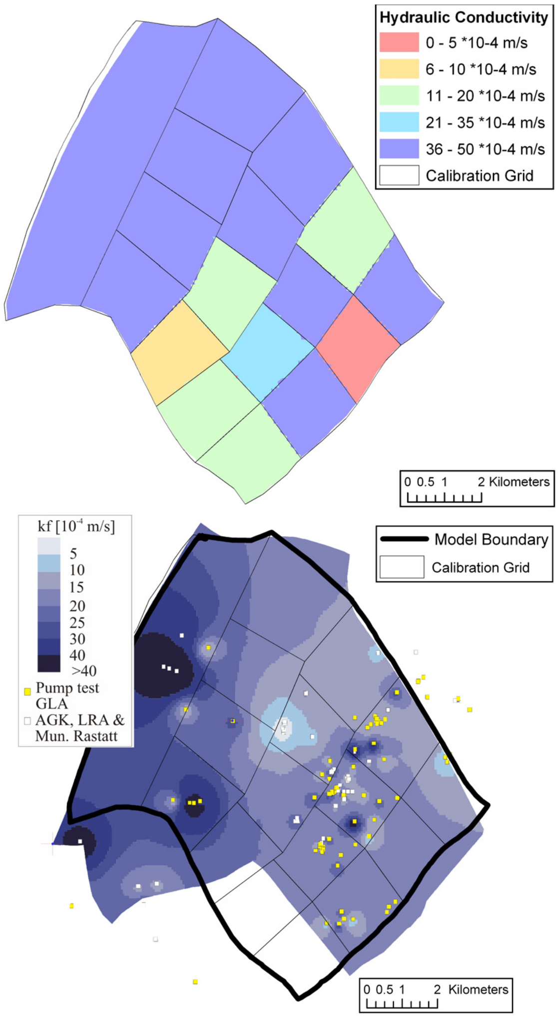



Fig. 6-9: Hydraulic conductivities specified in the numerical model for the Middle Gravel Layer (left) compared with literature values integrated over the combined aquifer system (right) (Eiswirth, 2002).

The gravel pits close to the river Rhine mostly intersect the Upper Gravel Layer only and were implemented with a hydraulic conductivity of 1 m/s into the upper model slices.

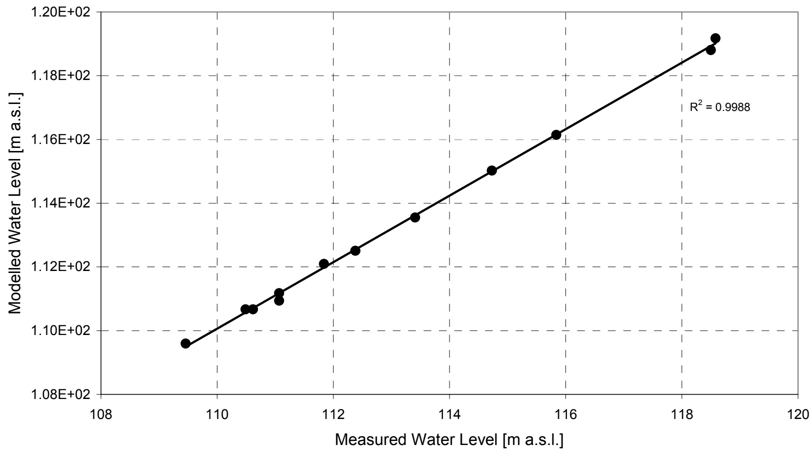



Fig. 6-10: Modelled versus measured water levels (20.10.1986).

With the distribution of hydraulic conductivity in the numerical model depicted in Fig. 6-9, a good agreement between measured and observed hydraulic heads was reached (Fig. 6-10).

Tab. 6-3: Fluid flux balance for the entire model domain.

| | In [m³/d] | % | Out [m³/d] | % | Net [m³/d] |
|---|---|---|---|---|---|
| Fixed head | 260503 | 70.01 | 314322 | 84.56 | -53819 |
| Prescribed Flux | 13299 | 3.57 | 0 | 0.00 | 13299 |
| Transfer (Rivers) | 28250 | 7.59 | 38135 | 10.26 | -9885 |
| Wells | 220 | 0.06 | 19191 | 5.16 | -18971 |
| GW-Recharge | 69801 | 18.76 | 48 | 0.01 | 69753 |
| Total | 372074 | 100.00 | 371697 | 100.00 | 377 |

Tab. 6-3 shows the water volumes exchanged at ineer and outer boundaries of the model. The model shows an minor imbalance of 0.1 % which is derived from inaccuracies in the numerical solution.

### 6.4.5 Results of the city scale groundwater transport modelling

#### 6.4.5.1 Boron

The spatial distribution of boron concentrations was calculated using the boundary conditions and parameters specified above.

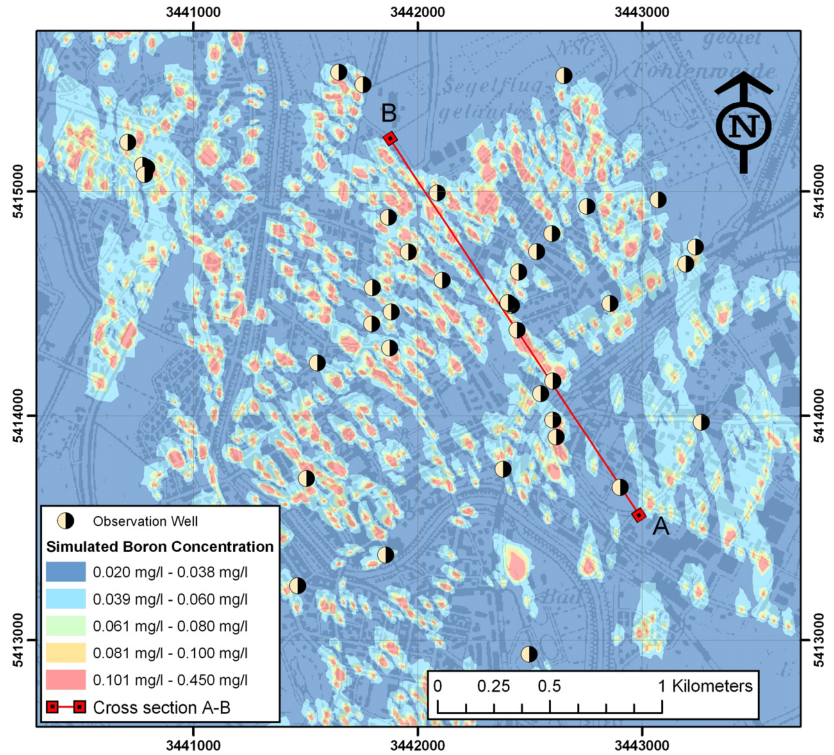



Fig. 6-11: Modelled concentrations of boron in the 4th model layer for an equivalent sewer leakage rate of 1 mm/a and an average boron concentration of 0.45 mg/l in the exfiltrating sewage.

Concentrations significantly exceeding the background concentration of 0.038 mg/l are only visible in the direct vicinity of sewer leaks. Both groundwater recharge and lateral flow exert a strong influence on the concentration distribution and effectively dilute the injected sewage within tens of meters from the leak. However, even with the 324,180 elements currently employed in the model, typical distances between the nodes range from 5 m to 50 m close to leaks and up to 500 m outside the city area. Consequently, the exact prediction of the water quality in an individual observation well cannot be expected. Nevertheless, if a sufficient number of wells is used for the comparison, the average concentrations of the model and the measurement should approach each other.

