Assessing Infiltration and Exfiltration on the Performance of Urban Sewer Systems

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1. INTRODUCTION

Urban sewer systems constitute a very significant patrimony in European cities. Their structural quality and functional efficiency are key parameters to guarantee the transfer of domestic and trade wastewater to treatment plants without infiltration nor exfiltration. Infiltration of groundwater is particularly detrimental to treatment plant efficiency (hydraulic overloading due to the infiltrated volume of water which can reach up to 100 % of the wastewater volume in some cities, dilution of pollutant concentrations leads to a lower pollutant removal efficiency), while exfiltration of wastewater can lead to groundwater contamination (especially where groundwater is a water resource for drinking water production). Both problems are critical on a long term basis for sustainable urban water management and have important economic consequences for cities and sewer systems operators through the European Union. The European standard EN 752-2 (CEN, 1996) indicates basic performance criteria applicable to any sewer system. Among these criteria, the two following ones are especially relevant: i) receiving waters should be protected against pollution; ii) the structural integrity of urban sewer systems, including water tightness, should be guaranteed. In order to evaluate the performance of urban sewer systems, public and private operators need appropriate methods and techniques.

During the period 2001-2004, the APUSS project (Assessing infiltration and exfiltration on the Performance of Urban Sewer Systems), associating universities, SMEs and municipalities in seven European countries and financed by the European Commission under the 5th R&D Framework Programme, was devoted to sewer infiltration and exfiltration questions. Four main work areas were established: WA1) development of new measurement methods, WA2) tests and field applications of the new methods under various contexts, WA3) implementation of associated models and tools, and WA4) socio-economic aspects linked to infiltration and exfiltration (I/E). After a brief introduction of the APUSS partners, the following chapters describe synthetically the objectives, methods and main results obtained in each work area. It is of course not possible to give all scientific and technical details in the limited frame of this final report: references are given to APUSS reports, documents, and scientific and technical papers for readers who are interested in detailed information. All documents are publicly available on the APUSS website at http://www.insa-lyon.fr/Laboratoires/URGC-HU/apuss.

Ten scientific partners from seven European countries have defined and carried out the APUSS project. Some of them were associated to end-users (sewer operators) interested in I/E, who contributed by providing assistance, access to experimental sites, operational data, and additional funding (Table 1). The APUSS project has also been part of the 5th R&D Framework Programme European cluster CityNet including six individual projects dealing with integrated urban water systems (Schilling et al., 2002).

Table 1: APUSS scientific partners, their main activities and associated end-users.

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<th>Scientific partner</th>
<th>Main activities</th>
<th>Associated end-user</th>
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</thead>
<tbody>
<tr>
<td>INSA de Lyon (France, co-ordinator)</td>
<td>WA2, WA4</td>
<td>Greater Lyon</td>
</tr>
<tr>
<td>EAWAG (Switzerland)</td>
<td>WA1</td>
<td>city of Zürich</td>
</tr>
<tr>
<td>Dresden University of Technology (Germany)</td>
<td>WA2, WA3</td>
<td>cities of Dresden and Berlin</td>
</tr>
<tr>
<td>Czech Technical University in Prague (Czech Republic)</td>
<td>WA1</td>
<td>city of Prague</td>
</tr>
<tr>
<td>DHI Hydroinform a.s. (Czech Republic)</td>
<td>WA2, WA3</td>
<td>city of Prague</td>
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<td>Hydroyproekt a.s. (Czech Republic)</td>
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<tr>
<td>Middlesex University (United Kingdom)</td>
<td>WA1, WA2</td>
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<td>National Laboratory for Civil Engineering (Portugal)</td>
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<tr>
<td>Emschergenossenschaft (Germany)</td>
<td>WA2, WA4</td>
<td>cities of Bottrop and Gladbeck</td>
</tr>
<tr>
<td>IRSA-CNR (Italy)</td>
<td>WA2</td>
<td>city of Rome</td>
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2. CONTEXT AND OBJECTIVES

Modern sewer systems have been developed in European large cities since the middle of the 19th century, in order to protect public health, safety and quality of life in cities. This hygienist point of view was the starting point of all further efforts and investments. The collection of wastewater was a solution to avoid pollution of groundwater and of surface water resources. Nevertheless, there were negative impacts on downstream receiving waters. But these effects were usually not appropriately considered during the first decades. Today in Europe, sewer systems still contribute to public health. Dry weather effluents are normally treated in wastewater treatment plants (WWTP) and discharges into receiving waters through overflow structures during storm events should be limited. The urban water cycle and the sustainable integrated management of water are the new paradigms.

However, sewer systems are not fully satisfactory. Because of i) age of pipes (some parts of sewer systems in European cities are more than 100 years old), ii) bad construction quality (poor quality of pipe material, bad laying conditions, ignorance or underestimation of the effects of geotechnical and road traffic conditions…), iii) lack of or insufficient maintenance, iv) lack of appropriate investment and rehabilitation strategies, and v) high costs of construction and rehabilitation, many sewer systems across Europe are not watertight and suffer from significant infiltration and/or exfiltration. Where infiltration and exfiltration are observed, rehabilitation is needed. The investments to build or, more frequently, to renew, rehabilitate and upgrade urban sewer systems need substantial resources.

Water Authorities try to recognize this problem. However, a call for comprehensive knowledge and economic pricing generally cannot be answered because of a lack of knowledge. As an example, it is estimated that up to 40 % of UK sewers have structural defects and about 1 in 12 (of the 302 000 km of public sewers) have serious defects with some €11.6 billion needed to alleviate the most immediate problems. Very little attention has been given to potential exfiltration losses from sewers in contrast to infiltration and only some 70 officially documented cases of sewer leakage exist over the past 60 years or so. However, some 5000 sewer collapses are recorded per annum and there is widespread anecdotal evidence that suggests that sewer exfiltration may be more important than previously assumed. The UK water companies have requested the UK regulator (OFWAT) for €12.3 billion in the current Asset Management Plan (AMP5) review of the industry in order to redress a historic neglect of sewers. It is also clear from the EU Water Framework Directive (WFD) that such diffuse sources of pollution to groundwater will be targeted for much stricter control in the future and that there will be a need to identify and quantify potential losses in Catchment Management Plans prior to the introduction of any Programme of Measures as required by the Directive.

Moreover, between 5000 to 7000 UK properties (0.1 %) are flooded each year by sewage although there may be three to ten times as many external sewer flooding incidents (gardens, streets, etc) which are unaudited (National Audit Office, 2004). In 2002-2003, the UK water companies invested over €940 million on the sewage network of which €275 million was spent on sewer network maintenance, including sewer replacement. However, very little information exists on the rates of deterioration of the UK sewer network which are needed for a robust cost-benefit analysis to inform decisions on sewer flooding policy. The effects of climate change have already increased the incidence of sewer flooding and considerably raised the profile of infiltration contributions to such flooding. The increased risks from future climate change and extreme events are exacerbated by new housing and legislative demands (e.g. the EU WFD) which will also impact upon future sewer performance. In addition, the WFD requires a very different holistic, integrated approach to land and water management which demands that the interfaces of groundwater, surface water and wastewater flows be considered and managed interactively. However, the cost of dealing even with the most severe internal flooding problems (associated with the 32 000 km or 10 % of sewer grades 4 and 5) are estimated at €1.6 billion with the worst external cases costing a further €0.7 billion.