The problem is further complicated by the vertical distribution of solute concentrations as demonstrated in Fig. 6-12. Due to the small water volumes introduced into the model, the regions with elevated solute concentrations do not penetrate to significant depths. The vertical exaggeration must be considered when interpreting Fig. 6-12. The exaggeration suggests that the plume is mainly moving downwards, which in reality is not the case

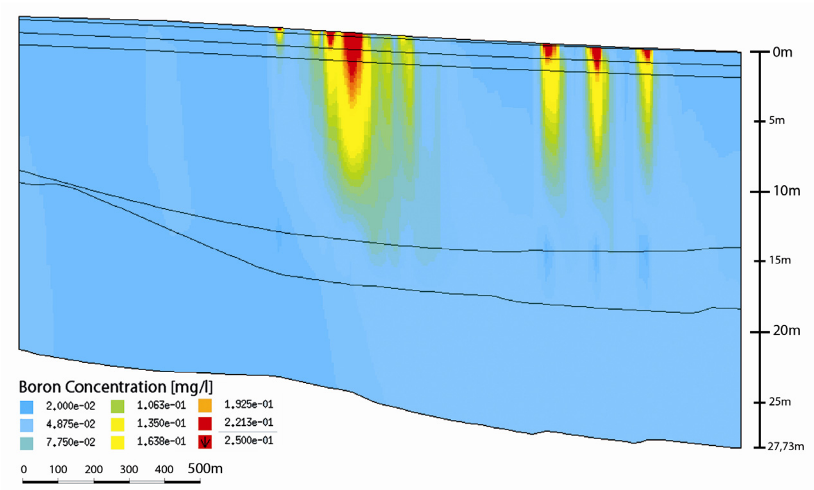



Fig. 6-12: Modelled concentrations of boron along the cross section AB. The vertical dimension is exaggerated 36 times.

The screened depth of the observation well and the position of the pump during the sampling procedure exert a significant influence on the comparison between modelled and measured concentrations. This especially holds true for the 12V submersible pumps used in this study which have pumping rates of approx. 5l/min and attract water from quite a limited radius.

Fig. 6-13 and Tab. 6-4 show the correlation between measured and modelled boron concentrations for 46 groundwater observation wells sited within the urban area. While the agreement at individual measurement spots is weak, the measured average concentration of all wells (0.064 mg/l) is very close to the average modelled concentrations at observation wells in model layer 4 (0.073 mg/l). In layer 5, which marks the top of the Upper Interlayer, the modelled concentrations are generally too low (0.045 mg/l).

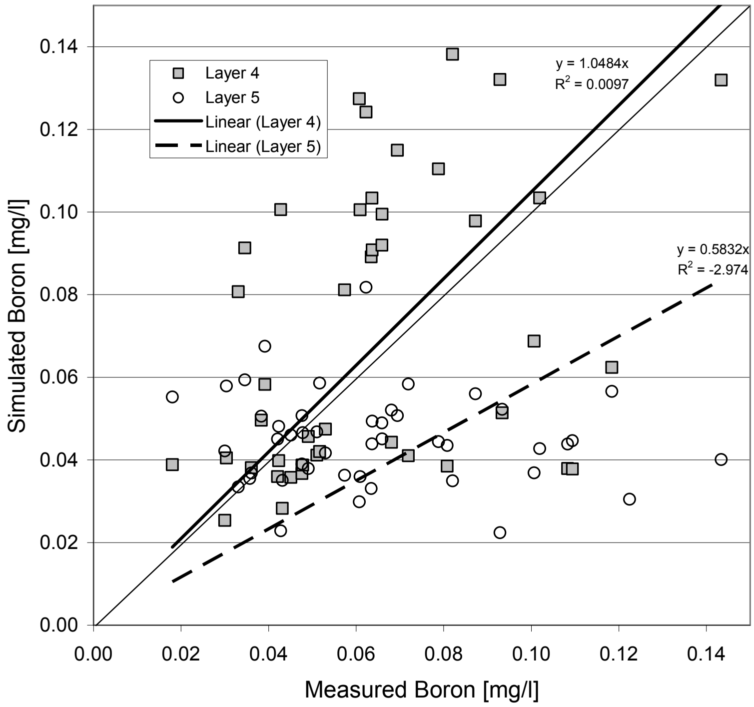



Fig. 6-13: Comparison between modelled and measured boron concentrations at 46 groundwater observation wells within the city area (focus groups 1, 2, 3).

It must be noted that the all the measured boron concentrations derive from the same set of observation wells. The observation wells mainly show a depth between 6-12 m b.g.l. and therefore relate more to the modelled concentrations at layer 4. This also explains the systematically low modelled boron concentrations in layer 5. However, a depth specific sampling programme using packer tests or multilevel piezometers would be necessary to explore and validate the imposed assumptions on the vertical mass transport.

The spatial distribution of the deviation between modelled and measured concentrations is displayed in Fig. 6-14. No clear general pattern or trend is visible in this distribution. Locally, the wells at the eastern border of the town all exhibit modelled concentrations which are too high. This reflects the too high estimated background concentration of boron in this part. Wells modelled with boron concentrations that are too low are usually located within the city. This indicates that the influence from urban boron sources is generally underestimated.

Tab. 6-4: Comparison between modelled and measured boron concentrations at 46 groundwater observation wells within the city area (focus groups 1, 2, 3) for three different model layers.

| Parameter | Unit | Measured | Layer 3 | Layer 4 | Layer 5 |
|---|---|---|---|---|---|
| Min boron concentration | mg/l | 0.018 | 0.023 | 0.025 | 0.022 |
| Max boron concentration | mg/l | 0.143 | 0.208 | 0.170 | 0.082 |
| **Average boron concentration** | **mg/l** | **0.064** | **0.081** | **0.073** | **0.045** |

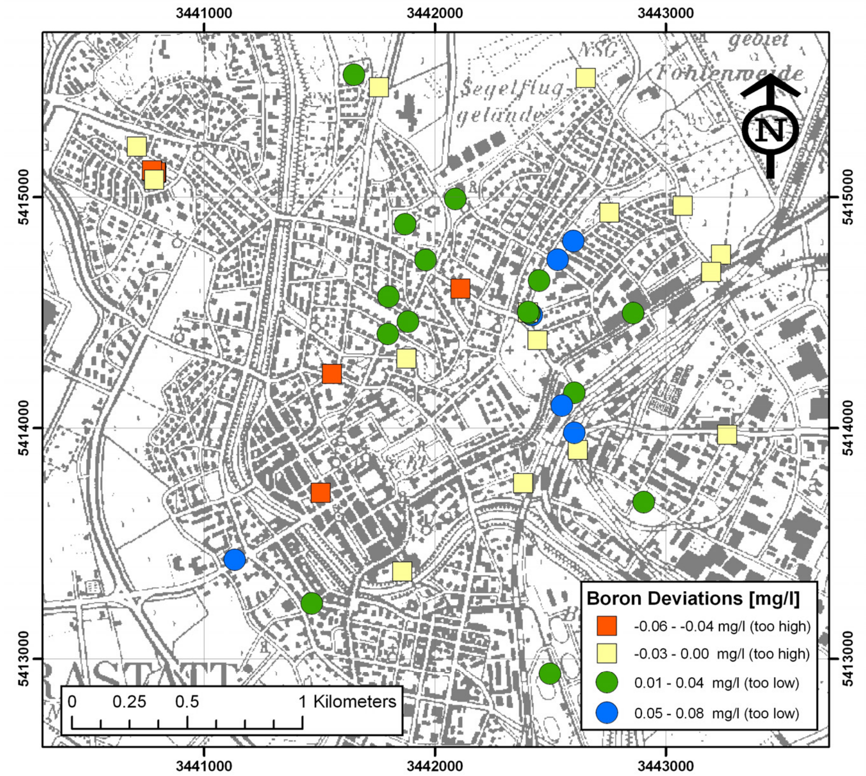



Fig. 6-14: Comparison between modelled and measured boron concentrations at 46 groundwater observation wells within the city area (focus groups 1, 2, 3), approx. 2 m below the groundwater table.

#### 6.4.5.2 Chloride

A different picture evolves for chloride as depicted in Fig. 6-15. The chloride concentrations modelled for slice 4 and 5 generally remain below the measured concentrations. The measured chloride concentrations exhibit an average of 20.67 mg/l chloride, compared to modelled concentrations of 20.62 mg/l chloride in slice 4 and 16.47 mg/l chloride in slice 5. The background concentration of chloride was set to 16.3 mg/l.

Wells with high measured chloride concentrations above 25 mg/l always exhibit concentrations that are modelled too low. This suggests the existence of additional chloride sources not implemented in the model. Wells with low chloride concentrations below 15 mg/l always exhibit concentrations which are modelled too high. One possibility could be that the chloride background concentration is more heterogeneous than assumed, but more likely the event-based and spatial character of road salting and other urban sources are responsible for the observed discrepancies.

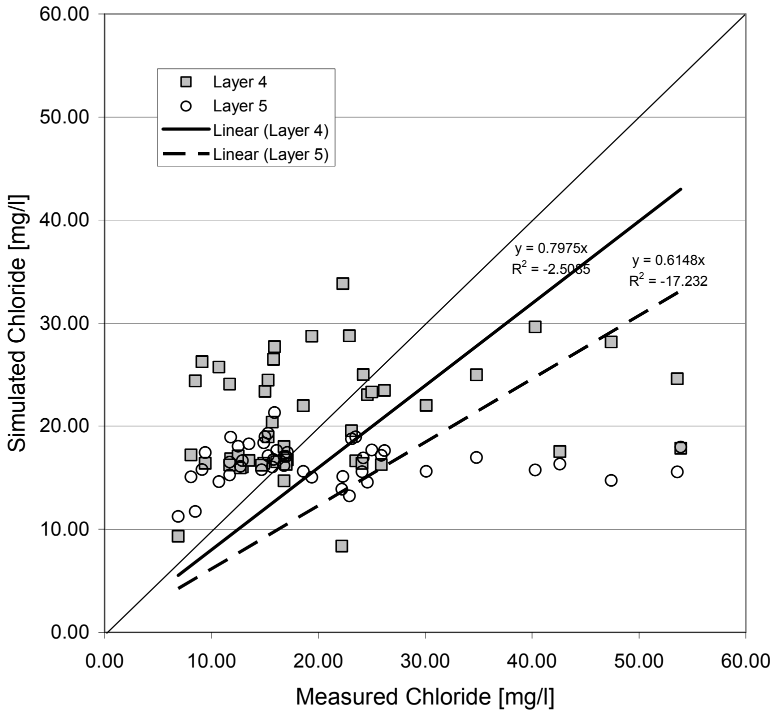



Fig. 6-15: Comparison between modelled and measured chloride concentrations at 46 groundwater observation wells within the city area (focus groups 1, 2, 3).

### 6.4.5.3 Limitations of the transport modelling exercise

A rough agreement between the overall averages in modelled and measured boron concentrations is found only in the comparison at two meters below the groundwater table. If the results from layer 4 and layer 5 are combined, as would be expected from the groundwater sampling exercise, the modelled concentrations are too low. This indicates that the total amount of sewer leakage must be higher than the assumptions for the public sewer network derived from the Monte Carlo analysis. If the concentrations are compared at individual wells, the correlation is very weak, regardless of the depth interval considered.

Reasons for the lack of correlation are:

- Sewer network operators report frequently that the visual inspection of a sewer does not necessarily reflect the water-tightness of the sewer. Comparison between CCTV data and packer tests indicated that many of the noticed cracks do not penetrate the pipe wall. On the other hand, leaky joints are difficult to detect by visual inspection. All in all, the lacking reliability of the CCTV inspection contributes to the mismatch between model and measurement.
- Other sources of boron might be of importance locally, but are not implemented in the model. For instance, even outside of the city, boron concentrations differ between 0.01 mg/l and 0.08 mg/l, without there being a sewer system present. Input from the application of fertilisers must be considered here. In addition, boron stemming from indus-trial processes and industrial soil contamination are also present in the urban area, but the source locations are unknown and not implemented in the model.
- Clearly elevated boron concentrations are predicted within a radius of 10-50 m around a leak. However, due to the reduction from 5295 leaks to 1216 leaks in the model, an additional spatial uncertainty of approx. 25 m (half length of an asset) was introduced into the model.
- The impact of leaky house connections is not considered in the model due to the lack of condition monitoring information. However, as shown by the detailed observations at the test site Danziger Strasse, this influence can be very significant. For example, the boron concentrations upstream the sewer leakage were found to exceed the downstream concentrations. Obviously, a model focusing on leakage from public sewer networks would not be able to reproduce this.

The main conclusions from the transport modelling exercise may be summarised as follows:

- The area of measurable groundwater deterioration around a leak is typically within 10-50 m.
- The model predicts pronounced vertical concentration gradients which are relevant for the comparison between sampled and modelled water quality.
- The measured average boron concentration of 46 wells inside the urban area of Rastatt is comparable to the modelled concentration at 2 m depth below the water table. With regard to the integrated concentration average of the Upper Gravel Layer. The modelled concentrations are too low.
- The measured average chloride concentration of 46 wells inside the urban area of Rastatt exceeds the modelled concentration at the wells.
- The correlation between measured and modelled concentrations at individual wells is weak.
- The numerical groundwater model points to higher sewer leakage rates than the results from the Monte Carlo analysis with a mean sewer leakage rate of 1 mm/a.
- The unreliability of CCTV inspections, the unknown distribution of defective private house connections, the existence of other boron sources than sewers and problems in the mesh discretisation were identified as causes for the lack of correlation between measured and modelled concentrations.

## 6.5 Simplified mass balance approach

Apart from the results from the numerical transport modelling it is also feasible to use the water fluxes quantified with only the numerical flow model. Combined with the results from the different sampling campaigns, a mass balance can be constructed for the central urban area. The balance area outlined in Fig. 6-16 and Fig. 6-17 covers an area of 14.18 km².

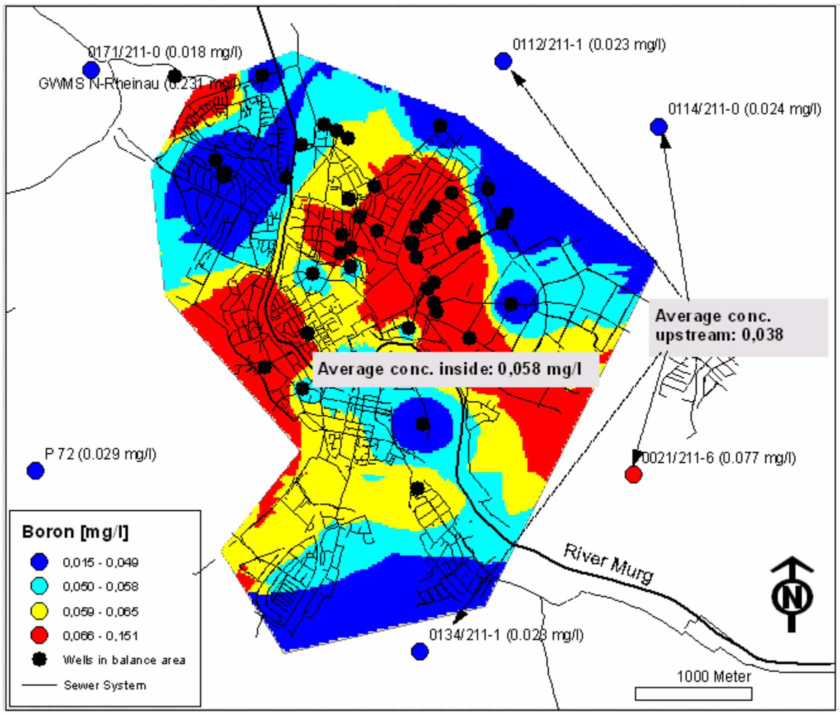



Fig. 6-16: Interpolated concentrations of boron in the Upper Gravel Layer to determine averages inside and upstream of the balance area.

Based on the measurements of boron concentrations during the years 2001-2005 an interpolated map depicting a likely distribution of boron was constructed. Different interpolation methods (Burger & Schafmeister, 2000; Schafmeister, 1999; Zimmerman *et al.*, 1999) were briefly explored but could be visited in more detail. The variogram analysis showed that the density of the observation well network needs to be improved to allow for reliable interpretations. This corresponds to the small scale variations also predicted by the groundwater transport model. Finally, an inverse distance weighting method (power 2, considering 15 neighbours) was applied as it constitutes an exact deterministic interpolator. This included the transfer of information from the randomly distributed observation wells to a regular raster with a cell size of 20 m. Using the concentrations attached to each raster cell, an average concentration for the entire balance area was determined.