In France, no national statistics are available, but since 20 years the six French Water Agencies have promoted diagnostic studies of sewer systems, one of the most important objectives being the detection and the quantification of infiltration. Following diagnostics, large investments are devoted to sewer rehabilitation. As an example, approximately €25 billions have been spent in the EU up to now to meet the requirements of the Urban Waste Water Directive 91/271. As it is not conceivable to increase the wastewater treatment efficiency without increasing the efficiency of wastewater collection, i.e. the quality of sewer networks, considerable resources will be needed in order to reach the objectives of the European Directives.
But public parts of sewer systems are only one aspect of the problem. House connections are often the most neglected components of the whole sewer system and may contribute to as much as 50% of total infiltration of groundwater. Exfiltration from leaky house connections can be substantial and of dramatic importance in terms of protecting groundwater against pollution as well as preserving foundations of buildings. Rehabilitation work aimed at achieving a decrease in water infiltrated from surrounding subsoil concentrates mainly on the core sewers thus leaving the house connections neglected in the majority of cases.

Infiltration and exfiltration appear as crucial issues. However, information about infiltration and exfiltration from sewer systems is relatively poor and investments are frequently based on weak information, limited data sets with rather high uncertainties. Especially the assessment of water-tightness under operational conditions is in practice regularly derived by approximated assumptions rather than estimated through validated methods. Consequently, there is commonly little or no policy awareness of treating this issue until the economic or social consequences have become impossible to ignore. It is clear from the EU WFD as well as from current developments of national water regulations that deficiencies of sewers will not be longer tolerated in the future. A better knowledge of infiltration and exfiltration rates is then absolutely necessary. Such a knowledge will contribute to evaluate performance indicators which are more and more frequently used to assess the structural and functional states of sewer systems. New maintenance and rehabilitation strategies will also be (at least partly) based on these performance indicators.

In this context, the APUSS project had the following main objectives:

- to elaborate a set of new measurement methods to directly estimate exfiltration and infiltration rates in sewers by means of tracers: at subcatchment scale for infiltration, at sewer reach scale for exfiltration, with standard operation protocols including detailed uncertainty analysis and data processing in order to obtain accurate I/E rates.
- to develop and test volumetric measurement methods for house connections, with a matrix approach to extrapolate measurements on some house connections to wider catchments.
- to test and validate the above new measurement methods in various contexts with associated end-users, in order to contribute to their improvement and to explore their advantages, possible difficulties and conditions of use or application.
- to develop and validate modelling approaches at various time scales, including statistical methods to i) attempt to extrapolate experimental results to wider and/or similar unmonitored subcatchments and sewer reaches, and ii) to optimally choose the measurement locations.
- to provide end-users with a unified software tool, including experimental data storage, automatic model calibration, graphical display of results, and links with performance indicators.
- to provide the end-users with a fully documented large scale example of application of the methods and softwares.
- to define a set of specific performance indicators accounting for infiltration and exfiltration.
- to propose a multi-criteria approach to compare and rank investment strategies to solve I/E problems.
- to analyse the economic value of sewer systems and to evaluate the cost of application of the new methods.
- to make all above elements publicly available at the end of the project.
3. NEW MEASUREMENT METHODS

Traditionally, the assessment of I/E in sewer systems is typically based on rather inaccurate methods of flow measurement, analysis of diurnal flow and load variation and balancing of water in-and outputs (e.g. De Bénédictis and Bertrand-Krajewski, 2004; Rutsch et al., 2005). One of the key objectives of the APUSS project consisted to develop new measurement methods allowing differentiating sewer zones with infiltration and exfiltration, based on limited analytical effort and with little environmental risk. By means of an optimised design of the experiments and a detailed analysis of the resulting data and of their associated uncertainties, a higher degree of accuracy and a better quality knowledge than in traditional methods was expected.

Work area 1 included the development, testing and validation of methods to accurately quantify:
- exfiltration from sewer reaches (section 3.1)
- infiltration into sewer systems (section 3.2)
- effects and interactions with sewer solids and sediments (section 3.3)
- infiltration and exfiltration from house connections (section 3.4).

The methods are mainly based on selected chemicals and natural isotopic tracers and have been tested and validated under various operational conditions during dry weather, at different space scales (from the pipe reach to the whole catchment) and under different conditions (steady and dynamic groundwater levels, seasonal effects, etc.). They have been applied in different catchments chosen in the associated cities (see Chapter 4) and compared to traditional approaches based e.g. only on flow rate measurements or on large scale water mass balance. Standard operation protocols and routines for data processing and detailed uncertainty analysis have been specifically established.

3.1 EXFILTRATION METHODS

3.1.1 Concepts

The basic idea of the QUEST method (Quantification of Exfiltration from Sewers with artificial Tracers) is that exfiltration measurement is feasible by a mass balance of a tracer substance over the investigated sewer line (Rieckermann et al., 2005c). At the upstream extremity of the investigation reach, a known mass of tracer is dosed. The tracer is passed through the sewer line and if exfiltration occurs, tracer substance is lost with the seeping wastewater. From the remaining tracer mass at the end of the reach, exfiltration can be determined as:

\[ E = 1 - \frac{\text{mass}_\text{out}}{\text{mass}_\text{in}} \]  

where \( E \) = exfiltration, expressed as a proportion of wastewater discharge, and \( \text{mass}_\text{in} \) and \( \text{mass}_\text{out} \) = upstream and downstream mass of tracer, respectively.

Computation of the remaining tracer mass requires integration of the product of instantaneous tracer concentration and wastewater discharge. However, discharge measurement in sewers may not be very accurate, especially if portable flow meters are used (Bertrand-Krajewski et al., 2000). For this reason, we propose to dose one or more downstream reference signals in addition to the upstream indicator signal (Figure 1) to eliminate the need for accurate discharge measurement. Then, relative differences in the mass of tracer between the points of injection and the sampling point can be used to estimate exfiltration over the investigation reach according to:

\[ E = 1 - \frac{\text{mass}_{\text{REF},\text{in}}}{\text{mass}_{\text{IND},\text{in}}} - \frac{\text{mass}_{\text{IND},\text{out}}}{\text{mass}_{\text{REF},\text{out}}} = 1 - \frac{\text{mass}_{\text{REF},\text{in}}}{\text{mass}_{\text{IND},\text{in}}} \left( \frac{\int \text{Q}(t)\text{C}_{\text{IND}}(t)\,dt}{\int \text{Q}(t)\text{C}_{\text{REF}}(t)\,dt} \right) \]  

where \( \text{mass}_{\text{REF},\text{in}} \) and \( \text{mass}_{\text{IND},\text{in}} \) = introduced mass of reference and indicator tracer, \( \text{mass}_{\text{REF},\text{out}} \) and \( \text{mass}_{\text{IND},\text{out}} \) = downstream mass of reference and indicator tracer, \( \text{C}_{\text{REF}} \) and \( \text{C}_{\text{IND}} \) = downstream concentration of reference and indicator tracer, and \( \text{Q}(t) \) = downstream discharge.
The computed exfiltration ratio is also not affected by water losses downstream of the addition of the reference tracer (Figure 1), because it can be assumed that they affect both signals equally. When the discharge is steady during the experiment, only downstream concentration measurements are required, not discharge. In case that discharge is monitored, then the reference tracer can be used to correct for systematic errors in the flow measurements.