The determination of an average concentration of the upstream groundwater was done by using four wells as indicated in Fig. 6-16. The average value of 0.038 mg/l boron is strongly influenced by the concentrations of 0.077 mg/l measured at observation well 0021/211-6. This concentration is significantly above the usual background value of 0.02 – 0.03 mg/l boron. Nevertheless, the number of available observation points is not enough to discard this observation well from further interpretations. It must however be kept in mind that all further mass balance equations would indicate more sewage influence if a background concentration of just 0.024 mg/l is applied.

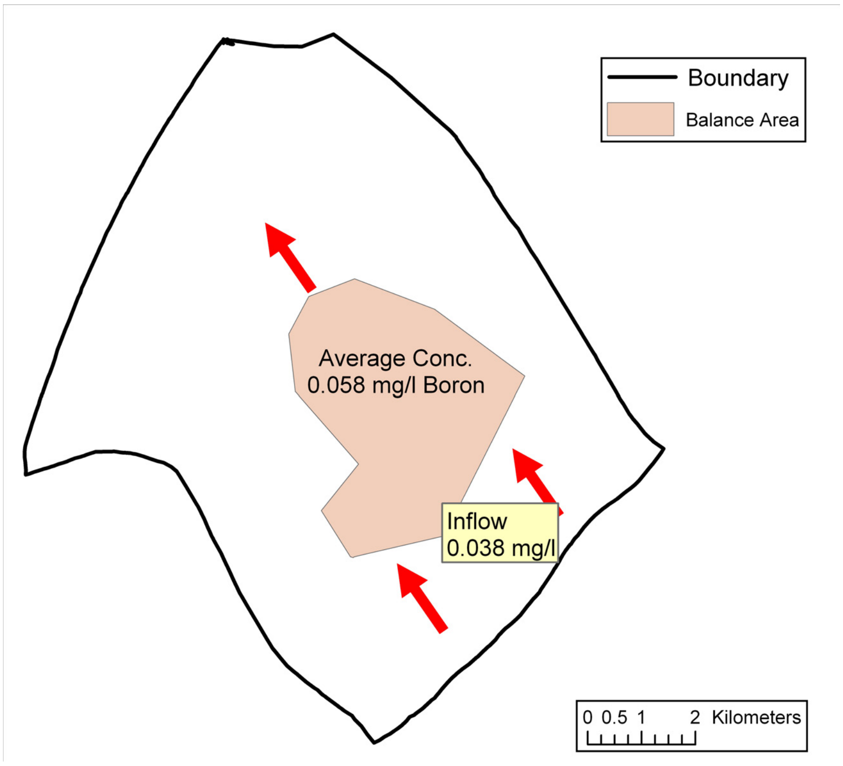



Fig. 6-17: Treating the balance area as a uniform black box.

Fig. 6-17 demonstrates the simplified mass balance concept. Using the budget and fluid flux analyser functions in Feflow, the water balance was set up as shown on the left side of Tab. 6-5. However, the fluid flux analyser implemented in Feflow is only able to provide a first order approximation of the lateral groundwater flow component if a complicated flow field is present. Inaccuracies in the order of 10 % must be accepted (personal communication, WASY GmbH 2006). Compared to the default settings, the error could be diminished by using the discontinuous analysis of the Darcy flux based on nodes rather than elements. All other parts of the water balance were quantified with higher precision using the budget analyser functionality. In a second step, concentrations are assigned to the input water flows. For surface waters a concentration of 0.02 mg/l boron was assumed. The concentration of the groundwater entering the balance area via the lateral groundwater flow was set to 0.038 mg/l. All water leaving the balance area with the lateral groundwater flow was assumed to have the same concentration as the average over the entire balance area, in numbers 0.058 mg/l.

After considering the available information, the following mass balance equation was constructed:

$$SW_{in} \cdot C_{SW} + LGF_{in} \cdot C_{back} + GWR \cdot C_{back} + SL \cdot C_{SL} + A = \left(SW_{out} + WE + LGF_{out} + VGF_{out}\right) \cdot C_{city}$$

| | | |
|---|---|---|
| using: | $SW_{in}$ | Volume of infiltrating surface water |
| | $SW_{out}$ | Volume of exfiltrating surface water |
| | $C_{SW}$ | Concentration in the surface water |
| | $LGF_{in}$ | Lateral groundwater flow entering the balance area |
| area | $LGF_{out}$ | Lateral groundwater flow leaving the balance |
| | $VGF_{out}$ | Vertical groundwater flow leaving the balance area |
| | $C_{back}$ | Background concentration in the groundwater |
| | SL | Sewer leakage |
| | $C_{SL}$ | Concentration in the sewage |
| | $C_{city}$ | Average concentration within the balance area |
| | A | Mass flux from additional sources |
| | GWR | Groundwater recharge |
| | WE | Water abstraction by wells |

Based on the calibrated groundwater flow model, only the total amount of groundwater recharge was known. Therefore, the following relationship was implemented in the calculations:

$$TGWR = GWR + SL + AS$$

using: TGWR Total groundwater recharge
GWR Natural groundwater recharge
SL Recharge received from sewer leakage
AS Recharge received from additional sources

It is now possible to calculate the necessary input from additional sources with a concentration of 0.45 mg/l boron as summarized in Tab. 6-5.

Tab. 6-5: Using a mass balance approach to estimate the amount of sewer leakage based on measured boron concentrations.

| | Water Flux | | | Concentrations | | Mass Flux | | Water Flux | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | In | Out | Net | In | Out | In | Out | In | Out | Net |
| | [m³/d] | [m³/d] | [m³/d] | [mg/l] | [mg/l] | [mg/d] | [mg/d] | [mm/a] | [mm/a] | [mm/a] |
| Surface Waters | 10366 | 0 | 10366 | 0.02 | 0.038 | 207 | 0 | 266.72 | 0.00 | 266.72 |
| Wells | | 1057 | -1057 | | 0.058 | 0 | 61 | 0.00 | 27.19 | -27.19 |
| Layer 1 Lateral GW-Flow | 0 | 0 | 0 | 0.038 | 0.058 | 0 | 0 | 0.00 | 0.00 | 0.00 |
| Layer 2 Lateral GW-Flow | 1701 | 1405 | 296 | 0.038 | 0.058 | 65 | 81 | 43.76 | 36.15 | 7.62 |
| Layer 3 Lateral GW-Flow | 4886 | 15789 | -10903 | 0.038 | 0.058 | 186 | 916 | 125.72 | 406.25 | -280.53 |
| Flux Analyzer Deviation | 3076 | | 3076 | 0.038 | | 117 | 0 | 79.14 | 0.00 | 79.14 |
| Vertical Flow | | 14940 | -14940 | | 0.058 | 0 | 867 | 0.00 | 384.40 | -384.40 |
| GW-Recharge | 11097 | | 11097 | 0.038 | | 422 | 0 | 285.53 | 0.00 | 285.53 |
| Implemented Sewers | 29 | | 29 | 0.45 | 0.058 | 13 | 0 | 0.73 | 0.00 | 0.73 |
| Additional Sources | 2036 | | **2036** | **0.45** | | **916** | | **52.39** | **0.00** | **52.39** |
| Total | 33190.82 | 33190.82 | 0.00 | | | 1925 | 1925 | 853.99 | 853.99 | 0.00 |

In order to balance input and output of boron for the balance area, an additional input of 2036 m³/d sewage (equivalent to 52.38 mm/a) with a concentration of 0.45 mg/l would be necessary. Alternatively, the additional boron input might result from industrial spillages or fertilizer application. However, if leaky sewers are regarded as the only source for boron in the urban area, a leakage rate of 53 mm/a needs to be assumed.

A similar exercise was performed for chloride. A concentration of 71.41 mg/l was assigned to the exfiltrating sewage. In order to balance input and output of chloride for the balance area, an additional input of 3114 m³/d sewage (equivalent to 80.12 mm/a) with a concentration of 71.41 mg/l would be necessary. A part of this input originates from defective house connections. However, the major part of additional chloride input will result from road salting. Furthermore, a former dumpsite of building debris in the industrial part of Rastatt is known to emit signficant amounts of chloride; a problem that was also observed within the BW-Plus project on groundwater quality prognosis using neuronal networks (Hötzl & Liesch, 2005). Chloride is therefore only suitable to constrain and control mass balances set up for other marker species. In order to determine the amount of sewer leakage via the mass balance approach, a quantification of the mass flux from road salting and other sources is necessary. The tools developed within the AISUWRS project would be suited to address this issue (Wolf et al., in print).

Tab. 6-6: Using a mass balance approach to estimate the amount of sewer leakage based on measured chloride concentrations.

| | Water Flux | | | Concentrations | | Mass Flux | | Water Flux | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | In | Out | Net | In | Out | In | Out | In | Out | Net |
| | [m³/d] | [m³/d] | [m³/d] | [mg/l] | [mg/l] | [mg/d] | [mg/d] | [mm/a] | [mm/a] | [mm/a] |
| Surface Waters | 10366 | 0 | 10366 | 6 | | 62196 | 0 | 266.72 | 0.00 | 266.72 |
| Wells | | 1057 | -1057 | | 18.3 | 0 | 19341 | 0.00 | 27.19 | -27.19 |
| Layer 1 Lateral GW-Flow | 0 | 0 | 0 | 16.3 | 18.3 | 0 | 0 | 0.00 | 0.00 | 0.00 |
| Layer 2 Lateral GW-Flow | 1701 | 1405 | 296 | 16.3 | 18.3 | 27725 | 25710 | 43.76 | 36.15 | 7.62 |
| Layer 3 Lateral GW-Flow | 4886 | 15789 | -10903 | 16.3 | 18.3 | 79642 | 288939 | 125.72 | 406.25 | -280.53 |
| Flux Analyzer Deviation | 3077 | | 3077 | 16.3 | | 50147 | 0 | 79.16 | 0.00 | 79.16 |
| Vertical Flow | | 14940 | -14940 | | 18.3 | 0 | 273402 | 0.00 | 384.40 | -384.40 |
| GW-Recharge | 10019 | | 10019 | 16.3 | | 163316 | 0 | 257.80 | 0.00 | 257.80 |
| Implemented Sewers | 28 | | 28 | 71.41 | | 1999 | 0 | 0.72 | 0.00 | 0.72 |
| Additional Sources | 3114 | | **3114** | **71.41** | | **222371** | | **80.12** | **0.00** | **80.12** |
| Total | 33190.82 | 33190.82 | 0.00 | | | 607396 | 607392 | 853.99 | 853.99 | 0.00 |

Mass balance approaches like the procedure outlined above are frequently applied in the literature. For example, Rueedi *et al.* (2005) applied a slightly different methodology at the city of Doncaster (UK) and calculated a total recharge from sewage and stormwater of 20-50 mm/a (5-12 % of the total urban drainage) using a combination of sodium, potassium, boron, calcium and magnesium. However, the validity of the mass balance approach is limited by the often imprecise knowledge of the concentration of lateral inflowing water volumes and the uncertain definition of an average over the balance area.

In conclusion it can be stated that the mass balance approach points to significant additional sources for boron and chloride in the urban area. However the result is strongly influenced by the employed interpolation techniques and the low density of the observation well network compared to the pronounced local concentration gradients. The results of the mass balance approach are in contradiction to the results from the numerical groundwater transport models which predicts that sewage input of 28 m³/d is sufficient to reproduce the measured concentrations of boron and chloride in groundwater.

The major part of this discrepancy originates from the consideration of the entire water volume inside the Upper Gravel Layer for the water balance calculations, whereas the water samples were usually taken in depths of 1-2 m below the water table from mostly very shallow observation wells. Furthermore, the hydrochemical datasets in the urban area are very heterogeneous and, considering the steep vertical gradients, the interpolation mechanism overestimates the contaminated mass in the total water volume.

# 7 Comparative overview of leakage rate estimations

A considerable number of publications on exfiltration rates are available and estimates were produced using different approaches and methodologies. The difficulties in quantifying sewer leakage, as also outlined in the previous chapters, have compelled most authors to postulated worst and best case scenarios. However, many results are difficult to compare at first sigth as they relate to different forms of sewer leakage. Laboratory studies generally report leakage rates connected to a certain leak geometry, with variable boundary conditions of water level (Forschergruppe Kanalleckagen, 2002; Rott & Zacher, 1999; Vollertsen & Hvitved-Jacobsen, 2003). These can be transformed to an exfiltration rate expressed as volume per time per defect area. Other authors already provide own estimates for upscaling and report values as volume per time per length of the sewer (Blackwood *et al.*, 2005a; Dohmann *et al.*, 1999; Härig & Mull, 1992; Trauth, 1995). This format along with the specification as percentage of dry weather flow is also used in some studies using tracer techniques (Rieckermann *et al.*, 2005). Studies focused on the impact to the groundwater often express the groundwater recharge from sewer leakage as volume per year per surface area of the aquifer (Wolf *et al.*, 2006; Wolf & Hötzl, in print; Yang *et al.*, 1999).

Tab. 7-1: Characteristics of the sewer network in Rastatt used for comparison of leakage estimates.

| Parameter | Value | Unit |
|---|---|---|
| Catchment area | 10420704 | [m²] |
| Length of network | 164736 | [m] |
| Number of leaks | 5295 | [-] |
| Average leak size | 0.0093 | [m²] |
| Total leak size | 49.23 | [m²] |
| Leak frequency | 30.11 | [m/Leak] |
| DWF | 13790 | m³/d |

In order to compare the leakage rates reported in the different sources, all estimates were applied to the characteristics of the sewer network in Rastatt as listed in Tab. 7-1. The application of the estimates to the network in Rastatt is very straightforward for estimates citing a leakage rate per km network. For studies citing leakage rates as volume per leak area per time the application is more uncertain as no provision is made for the variable fill level in the sewer.

This direct application of leakage rates determined for different cities with different assumptions on the sewer condition, network density and rainfall patterns is problematic. The overview provided in Tab. 7-2 does not question the validity of the estimations documented in the literature. Nevertheless, it serves as a good basis to discuss the relevance and transferability of different procedures.

The lowest estimates of less than 2 mm/a sewer leakage derive mainly from laboratory studies with comparatively constant flow conditions (Rott & Zacher, 1999) and comparatively intact colmation layers (Forschergruppe Kanalleckagen, 2002), or defects just arranged in the sewer bottom (Vollertsen & Hvitved-Jacobsen, 2003). All of these studies were conducted over a comparatively long time period of at least several days or weeks.

A rather detailed forward modelling exercise was performed within the AISUWRS project (Wolf et al., in print) for four case study cities using the Network Infiltration and Exfiltration Model, NEIMO (DeSilva *et al.*, in print). For the city of Rastatt exfiltration rates between 0.8 mm/a and 5 mm/a were calculated, compared to 3.6 mm/a for MtGambier (Cook et al, in print), 21 mm/a for Doncaster (Morris *et al.*, in print) and 59 mm/a for Ljubljana (Souvent *et al.*, in print). The employed model approach calculates the water level in the sewer based on precipitation data. The main drawback is the daily time step which effectively prevents high water levels in the sewer (e.g. a typical storm event of 20 min duration is dispersed over a period of 24 hours). The NEIMO software possessed several fit parameters and the NEIMO modelling exercises were sometimes fit to the expectations of the model appliers. Only for the Rastatt case study, CCTV data were available. The application of the generic defect functions supplied with the NEIMO model resulted in approximately five times higher leakage rates. Nevertheless, NEIMO is currently the most advanced and detailed modelling tool available and further research on the precise parametrisation of this tool including a rigorous assessment of uncertainty is recommended.