![Diagram](https://via.placeholder.com/150)

**Figure 1**: Conceptual sketch of the experimental set-up of the QUEST method (Rieckermann et al., 2005c).

Along this concept, two generic methods have been developed which differ in dosing strategy and applied tracers. The QUEST method uses instantaneous injections of a single tracer substance, whereas the QUEST-C uses two different tracers which are dosed continuously. Exemplary datasets, which illustrate the differences between the two approaches, are presented in Figure 2.

In the following paragraphs, a brief summary of both methods is presented. In-depth information on the specific topic is provided in the cited references and is also available on the APUSS website.

### 3.1.2 QUEST

For the QUEST method (Rieckermann et al., 2005c), the major achievements are:

- a novel experimental method to directly measure sewer leakage,
- a comprehensive framework to rigorously assess the accuracy of this method using reliable methods of uncertainty analysis.

Measurement campaigns have been performed mostly with NaCl and, to a lesser extent, fluorescent dyes (Uranin and Rhodamine WT). A basic requirement is in-line measuring technology with a measurement frequency of a few seconds. This is because the tracer signals resulting from slug injections are highly dynamic, and discrete sampling prohibits accurate monitoring of the tracer peaks.

#### Data analysis

To calculate exfiltration from the experiment, the tracer concentration is deconvoluted into three components: indicator signal, reference signal and baseline. For the identification of the three components, a model is applied that uses information on injected tracer masses and discharge (if available) along with parameterised functions for the tracer peaks and the baseline. The parameters of peak and baseline models, which are estimated by nonlinear regression, are used to compute the remaining tracer mass, from which exfiltration is calculated.

For the analysis of uncertainty, model parameter uncertainty, dosing errors, uncertainties in the calibration function and discharge measurements, as well as errors due to the unknown baseline are all considered explicitly. Bootstrap and regression techniques have been developed to assess the individual error contributions. Error propagation is performed by Monte Carlo Simulation.
Parameterisation of the tracer pulses with a known distribution has the advantage that an overlap of the different signals is feasible, which keeps the total time of passage of the pulses as short as possible. In this way, maximum information on the baseline is gained and the corresponding error is minimised. In order to obtain optimal results, an experimental design procedure has been developed, which is be discussed below.

3.1.3 QUEST-C

In the QUEST-C methodology (Rieckermann et al., 2005a), important scientific innovations are:
- an improved experimental design that reduces systematic errors,
- a methodology to perform an assessment of uncertainty for the obtained results,
- a new approach for data analysis that improves the conventional approach by accounting for dynamic flow.

The methodology, which is in principle independent of tracer substance, has been implemented mostly using Lithium (Lithium chloride) and Bromide (Sodium bromide). These substances proved to be conservative in most sewer applications. The advantage of the continuous dosing over the pulse dosing technique is that no inline measurement equipment is necessary. This avoids problems with the installation and operation of delicate sensors and devices in the sewer environment.

Data analysis

In the conventional approach for data analysis of continuous tracer experiments, the discharge is assumed to be steady during the experiment. This simplifies equation 2 considerably so that the data analysis can be performed in a spreadsheet calculation. Analysis of uncertainty considers seven major sources of uncertainty, the most relevant of which are: inaccuracy in the dosing rates, tracer measurement error in samples and standards, and the uncertainty of laboratory equipment.

It was further found that the separation of wave and fluid in the investigation reach might cause variations of the tracer ratio in the samples. It has to be considered that discharge fluctuations in time result in waves which travel at a higher speed than the main water body. For the QUEST-C method, this would mean that the two tracer substances that have been dosed to the same water element were diluted differently at the dosing point. The magnitude of this error fundamentally depends on the flow during the experiment and the transport characteristics of the investigation reach (length, roughness, slope, etc.). When the simplified model is applied, this effect is to some extent inherent in the captured data, but the overall magnitude cannot be assessed without additional discharge information. Uncertainty analysis of the conventional approach is implemented by Gaussian error propagation.

In the dynamic approach to data analysis, which requires considerably more information, discharge data is used to compute exfiltration ratios from the ratio of tracer loads instead of tracer concentration ratios. In this approach, additional errors to those mentioned above are introduced, including: errors in discharge measurement and integration error by different time resolution of flow monitoring and sampling. As the model for data analysis is non-linear and some errors are assumed to be uniformly distributed, the uncertainty analysis is performed by Monte Carlo Simulation.

3.1.4 Experimental Design

The QUEST and QUEST-C tracer methods have a high degree of freedom with regard to the choice of tracer and the dosing strategy. The application of any of these techniques requires the investigator to make a number of decisions regarding experimental design which influence the amount of uncertainty in the final estimate of exfiltration. A procedure for optimal experimental design has been proposed, which is based on the analysis of uncertainty (Rieckermann et al., 2005b).

For the QUEST method, it is necessary to choose the starting time of the experimental campaign, the number of tracer additions to use, the mass of tracer used in each addition and the relative timing of additions. We found that different experimental setups can lead to very different degrees of uncertainty in the measured exfiltration ratio. However, there is no universal answer to the question what is the optimal choice of options.

One important finding is that the best experimental design depends on how much the investigator is willing to spend on the monitoring process (in terms of effort and financial resources) and how these expenditures compare with the consequences of an incorrect determination of exfiltration. This issue is addressed using the formal framework of decision analysis (DA). The core of the procedure consists of two important elements: a simulation model that is used to predict the outcome (accuracy of exfiltration estimates in a reach) and a preference model that is necessary to evaluate what level of accuracy is desired. We decided to use DA for the experimental design
because it can be easily adapted to the individual requirements of the situation (e.g. drinking water protection zone or rural area, cost limits, etc.) of the end user and as such provides an open framework that can be easily extended if necessary.

3.1.5 Dispersion in sewers

An important task which usually precedes the experimental design is the estimation of the necessary mixing length in the sewer and the determination of the time of passage of a tracer peak at the sampling station. This is often investigated by preliminary measurement campaigns to examine the transport properties of the sewer reach under investigation. However, in some cases it might be desirable to predict the development of the tracer signal from simple transport models without much experimental effort.