Tab. 7-2: Application of different leakage estimates to the network in Rastatt. Values extracted from the respective literature source are displayed in bold.

| Source | Exfiltration rate | | | | | |
|---|---|---|---|---|---|---|
| | related to the city area | related to sewer length | | | related to dwf | related to defect area |
| | mm/a | m³/a/km | l/d/m | l/s/km | % | l/d/cm² |
| Rott & Zacher (1999), Minumum | 0.2 | 13.1 | 0.036 | 0.000 | 0.04 | **0.012** |
| Vollertsen and Hvitved-Jacobsen (2003), Min. | 0.3 | 21.8 | 0.060 | 0.001 | 0.07 | **0.02** |
| Klinger et al (2006), CCTV data | **0.8** | 50.6 | 0.139 | 0.002 | 0.17 | 0.046 |
| Wolf (2006) MC Mean | **0.99** | 62.6 | 0.172 | 0.002 | 0.20 | 0.057 |
| Vollertsen and Hvitved-Jacobsen (2003), Max. | 1.0 | 65.4 | 0.179 | 0.002 | 0.21 | **0.060** |
| Dohmann (1999), Minimum | 1.8 | **111.1** | 0.3044 | 0.0035 | 0.36 | 0.102 |
| Wolf (2006): MC 95 % Cumulative probability | **2.24** | 141.7 | 0.388 | 0.004 | 0.46 | 0.130 |
| Cook et al (2006), baseline | **3.6** | 227.7 | 0.624 | 0.007 | 0.75 | 0.209 |
| Fenz et al.(2004) | 4.8 | 305.5 | 0.837 | 0.010 | **1.00** | 0.280 |
| Yang et al. (1999), Minimum | **5** | 316.3 | 0.867 | 0.010 | 1.04 | 0.290 |
| Klinger et al (2006), Generic defects | **5.1** | 321.3 | 0.880 | 0.010 | 1.05 | 0.295 |
| Forschergruppe Karlsruhe (after 100 days) | 6.2 | 389.5 | 1.07 | 0.012 | 1.27 | **0.357** |
| Wolf et al.(2005) AISUWRS D20 | **7.4** | 468.1 | 1.28 | 0.015 | 1.53 | 0.429 |
| Trauth et al.(1995), Minimum | 8.7 | 547.5 | **1.50** | 0.0174 | 1.79 | 0.502 |
| Wolf (2006): MC Max | **10.38** | 656.6 | 1.80 | 0.021 | 2.15 | 0.602 |
| Rott & Zacher (1999), Maximum | 11.7 | 741.7 | 2.03 | 0.024 | 2.43 | **0.680** |
| Karpf and Krebs (2004b) | 13.5 | 855.5 | 2.34 | 0.027 | **2.80** | 0.784 |
| Rauch and Stegner (1994), Minimum | 15.0 | 949.0 | 2.60 | 0.030 | 3.11 | **0.87** |
| Yang et al.(1999), Maximum | **20** | 1265.1 | 3.47 | 0.040 | 4.14 | 1.16 |
| Morris et al (2006), baseline | **21** | 1328.4 | 3.64 | 0.042 | 4.35 | 1.218 |
| Dohmann (1999), Maximum | 25.1 | **1587.4** | 4.35 | 0.0503 | 5.20 | 1.46 |
| Ullmann (1994), Minimum | 27.6 | 1745.2 | 4.78 | 0.055 | 5.71 | **1.60** |
| Rieckermann et al.(2005), Minimum | 43.5 | 2749.8 | 7.53 | 0.087 | **9.00** | 2.52 |
| Härig (1991), Minimum | 49.9 | 3153.6 | **8.64** | 0.1000 | **10.32** | 2.89 |
| Wolf (2006):Mass balance boron | **53** | 3352.6 | 9.19 | 0.106 | 10.97 | 3.074 |
| Souvent et al (2006), baseline | **59** | 3732.2 | 10.2 | 0.118 | 12.22 | 3.422 |
| Rieckermann et al.(2005), Maximum | 62.8 | 3971.9 | 10.9 | 0.126 | **13.00** | 3.64 |
| EPA ( 2000), Minimum | 82.1 | 5194.0 | 14.2 | 0.165 | **17.00** | 4.76 |
| Rauch and Stegner (1994), Maximum | 89.7 | 5672.0 | 15.5 | 0.180 | 18.56 | **5.20** |
| Härig (1991), Maximum | 219.3 | 13870.0 | **38.0** | 0.4398 | 45.40 | 12.72 |
| Trauth et al.(1995), Maximum | 225.0 | 14235.0 | **39.0** | 0.4514 | 46.59 | 13.05 |
| EPA (2000), Maximum | 270.5 | 17109.8 | 46.9 | 0.543 | **56.00** | 15.69 |
| Rieckermann et al (2003) | 436.2 | 27594.0 | 75.6 | **0.875** | 90.31 | 25.30 |
| Blackwood et al (2005) | 997.1 | 63072.0 | 172.8 | **2.000** | 206.43 | 57.82 |
| Ullmann (1994), Maximum | 1000.1 | 63264.8 | 173.3 | 2.006 | 207.06 | **58.00** |

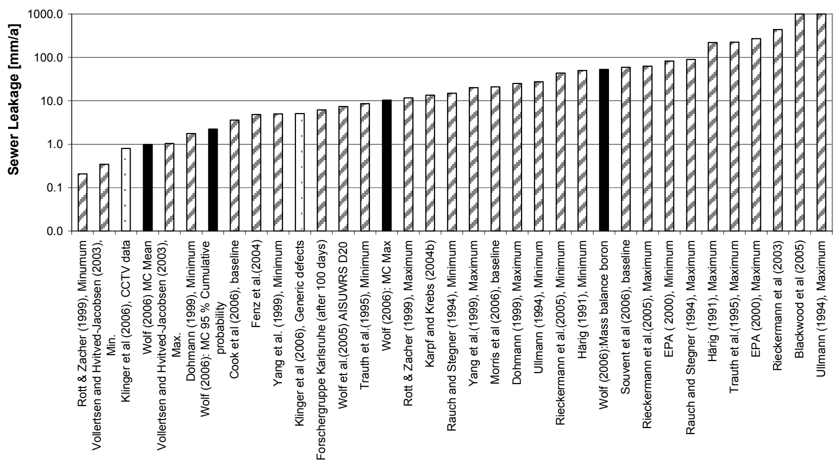



Fig. 7-1: Comparison of leakage rates applied to the city of Rastatt expressed as percentage of dry weather flow. Estimates produced within this thesis are marked black. Estimates obtained within the AISUWRS project appear as dotted bars.

The estimate of 0.99 mm/a sewer leakage from public sewers as the mean from the Monte Carlo analysis ranges at the lower boundary of the existing literature. The overall credibility of these values is quite high as all relevant processes are incorporated into the calculations, the variation and uncertainty of input parameters was addressed and the results of different approaches were compared. It compares also well with the estimates from the NEIMO model (0.8 mm/a). The simplified mass balance approach demonstrated in Rastatt results in an exfiltration rate of 53 mm/a which is believed to be an over exaggeration of the problem and not in accordance with the numerical flow and transport model results. Nevertheless the predicted exfiltration from the mass balance is well within the range of the concurrent literature estimates.