60 tracer experiments carried out by APUSS partners in 37 different sewer reaches have been analysed for longitudinal dispersion. It was found that dispersion coefficients of sewers are two to three orders of magnitude smaller than those measured in rivers and do not differ much from system to system. Suitable equations were identified to: i) deduce dispersion coefficients from tracer data and ii) predict reasonable dispersion coefficients in sewer reaches with uniform geometry and stable flow conditions. Limitations of dispersion theory were also investigated. This research is documented in Rieckermann et al., 2005d.

3.1.6 Limitations of the methods

In order to develop a sound framework of exfiltration measurements with tracers, it is also necessary to point out the potential limitations of the QUEST and QUEST-C methods. From the experience gathered in the APUSS project, four main limitations may hinder their application or even make it impossible: insufficient flow, incomplete mixing, highly unsteady flow conditions and application in certain network situations with highly non-uniform flow.

Therefore, further effort will be devoted to investigate whether procedures can be found that allow for an identification of sewer reaches in which the transferability of tracer mass loss to meaningful exfiltration values may be questionable (Rieckermann, 2005).

3.1.7 Documentation, applicability and practical experience

Both the QUEST and QUEST-C methods have been documented thoroughly. Standard operation procedures for the performance of field experiments and data analysis have been provided. The underlying theoretical concepts and statistical methods for data analysis and experimental design are described in scientific publications. The algorithms for data analysis and experimental design, which have been programmed in the R language (Ihaka and Gentleman, 1996), are packed in libraries. R is available under the GNU public license; all documents, libraries and code examples are freely available on the APUSS website for public use.

Several test runs have been performed under a variety of conditions in different European cities and it was found that both QUEST and QUEST-C are generally applicable in practice. Our practical experience with the tracer methods is limited to open channel flows and most experiments have been performed under dry weather conditions.

The success of the tracer methods crucially depends on a thorough preparation of the experimental campaign and the local boundary conditions at the investigation site (flow, tracer background, accessibility, etc.). As a result, a final conclusion on the overall precision of the tracer methods does not seem conclusive. However, we expect that the uncertainty in the obtained tracer loss should be in the order of a few percent of total flow, if the measurement campaign is prepared and performed with the required care and according to the provided documentation on error analysis.

3.2 INFILTRATION METHODS

Two innovative tracer methods for the quantification of parasitic discharges in sewers have been developed. The stable isotopes method uses the different isotopic signatures of mains water and infiltrating water as a direct natural tracer. A sufficient isotopic separation can exist, when drinking water originates from a distant hydrological regime, but parasitic water stems from subsurface waters recharged by local precipitation. The pollutant time series method estimates the fraction of infiltrating water from a combined analysis of measured time series of pollutant concentrations and wastewater discharge. It requires a certain dynamics of the wastewater hydrograph and pollutograph, which is usually suited by the diurnal variations of the wastewater discharge. The methods were evaluated and tested under field conditions and have been validated in the course
of comparative experimental studies (see also Chapter 4). They are proposed for routine applications on the catchment or subcatchment scale.

In conventional practice, the quantification of infiltration/inflow is based on more or less complex statistical analyses that only take into account wastewater flow time series. The expressiveness of the results obtained with such procedures is often subject to some restrictions, due to their rather simplified underlying assumptions (i.e. that the hydrographs diurnal night-time through equals the amount of extraneous discharge). In order to improve the accuracy of wastewater hydrograph separation and therewith obtain more reliable information on the hydraulic interaction between leaky sewers and their ambient aquifers, the use of natural tracer signals was considered to be an optimal alternative. Both novel approaches are using intrinsic physico-chemical characteristics of the wastewater as a natural tracer for the differentiation of its constituting components (i.e. “real” foul sewage or infiltrating water).

3.2.1 Concepts

The wastewater hydrograph separation with both methods basically makes use of a tracer mass balance approach to differentiate discharge components from two or more sources. In a simple binary mixing system, the fraction of wastewater discharge that is stemming from infiltration (infiltration ratio \( X_{\text{Infiltration}} \)) and the corresponding amount of infiltration discharge (\( Q_{\text{Infiltration}} \)) are given by:

\[
X_{\text{Infiltration}}(t) = \frac{C_{\text{Foul Sewage}}(t) - C_{\text{Wastewater}}(t)}{C_{\text{Foul Sewage}}(t) - C_{\text{Infiltration}}(t)}
\]

\[
Q_{\text{Infiltration}}(t) = X_{\text{Infiltration}}(t) \cdot Q_{\text{Wastewater}}(t)
\]

where \( Q \) is the discharge and \( C \) the tracer concentration.

\( C_{\text{Wastewater}} \) and \( Q_{\text{Wastewater}} \) can principally be measured directly in the sewer. On the other hand, \( C_{\text{Infiltration}} \) and \( C_{\text{Foul Sewage}} \) are typically hardly accessible for investigations at the catchment or even subcatchment scale. When foul sewage or infiltration enter the sewer system, they are immediately mixed with the wastewater stream and thus any separate analysis is hindered. As a consequence, the investigation of foul sewage and infiltrating water must be performed indirectly. This leads to the idea of examining the drinking water and the local groundwater, which constitute their intrinsic origins. Yet, infiltration cannot be measured using artificial tracers. Artificial labelling of drinking water is prohibited and homogenous distribution of tracers throughout an entire aquifer is neither feasible nor desirable for environmental reasons. Rather, specific natural characteristics of the local drinking water, groundwater and sewage have to be used as natural indicators of mixing and dilution processes. However, since generally a vast number of dissolved species are added to the foul sewage, it is basically rare to find such suitable natural tracers. Most chemical components exhibit large daily fluctuations and therewith obscure the natural tracer signals.

To overcome the above limitations, two generic methods have been developed which principally differ in the utilized tracer signal and the associated experimental setup and investigations.

The stable isotopes method uses the stable isotopes compositions of mains water and local groundwater as suitably proxies for the stable isotopes compositions of the foul sewage and the infiltrating water respectively. It therewith allows for a direct calculation of infiltration ratios from eq. 3. The method is suited to quantify infiltration, where useable differences in the oxygen or hydrogen isotope ratios of the drinking water and the infiltrating water exist. The proper use of this method requires a detailed hydrologic and hydrogeological investigation to define the principal hydrologic interactions and pathways of different waters in the catchment.

In contrast to the stable isotopes approach, the pollutants time series method does not depend on a direct investigation of drinking water or infiltration origins. The fraction of extraneous water is determined by analysing time series of pollutant concentrations and wastewater discharge that are obtained from measurements carried out at a single point in the sewer system. A characteristic feature of this method is the use of automatically operating in-line devices to obtain time series of pollutant concentrations with a high temporal resolution. Based on the recorded time series of wastewater discharge, a modelled time series of pollutant concentrations is calculated. By fitting this model series to the measured data, a set of parameters defining \( Q_{\text{Infiltration}} \) can be estimated.
In the following paragraphs, a synthetic overview on the two infiltration methods is given. In-depth information on the specific topic is provided in the cited references and is also available on the APUSS website.

### 3.2.2 Principle of the stable isotopes method

Natural water (H₂O) is composed of the three stable isotopes of oxygen (³¹O, ¹⁷O and ¹⁹O), the two stable isotopes of hydrogen (¹H and ²H or D = deuterium) and the unstable isotope Tritium (³H). Compared to most other natural water characteristics, the stable isotopes composition of the water molecule is very robust with respect to changes in water chemistry or biological activities. Particularly, it is supposed to be not affected by utilization in the urban infrastructures and therefore constitutes a suitable direct natural tracer for our purpose.

Within the global water cycle, the stable isotopes composition of water is predominantly controlled by large-scale meteorological evaporation and precipitation effects. Evaporation from the oceans is the major source of atmospheric water masses. As clouds moves across the continents, the heavy water molecules containing D and ¹⁸O will preferentially rainout first in coastal areas and the remaining water vapour is continuously depleted in these isotopes, i.e. isotopic fractionation (Figure 3). The isotopic composition of natural waters with meteoric origin is in consequence determined by the continental setting (continental effect) and topographic elevation (altitude effect) of the region where ground water and surface water are recharged by precipitation (Craig, 1961, Rozanski et al., 1993).

![Figure 3 : Modelled evolution of the isotopic composition in precipitation from a cooling vapour mass.](image)

The isotopic composition of water can be utilized as an infiltration tracer for our purposes, when a city uses drinking water from a watershed that is situated in a different hydrological regime than the urbanized area. As a possible result of such a setting, significant differences in the isotopic compositions of the drinking water, the wastewater and the groundwater may exist, allowing for quantification of the various mixing ratios (Kracht et al., 2003). Beyond this prerequisite of a sufficient isotopic separation between the mixing endmembers, it is critical to verify a low spatial variability of the isotopic composition within the ambient aquifer (Kracht and Gujer, 2005).

![Figure 4 : Decomposition of a diurnal wastewater hydrograph based on the stable isotopes approach. Thick lines mark estimated mean values, thin lines indicate the 2σ-level confidence interval (not shown for QWastewater).](image)
Figure 4.a displays measured $\delta^{18}$O values from an experimental catchment site in Rümlang, Switzerland. The groundwater and drinking water endmember ranges are compared with $\delta^{18}$O values of a series of 28 hourly wastewater samples taken from a connector sewer at the outlet of the village catchment. As expected, the wastewater is isotopically enriched during night time, indicating a higher fraction of “heavy” infiltrating water being present in the sewer. During day time the wastewater shows isotopic depleted values, due to the larger amount of lighter foul sewage being discharged. Based on these data, the infiltration ratio and the corresponding infiltration discharge are calculated, as shown on Figure 4.b.

3.2.3 Limitations of the stable isotopes method

The underlying concept of the stable isotopes method defines parasitic infiltration as a fraction of the total wastewater that is carrying a specific isotopic signature. It is therefore generally limited to catchments or subcatchments where both drinking water and groundwater have homogenous isotopic signatures, and where only two components (one drinking water source, one groundwater source) are interacting. The method has been tested more extensively in Rümlang (CH), Lyon (F) and Rome (I) (see Chapter 4). The two critical system properties are the existing isotopic separation between drinking water and infiltration and the natural variability of possible infiltration sources. The question of inhomogeneities of the local groundwater or other origins of parasitic waters can be crucial for the applicability of the method. For this reason it is mandatory to conduct a broader hydrologic and hydrogeological investigation to obtain a general conceptual model describing the principal hydrologic interactions and pathways of different waters in the catchment. The principle sources of infiltration (i.e. groundwater) need to be definable, accessible for sampling and statistically describable with sufficient precision.

3.2.4 Principle of the pollutant time series method

The method quantifies infiltrating non-polluted waters based on measured time series of wastewater flow and pollutant concentrations. Based on a transformation of eq. 3, the analysis uses a mixing model describing the pollutant concentration in dependency of wastewater discharge and time (eq. 5). In contrast to earlier tracer based approaches, it is based on high temporally resolved measurements with in-line devices, which allows for considerable refinements of the data analysis. It does not require the assumption of a stationary cyclic behaviour of the diurnal hydrograph and pollutograph. The underlying model structure can consider temporal fluctuations of the pollutant concentration in the foul sewage (eq. 6) as well as time dependencies of the infiltration rate (i.e. dynamics of rain-induced infiltration). For the latter, the amount of infiltrating water is conceptually divided into a constant baseflow and an exponentially receding interflow (eq. 7):

$$C_{\text{Wastewater, model}} = \left(\frac{Q_{\text{Wastewater}} - Q_{\text{Infiltration}}}{Q_{\text{Wastewater}}}\right) C_{\text{Foul Sewage}} + \frac{Q_{\text{Infiltration}} \cdot C_{\text{Infiltration}}}{Q_{\text{Wastewater}}} \cdot C_{\text{Wastewater}}$$  \hspace{1cm} \text{eq. 5}

$$C_{\text{Foul Sewage}} = f(t) + f(Q_{\text{Foul Sewage}})$$  \hspace{1cm} \text{eq. 6}

$$Q_{\text{Infiltration}} = Q_{\text{Baseflow}} + Q_{0,\text{Interflow}} \cdot e^{-k_{\text{rec}} \cdot (t-t_{0,\text{interflow}})}$$  \hspace{1cm} \text{eq. 7}

with $Q_{0,\text{interflow}}$: initial magnitude of interflow at the time $t_{0,\text{interflow}}$; $k_{\text{rec}}$: recession constant.

The application of eq. 5 requires information on the tracer concentration in the infiltrating water, which is typically not identifiable from the time series itself. A suitable parameter for this application is the chemical oxygen demand (COD), which was utilized in most of the APUSS applications. The COD concentration in the parasitic water can in most cases assumed to be negligible, which avoids some difficulties in characterizing the tracer mass contribution stemming from the infiltration itself. Recently available submersible spectrometer probes allow for a direct measurement of COD-equivalents in the media by the means of light absorbance in the UV-VIS range (Langergraber et al., 2003). The use of automatic measuring devices considerably simplifies the recording of longer time series in a high temporal resolution, which significantly improves the documentation of the wastewater composition dynamics. $Q_{\text{Infiltration}}$ is finally identified by fitting a modelled time series of pollutant concentrations to the measured data (Kracht and Gujer, 2004). Figure 5 displays results from the experimental catchment site in Rümlang. The measurements were performed during a dry weather period. Last rainfalls occurred up until three days before this campaign.
Infiltration measurements with the pollutant time series approach are very flexible. The capability to consider natural storage and interflow phenomena significantly broadens the practical applicability of the method and particularly simplifies the measurements in rainy seasons, when infiltration rates typically increase due to elevated groundwater tables. The experimental protocols have been tested within several measurement campaigns performed by APUSS partners in different catchments in Berlin, Bottrop, Dresden, Gladbeck, London, Lyon, and Rome (see Chapter 4).

Figure 5: Decomposition of a diurnal wastewater hydrograph based on the pollutant time series approach. Left: COD measurements; Right: calculated hydrograph separation.