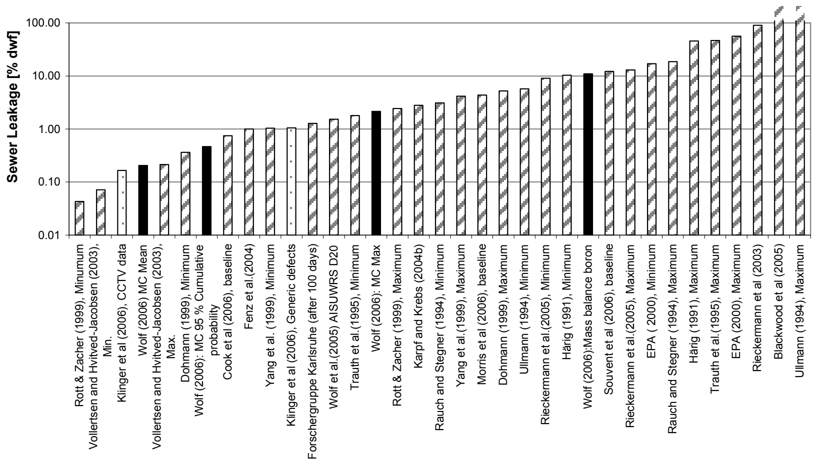



Fig. 7-2: Comparison of leakage rates applied to the city of Rastatt expressed as percentage of dry weather flow. Estimates produced within this thesis are marked black. Estimates obtained within the AISUWRS project appear as dotted bars.

Very high sewer leakage rates, which result in not plausible numbers if upscaled to the city area of Rastatt were reported in studies using pressurized packer tests (Ullmann, 1994) or short term laboratory experiments (Blackwood *et al.*, 2005b).

# 8 Summary and Outlook

## 8.1 Summary

Leakage from defect sewer systems is an omnipresent phenomenon in urban areas but its quantification is associated with high uncertainty. Several studies have identified leaky sewer systems as the dominating contaminant input to the urban aquifer (Eiswirth & Hötzl, 1997; Härig & Mull, 1992). On the other hand, contemporary literature also reports the dominating influence of the self-sealing effect which leads to a low risk potential. Furthermore, attempts to directly measure wastewater impact on groundwater quality have been unsuccessful in several experiments in Germany and the results from most other case studies worldwide are ambiguous. It has been the aim of this study to merge recent laboratory observations with direct field investigations and to prove the impacts of wastewater exfiltration on the groundwater in the sparsely protected sand and gravel aquifer beneath the city of Rastatt, Germany.

In the area of Rastatt, the Upper Gravel Layer is the aquifer which serves as the main drinking water resource. The vulnerability of this aquifer towards contamination was assessed by a reconnaissance mapping of the urban geology of Rastatt. This study underlined that the vast majority of sewers is in direct contact with the aquifer. Low permeable cover layers, corresponding to silty overbank facies, are only locally important.

A survey of historical recordings of groundwater levels showed that urbanisation including leakage from defect sewer systems has an almost negligible impact on the groundwater level around Rastatt. Even the large scale changes in terms of surface sealing, increased groundwater abstractions and changed courses of rivers did not provoke major changes in groundwater level. Since 1913 the groundwater levels in Rastatt have dropped by approx. 1 m on average, with a natural oscillation of +/- 1 m.

Strong hydrochemical evidence for exfiltration from leaky sewers was found within the experimental field investigations. The placement of focus observation wells in the direct vicinity of defect sewers with help of the geographically referenced sewer defect database provided opportunities for direct observation of sewage-influenced groundwater. The observation wells were divided into four groups according to their distance to the next sewer. It can be seen that, compared to the average of the other three well groups, the focus observation wells (Well Group 3) exhibit significantly elevated values for the parameters electrical conductivity, temperature, sodium, chloride, potassium, phosphate, ammonium and boron. The highest contrast can be observed for ammonium (+ 446.5 % in the arithmetic mean) potassium (+ 116 % in the arithmetic mean), sodium (+ 80.5 % in the arithmetic mean), chloride (+ 77.3 % in the arithmetic mean) and boron (+ 65.2 % in the arithmetic mean). An inverse relationship was found for dissolved oxygen and nitrate, which indicates active oxidation of ammonium. The marked contrast between the sewer focus group and the urban background group shows that the obvious quality deterioration is limited to the immediate surrounding of defects rather than the whole network. However, the elevated concentrations of the urban background group compared to the rural group also demonstrate the diffuse pollution originating from the superposition of a large number of leaks.

Online probes in the focus observation wells which monitored physico-chemical characteristics detected fast responses in the specific electrical conductivity of groundwater in the course of rain events. This proved that the sewers are in close contact with the aquifer and travel times through the unsaturated zone are short (< 1 day). Large volumes of wastewater exfiltrate during storm events but result in a drop in electrical conductivity due to the diluting effect of the rain water on the wastewater chemistry.

The microbiological sampling programme comprised *Escherichia coli*, total coliforms, enterococci, faecal streptococci, sulphate reducing clostridia, *Clostridia perfringens*, coliphage and *Pseudomonas aeroginosa*. Indicators of faecal contamination were found in several groundwater samples. Groundwater from 6 out of 12 wells is not suitable for human consumption according to German and international drinking water guidelines. However, no correlation between leak geometry, distance to the leak and microbiological indicators was found. In addition, the correlation between other wastewater indicators (e.g. ammonium, boron, pharmaceutics) and microbiological indicators was very weak. The findings of *Escherichia coli* in well D1 support the idea of a defect house branch connection as an additional sewage source. It must be stated that wastewater in Rastatt contains high numbers of all the organisms it was screened for, but a removal of several magnitudes takes place within the sediments surrounding the sewer. Nevertheless, some of the bacteria and viruses apparently pass the natural barrier and continue to pose a threat to human health.

The screening for pharmaceutical residues in groundwater and sewage samples revealed the existence of betablockers (Metoprolol, Sotalol) in the urban groundwater. Compared to the state-wide survey undertaken by the LfU (2002) pharmaceutics were found with elevated frequency in the Rastatt groundwater samples. These results correspond to other parameters indicating sewer leakage. Only limited degradation or attenuation of most pharmaceutics occurs on the seepage route. All substances identified in the sewage were also recorded in the seepage water in a similar concentration range.

The pharmaceutical group of iodated x-ray contrast media received special attention in the sampling programme. 114 samples from 46 wells were analysed. Positive detects occurred for amidotrizoic acid (30), iothalaminic acid (10), ioxithalaminic acid (10), iopamidol (4), iohexol (4), iomeprol (3), iopromide (2). Highest concentrations were observed for amidotrizoic acid (1703 ng/l), iomeprol (1655 ng/l) and iothalamic acid (238 ng/l). The frequency of the positive detects as well as the maximum concentrations exceed the values from the state-wide sampling campaign in Baden-Württemberg (LfU 2002). No x-ray contrast media were found in wells which are within a distance of more than 60 m to the nearest upstream sewer. The occurrence of the iodated x-ray contrast media clearly proves the existence of major amounts of sewage in the urban groundwater. Due to their chemical robustness, iodated x-ray contrast media are well suited as a marker species. However, they are irregularly and selectively distributed over the city area.

A comparison of the average concentrations of EDTA in the different well groups shows that the wells outside or at the fringe of the urban area (focus groups 0 and 1) exhibit a mean concentration of 0.67 µg/l EDTA. For the urban background (focus group 2) an average of 1.2 µg/l EDTA was measured. The sewer focus wells (focus group 3) exhibited an average of 2.14 µg/l EDTA.

The measurements performed at lab-scale and test-sites in Karlsruhe and Rastatt within the research group on sewer leakage were reviewed and compared with literature results. For this purpose it was necessary to apply the Darcy equation to the leakage process with slight modifications of the already documented applications. The resulting hydraulic conductivities of the colmation layer span from $3.2 \times 10^{-8}$ m/s to $6.2 \times 10^{-5}$ m/s. Despite the common application of the Darcy equation to the leakage problem, most documented experiments do not show a clear correlation between defect size and exfiltration rate. Other factors like leak geometry, preferential flow at the crack walls, turbulent water flow and highly variable shear stress at the interface between sewage flow and the leak exert a major influence on the exfiltration rate. However, in the absence of a reasonable alternative to the assumption of a flow proportional to the wet defect area, the Darcy equation was also applied for the forward modelling exercises in this thesis. Future research should lead to a set of correction factors to account for the different leak geometries and hydraulic stress.