3.2.5 Limitations of the pollutant time series method

From the experience gathered in the APUSS project, we can conclude that the successful implementation of the methods crucially depends on a thorough preparation of the experimental campaign and the local boundary conditions at the investigation site (flow conditions, accessibility). One of the main limitations of the method seems to be bound to the required bias and drift free operation of the submersible UV-VIS spectrometer (that has been chosen by all investigators). As one finding it has to be accepted that the continuous monitoring of wastewater quality parameters with optical measurements in raw sewage is obviously not a trivial task. Measurement quality strongly depends on the skills and experience of the technical field team and the required investments for a successful measurement campaign should not be underestimated.

The method may be used at the outlet of any subcatchment where a continuous discharge of wastewater can be assured. However, a minimum amount of wastewater flow is required for the disturbance free operation of the measuring devices, which may be critical during minimum night flow. Furthermore, predominant types of industrial effluents should be excluded, as this may hinder a regular data analysis.

3.2.6 Documentation

Both infiltration methods have been documented in standard operation procedures for the performance of field experiments and data analysis. Algorithms for data analysis of the pollutant time series have been programmed in the R language (Ihaka and Gentleman, 1996) and are packed in libraries. R is available under the GNU public license; all libraries and code examples are freely available on the APUSS website for public use.

3.3 Effects of sediments and sediment tracers interactions

3.3.1 Tracer adsorption and interaction

A series of laboratory batch tests were carried out to investigate the interactions of fluorescent and non-fluorescent tracers with a range of sewer sediments and fresh raw sewage samples in order to assess respectively the impact of in-pipe deposits and adsorption effects under sewer flow conditions. Little evidence of lithium adsorption (< 2% over 24 hours) was observed for any of the tested sediments (including sewer slimes and algal mats) but an overall impact of up to 10% adsorption was recorded for Rhodamine WT. Tests on raw and settled sewage samples showed no significant impact on lithium tracer concentrations according to whether the sewage is fully mixed due to stirring (to simulate turbulent sewer flow) or was allowed to settle under quiescent conditions (simulating sewer backwater effects). Similar results were observed when bromide was used as a tracer but Rhodamine WT demonstrated more complex interactions in which the aqueous tracer concentration...
continued to decrease during the stirring process resulting in an overall 10-15% reduction of the fluorescent signal. A clear relationship between turbidity and an increase in the fluorescent signal was established due to the measurement technique and so turbidity corrections were applied to compensate for the effect of turbulence. The use of other fluorescent tracers (Sulphorhodamine B and Sodium Fluorescein) with raw sewage suggests that the former is more susceptible to adsorption (25-35%) but that Sodium Fluorescein is less effected by adsorption with some sewage samples showing no impacts. The adsorption characteristics of Rhodamine WT were accurately modelled by Freundlich isotherms suggesting that removal occurs by heterogeneous surface binding rather than as a result of a surface monolayer formation.

In-sewer tests using a fluorescent probe at locations in N London and Reading for both individual flush events and continuous dry weather flow demonstrated the practicality of the instrumental procedure for sensitive, real-time measurement. The experimental work would thus suggest limitations to the use of fluorescent tracers as quantitative indicators of exfiltration in sewers particularly over long travel times within a sewer network. In contrast, lithium and bromide demonstrated negligible adsorption effects although, in both cases, considerable laboratory effort was required to overcome the significant analytical interference, in the case of lithium due to the high level of sodium in sewage.

3.3.2 Effect of sewer solids and sediments on exfiltration

A laboratory experimental sewer rig using clean water was established to evaluate exfiltration losses under varying joint openings (1.0 to 3.5 mm with varying geometries) and flow conditions (0.01-7.8 L/s). Joint surface areas varied between a 10 mm basal hole and half horizontal and vertical slots (to simulate cracks in the sewer wall). In addition to free draining tests, runs were also undertaken using a 1 m x 0.65 m x 0.85 m trench backfill surrounding the joint/gap opening with test runs conducted under both dry gravel and fully saturated gravel conditions. In the latter case “groundwater” level in the gravel backfill was maintained at surface level in the simulation trench. The test outcomes for free draining conditions confirm a general linear exfiltration rate against both static and pressure head for all 1 mm gap geometry experiments as well as for all 10 mm hole experiments at the 2 mm and 2.5 mm gap sizes. However, for the remaining 2 mm and 2.5 mm experiments, exfiltration rates demonstrate a general exponential trend with rapid initial losses. At equivalent dry weather flows i.e. 10-20% sewer flow levels, the overall losses appear to be in the order of 1 to 2%. At discharge rates above Q25 (Qn corresponds to a n% pipe filling ratio expressed a the fraction of the sewer height), the forward velocity is sufficient to convey the effluent across the gap without incurring additional exfiltration loss. There was a tendency for exponential exfiltration rates to occur for all dry and wet trench backfill test runs with increasing inflow and pressure heads apart from the 10 mm hole rates where very low linear losses were recorded. Test runs using introduced toilet paper and a variety of sediment types (sand, peat, synthetic non-cohesive particles and mixes) showed that plugging and bridging of the joint gaps occurs such that leakage rates exponentially decline to levels of 10-4-5 L/s with the joints becoming effectively sealed over the long term (Ellis et al., 2003).

A pilot field experimental sewer rig was also established at Dundee with additional separate funding provided by a consortium of UK Water Companies to reproduce the laboratory test conditions on live wastewater. Complete sealing was eventually observed in 50% of the experiments for the 10 mm hole and the other runs produced lowest recorded exfiltration rates of between 10-2 and 10-4 L/s. More effective sealing was associated with higher heads, except in the case of the widest gap (10 mm). There was a considerable variation in the time taken to reach the lowest recorded exfiltration rates and this is probably due to randomness in the time of arrival of suitable sealing material in the sewage flow. The lowest recorded levels in each experiment for radial defects with 3 mm and 6 mm gap sizes were also in the order of 10-2 to 10-4 L/s, with more effective sealing occurring at higher heads. However, much higher exfiltration rates, of the order of 10-3 L/s, were recorded for the 10 mm gap width. Generally, the exfiltration rates were tending towards equilibrium values after between 60 and 120 minutes. The rates of exfiltration as a percentage of flow in the pipe were also calculated and, in most cases, were between 0.1 and 1%. However, values of around 5% for lower heads and between 15 and 30% for the largest gaps were found. Components of the data sets have been analysed by a grouping process to provide a prediction of exfiltration rates after 4, 8 and 16 hours. Whilst the predicted range was wide, the extrapolation suggests that exfiltration rates would reduce from those observed above by an order of magnitude to 10-4 to 10-5 L/s over an 8 hour period and approach zero after around 16 hours. This finding reinforces the previous laboratory rig work but it should be noted that the limited nature of the data set and the degree of fit of the selected function, prevents any clear conclusions from being drawn.

An additional series of experiments have provided a clearer understanding of the impact of the presence of a gravel surround on exfiltration rates. Complete sealing eventually occurred on every run for the 10 mm hole defect and the lowest recorded levels in each experiment for the radial defects were in the order of 10-3 to
The exfiltration rates tended towards equilibrium values over shorter run times than for the non-gravel runs with this condition typically being achieved between 12 and 144 minutes. The rates of exfiltration, expressed as a percentage of flow in the pipe were, in most cases, between 0.1 to 1% of the flow, but rates of around 1 to 5% were observed for the radial defects. Overall, the gravel surround leads to more effective sealing of the full range of defects compared to those observed for the free-draining runs. Full detail of the experimental rig, test run data and the study outcomes are given in Blackwood et al. (2004, 2005) and complete reports are available on the APUSS website.

The results of the test experiments have been scaled up to estimate leakage rates in sewer lengths and in associated subcatchments (Ellis et al., 2004). As an initial estimation, assuming that sewer joints occur at intervals of 2.5 m and that 10% of joints are defective in a 1 km sewer length, the wastewater rig outcomes for 10 mm wide defects in pipes having dry gravel trench surrounds, would imply a worst-case ultimate overall loss rate of between 130-1300 m$^3$/km.year. If this were scaled-up to the 25% grade 5 and 4 sewers of the total 64 374 km network of Thames Water, this would yield losses due to exfiltration of between 2.1M and 21M m$^3$/year. Whilst this figure is much less than previously reported (e.g. a CIRIA (1996) 3% estimate of 109M m$^3$/year), it is still a concern for areas having abstractive groundwater beneath cities in the light of the requirements of the Water Framework Directive. The pilot-scale experimental work also strongly suggested that sewers carrying a high sediment load are likely to seal gaps and cracks in the sewer walls and invert over relatively short periods of time, and thus even the reduced exfiltration values are likely to be over-estimates. The work would also suggest that sewer cleaning may substantially reduce the integrity of the sewer by re-opening the joints and this might also occur during periods of sewer surcharging under flood conditions.

The required dosing conditions (flow rates and concentrations) have also been optimised for the use of Rhodamine WT and lithium tracers to determine exfiltration rates within the experimental sewer rig. The performances of both tracers were successfully compared with direct volume measurements for the determination of percentage exfiltration values. The good agreement between the two approaches appears to be improved for smaller gap sizes but procedures have been identified to overcome these discrepancies through modifications to the experimental design. No evidence was found for adsorption of Rhodamine WT on to sewage solids during the pathways existing within the sewer rig and initial investigations suggest that the ease of detection of this tracer makes it ideal for further pilot-scale experiments and for use in real sewer experiments.

### 3.4 INFILTRATION AND EXFILTRATION IN HOUSE CONNECTIONS

The above measurement methods (sections 3.1 and 3.2) are applicable to infiltration and exfiltration in public sewers pipes. But it is well known that house connections (HC) are also key components in I/E phenomena. Measurements are performed on selected catchments, in order to obtain a large data set from different types of built-up area, age of pipes, geology and hydrology conditions, etc. Volumetric methods have been tested and applied on experimental sites to measure I/E in HC. As there are many thousands of HC in a city, it is impossible in practice to measure I/E in all of them. This is the reason why a matrix procedure for extrapolation to wide catchment offering similar HC characteristics has been established.

#### 3.4.1 Infiltration

Infiltration into HC occurs principally as a result of one or more of the following conditions:
- the pipe is damaged,
- the level of surrounding water (ground-water or percolation of rainwater) lies above the damaged pipe section,
- the damaged pipe section is surrounded by permeable soil or a soil opening.

The evaluation of the amount of water infiltrated into a HC includes i) the identification of the defect and ii) the measurement itself. The application of a particular measurement method depends on the arrangement of the HC to the sewer system. Typical arrangements of HC to main sewers are as follows:

A. HC in manhole  
B. HC to a man-entry sewer  
C. HC to a non-man-entry sewer (sewer diameter smaller than 0.8 m).

In cases A and B, a proper volumetric measurement is possible. In case C, it is impossible to measure individually each house connection, but it is possible to measure a group of HCs – which essentially make up a small subcatchment.

Two infiltration measurement methods have been tested:
- one based on accurate flow monitoring during a couple of days at the outlet of a HC or in a manhole connecting a few houses to public sewers. This allows estimating the infiltration rate as equivalent to the minimum night flow. As the discharges to be measured are very small, specific weir and water level sensors have to be installed on site.
- another one based on chemical analyses of samples collected during the night for comparison with the minimal discharges. These chemical indicate the dilution of wastewater. This result is compared with values of wastewater pollution obtained during daylight hours. Appropriate indicators of pollution are typical wastewater quality indicators, like COD (mg.L⁻¹), N-NH₄⁺ (mg.L⁻¹), temperature (°C) and conductivity (S.m⁻¹).

3.4.2 Exfiltration

The methodology of exfiltration measurement is based on water tightness tests of sewers. The value of leakage must be compared to the permissible values as stated in European or national technical standards (e.g. the German ATV Standard M149 (ATV, 1999a).

The leakage of sewage water from a HC suffering from a particular damage occurs only upon the fulfilment of the following conditions:
- the pipe is damaged (cracks, broken joints, holes, ...),
- the ground water level (GWL) is lower than the level of the pipe or when the hydrostatic pressure in the pipe exceeds the groundwater pressure,
- the pipe is placed in a permeable environment.

The exfiltration assessment includes i) the evaluation of pipe condition as well as catchment conditions (relation between type of damage and main factors, soil environment, age,...), and ii) the measurement itself.

Two exfiltration measurement methods have been tested: the pressure method and the free surface method. The pressure method consists to block the HC pipe with sealing balloons, to fill it with water, and to measure the decrease of the water level as a function of time, from which the exfiltration rate is calculated. The free surface method consists to discharge a given volume of water in the HC pipe, to collect the volume downstream the HC pipe and to calculate the exfiltration rate from the comparison of both volumes. The first method overestimates exfiltration because of the pressure, while the second one requests an easy access in the public sewer downstream the HC, which is rarely possible.

The catchment is characterized by the relative exfiltration rate and by values obtained from the interactive matrix (see Figure 6) which is filled from CCTV inspection records to evaluate the structural state of the HC and its context. The exfiltration from a small subcatchment (i.e. a few house connections) is generally a function of the discharge Q and of the factor w which reflects the catchment evaluation given by the interactive matrix:

![Figure 6: Structure of the HC interactive matrix.](image)
3.4.3 Scaling-up method

As, for obvious practical reasons, all HC in a given catchment cannot be measured, it has been suggested (Princon and Kohout, 2003) to assess the I/E rate for all HC from a large catchment by extrapolation of measurement data obtained in a small reference catchment (i.e. some HC).