More than 95 % of the public sewer system were inspected by using CCTV cameras by the municipality of Rastatt. This condition assessment was given spatial reference and managed within a sewer defect database linked to a GIS system. The available CCTV information was used to estimate the surface area of leaks within the Rastatt sewer network. Within the 162 km long network, 5295 defects were considered for the calculation of leak areas. This corresponded to a total leak area of 49.23 m² and a leak frequency of one major leak per every 30 m of sewer.

A new method was described for the upscaling of the laboratory results to calculate the amount of sewage exfiltration for the entire sewer network of Rastatt. Besides the defect area known from the visual inspection, the proposed equation takes into account the different exfiltration characteristics during dry weather flow and extreme rain events. Furthermore it considers the water level in the sewer, the wetted proportion of the leak area, the thickness of the colmation layer, the percentage of sewers beneath the groundwater table and two different hydraulic conductivities of the colmation layer during dry weather and during storm water flow. It assumes that all pipes are embedded in sandy material.

The method relies on a number of input parameters and hypotheses which contain significant uncertainties. In order to evaluate the influence of the uncertainties of the input data on the calculation of leakage rates, a Monte Carlo analysis was performed. 50000 runs were performed with each input variable randomly chosen within the specified boundaries and probability density functions. The forecast returned a mean leakage rate of 0.99 mm/a related to the entire city area. The forecast is characterised by a rather high standard deviation of 0.66 mm/a and a total variance of 0.43 mm/a. The minimum is at 0.02 mm/a and the maximum at 10.38 mm/a. With a probability of 95 % the leakage rate will be below 2.24 mm/a. For the mean, this equals leakage rates of 5.34 l/d at a single leak which corresponds reasonably well with the measurements at the test site Kehler Strasse where an average rate of ca. 8 l/d was measured for the entire test duration and approx. 3 l/d after stabilisation of the exfiltrating conditions.

Besides the exfiltration volumes the load of boron to the soil-aquifer system from sewer leakage was investigated. The results reflect a similar pattern as the exfiltration volumes but are not linear dependent on them. The specification of low boron concentrations during periods of high exfiltration rates gives more weight to the exfiltration process during dry weather flow. The forecast returned a mean boron load of 0.45 mg/m²/a related to the entire city area. With a probability of 95 % the boron load will be below 1.1 mg/m²/a. To demonstrate the flexibility of the chosen approach, chloride was also calculated as a second marker species. The forecast returned a mean chloride load of 71.14 mg/m²/a related to the entire city area. With a probability of 95 % the boron load will be below 186 mg/m²/a.

A sensitivity analysis was performed for each of the input parameters. The variance in the predicted exfiltration volumes is mainly caused by the hydraulic conductivity of the colmation layer during storm events (31 % contribution to the variance). In order to achieve a more reliable estimation of the exfiltrating sewage volumes, it is therefore of prime importance to investigate the hydraulic conductivities of the colmation layer, especially with regard to the storm events in the sewer. This directly refers to a lack of measurements of the thickness of the colmation layer in different sectors of the pipe. Regarding the solute load to the soil-aquifer system the most sensitive parameter is the hydraulic conductivity of the colmation layer during dry weather flow (44 % contribution to the variance).

Finally, a numerical groundwater flow and transport model was set up to allow for a comparison between measured concentrations and concentrations derived from the forward modelling using the described Monte Carlo simulation approach. The 3-dimensional steady state groundwater model describes the Upper and the Middle Gravel layer, separated by the Upper Interhorizon. It utilizes groundwater recharge data obtained with the AISUWRS modelling suite for the urban area of Rastatt and was calibrated by changing the hydraulic conductivities of the sediments. The 5295 sewer leaks known from the CCTV inspection were summarized to form only one source per asset and subsequently implemented into the model via 1218 injecting wells, using a fixed recharge concentration.

The modelled distribution of boron in the aquifer shows that concentrations significantly exceeding the background concentration of 0.038 mg/l are only visible in the direct vicinity of sewer leaks. Both groundwater recharge and lateral flow exert a strong influence in the concentration distribution and effectively dilute the injected sewage within tens of meters from the leak. The modelled boron and chloride concentrations were compared with the measured concentrations at 46 wells in the urban area of Rastatt. The measured average boron concentration at the wells (0.064 mg/l) compares well with those modelled 2 m below the water table (0.073 mg/l) but is too low if a water depth of 10 m is considered. With regard to chloride, the measured average concentration of the 46 wells exceeds the modelled concentration in all depths. The correlation between measured and modelled concentrations at individual wells is weak. Overall, the numerical groundwater model supports the findings from the sampling campaigns which detected elevated concentrations only within a 10-50 m distance to a leak. The model results point towards a higher total leakage than the sewer leakage rate of 1 mm/a predicted by the Monte Carlo analysis for the public sewers. Pronounced vertical concentration gradients are predicted by the model.

A simplified mass balance approach which treats the groundwater beneath the city area as a black box model was employed using the flow quantities from the numerical groundwater model. In order to balance input and output of boron for the balance area, an additional input of 2036 m³/d sewage (equivalent to 52.38 mm/a) with a concentration of 0.45 mg/l would be necessary. If leaky sewers are regarded as the only source for boron in the urban area, a leakage rate of 53 mm/a needs to be assumed. A similar exercise was performed for chloride. In order to balance input and output of chloride for the balance area, an additional input of 3114 m³/d sewage (equivalent to 80.12 mm/a) with a concentration of 71.41 mg/l would be necessary. A part of this input originates from defective house connections. However, the majority of additional chloride input will result from road salting. In addition there is a general trend of increasing chloride concentrations towards the river Rhine (Eiswirth, 2002). Chloride is therefore only suitable to constrain and control mass balances set up for other marker species. In order to determine the amount of sewer leakage via the mass balance approach, a quantification of the mass flux from road salting and other sources is necessary.

At last, the findings of the Rastatt case study were compared with statements of 24 international literature sources on sewer leakage rates. For this comparison the characteristics of the Rastatt sewer network were applied to the different literature statements. Related to the sewer network in Rastatt, the estimates would range from 0.2 mm/a to 1001 mm/a sewer leakage from public sewer networks. The majority of the estimates is between 1 mm/a and 50 mm/a. The predictions produced within this thesis range from 1 mm/a to 10 mm/a for the emissions from the public sewer network and a maximum of 53 mm/a for the combined leakage from private and public sewers.

## 8.2 Outlook

As a summary, it can be stated that consistent evidence was found that significant amounts of wastewater are released to the groundwater beneath the city of Rastatt. The self-sealing effect observed at the laboratory scale does not prevent a qualitative deterioration of the urban groundwater due to sewer leakage. The amount of sewer leakage is small compared to the overall urban water budget and impacts on the groundwater are only locally observable. Based on the substances screened for in this study, an immediate health risk would only be posed by microbiological contamination (*E.Coli*, SRC, Enterococci) where untreated groundwater from private wells in the city area is used. However, this situation can change suddenly if toxic substances are introduced into the sewer system through chemical spillages or deliberate misuse of the drainage systems. Furthermore, the effects of the constant release of pharmaceutics and human microbiology into the environment might foster the origin of resistant bacteria and viruses. Last but not least, it must be kept in mind that wastewater contains a very broad spectrum of substances and this study only screened for a small selection of them.

The described upscaling procedure combined with a Monte Carlo analysis provides a transferable methodology for the assessment of sewer leakage at the city scale. It is therefore an alternative method to the recently developed NEIMO software, which requires the complex and time-consuming specification of the flow directions, wastewater inputs from households and the calculation of urban surface runoff on a daily basis. In addition the proposed Monte Carlo methodology is capable of addressing the uncertainty introduced by the incomplete knowledge of the input parameters.

For future studies on sewer exfiltration, the confinement of obvious sewage impact to the close surrounding of a defect and the pronounced variation of marker species both in space and time need to be taken into account. The inclusion of private sewer networks into future assessments is highly recommended. Within urban areas, very dense monitoring networks are required. For a risk assessment, the elevated concentrations in the close surrounding of leaks must be considered rather than concentrations at observation wells downstream of the urban area. A good knowledge of groundwater users in the city, which might be both formal and informal, is required to assess the spatial separation of contamination sources and water withdrawals.

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