The procedure is as follows:

1) Pipe state evaluation in the reference catchment: classification of damages, evaluation of catchment conditions (ground water level, backfill, ...), filling of the interactive matrix.
2) I/E measurements in HC pipes in the reference catchment.
3) Pipe(s) evaluation in the large catchment: classification of damages, evaluation of catchment conditions (ground water level, backfill, ...), filling of the interactive matrix.
4) Extrapolation of results from the reference catchment to the large catchment, by means of the following equations:

\[
I_{LC} = I_{RC} \cdot \frac{\Sigma L_{LCi}}{\Sigma L_{RCi}} \left( \frac{A - w_{LC}}{A - w_{RC}} \right) \quad \text{eq. 9}
\]

\[
E_{LC} = E_{RC} \cdot \frac{\Sigma L_{LCi}}{\Sigma L_{RCi}} \left( \frac{A - w_{LC}}{A - w_{RC}} \right) \quad \text{eq. 10}
\]

where:

- \( I_{LC} \), resp. \( E_{LC} \) infiltration/exfiltration in large catchment
- \( I_{RC} \), resp. \( E_{RC} \) infiltration/exfiltration in reference catchment
- \( \Sigma L_{RCi} \) total length of HC pipe in reference catchment
- \( \Sigma L_{LCi} \) total length of HC pipe in large catchment
- \( w_{RC} \) reference catchment evaluation – based on evaluation using interactive matrix
- \( w_{LC} \) large catchment evaluation - based on evaluation using interactive matrix.
- \( A \) value expressing the best pipe condition or pipe without any damage according to standard using for classification of damages for optic inspection.

3.4.4 Conclusions

When estimating I/E in HC, the most important thing is to correctly classify pipe defects and to evaluate the catchment conditions (qualities).

The pressure method may be limited by the possible amount of water supplied to the tested HC: it is almost impossible to accurately evaluate the amount of exfiltration in HC having large-scale defects, e.g. a missing pipe segment, a large gap between 2 pipes, etc. However, these cases are extreme and that is why pipes with such large defects should be subject to thorough reconstruction first.

Separate measurement of I/E from individual HC is possible only in especially favourable cases. For this reason, it is suggested to investigate several house connections and connecting sewers in groups. The connecting sewer having similar characteristics to a HC could be considered equivalent to a house connection. The layout of the measurement depends thus strongly on local conditions.

Transferability of results between different catchments is possible only under the following conditions:
- only between close reference and large catchments in the same city.
- reference and large catchments are related or show similar characteristics from the interactive matrix.
- extreme cases must be excluded.

For scaling-up, it is important to correctly evaluate both the pipe condition in the reference catchment and all the pipes condition in the large catchment.
4. TEST AND FIELD APPLICATIONS OF METHODS

This chapter gives some brief examples of application of the methods presented in Chapter 3 in various cities in Germany (section 4.1), France (section 4.2), Italy (section 4.3) and the UK (section 4.3.1). These examples i) show that the methods are applicable for very different contexts and sewer systems, ii) give some orders of magnitude of infiltration and exfiltration rates, and iii) provide some practical comments on the use and application of the methods. Detailed reports and publications are available on the APUSS website.

4.1 APPLICATIONS IN GERMANY

4.1.1 Exfiltration

In total, 29 exfiltration measurements have been carried out in Dresden and Berlin by applying the QUEST and/or QUEST-C methods described in section 3.1. Some sewer lines have been investigated with both methods. In Figure 7, the exfiltration rates of the measurements linked to the mean flow rate during the particular measurement are shown. The measured exfiltration rates average to approximately 1 L/s, which equals 86 m³/d. After analyses of the CCTV records, the exfiltration rates were referred to the leakage area (Figure 8). The number of useful measurements further decreased since no CCTV data were available for the sewer lines in Dresden.

![Figure 7: Measured exfiltration rates [L/s] related to a mean flow rate.](image)

![Figure 8: Measured exfiltration rates [m³/d/cm²] depending on the leakage area [m²].](image)
As a plausibility test, the exfiltration rates obtained in real sewers were compared to the results of various laboratory experiments described in the literature. Field measurements exhibited exfiltration rates of only 10% of the exfiltration rates obtained in the laboratory experiments. The deviation can be attributed to a concentration on single defects in the laboratory experiments, to the steady state condition during one experiment (pressure test) and in some cases also to the soil conditions in the laboratory experiments.

The practical applicability of QUEST is good and not complicated. The provided standard operation procedure is a good basis to perform the measurements, but it has to be adapted to specific individual and local requirements. Preliminary campaigns are needed before carrying out the experiments, such as tracer baseline and flow recording. High background conductivity causes serious difficulties for the measurement with sodium chloride. The data analysis is rather sophisticated and provides exfiltration rates and measurement uncertainties.

The application of the QUEST-C method is relatively simple. However, the set-up is sophisticated and expensive (at least during the prototype phase of the APUSS project). As the utilised laboratory equipment is not necessarily adapted and appropriate for field application, a very careful handling is required. Furthermore, the analysis of the samples requires a high accuracy equipment in the laboratory.

In order to obtain reliable results, detailed information is required, especially on flow conditions and on natural tracer baseline concentrations. To diminish the measurement errors in both methods, a high number of tests has to be performed. It is highly recommended i) to follow the standard operation procedure and ii) to do some training on the method and on the equipment in the field.

4.1.2 Infiltration

The pollution time series method (see Section 3.2) was applied in 6 subcatchments:
- 2 measurements in collaboration with Emschergenossenschaft in Gladbeck and Bottrop
- 1 measurement in collaboration with BWB (Berlinerwasserbetriebe) in Berlin
- 3 measurements in collaboration with Stadtentwässerung Dresden in Dresden.

Site characteristics and results are summarised in Table 2.

<table>
<thead>
<tr>
<th>Site</th>
<th>Gladbeck</th>
<th>Bottrop</th>
<th>Berlin</th>
<th>Salzb Street</th>
<th>Mueva</th>
<th>Lockwitz</th>
</tr>
</thead>
<tbody>
<tr>
<td>Average dry weather flow</td>
<td>12 Ls⁻¹</td>
<td>10 Ls⁻¹</td>
<td>50 Ls⁻¹</td>
<td>7 Ls⁻¹</td>
<td>7.4 Ls⁻¹</td>
<td>12 Ls⁻¹</td>
</tr>
<tr>
<td>Characteristics</td>
<td>Suburban catchment with well described groundwater and sewer conditions</td>
<td>Homogenous densely populated area with a low groundwater table</td>
<td>varying groundwater levels induced by the river Elbe</td>
<td>Homogenous, suburban catchment, influenced by groundwater and a small stream next to the sewer</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Inhabitants</td>
<td>3 670</td>
<td>4 080</td>
<td>60 000</td>
<td>4 455</td>
<td>1 300</td>
<td>2 850</td>
</tr>
<tr>
<td>Infiltration rate</td>
<td>27.6 %</td>
<td>14.6 %</td>
<td>5 %</td>
<td>0 %</td>
<td>46 %</td>
<td>57 %</td>
</tr>
<tr>
<td>Measuring period</td>
<td>2 weeks</td>
<td>4 weeks</td>
<td>4 weeks</td>
<td>6 months</td>
<td>3 months</td>
<td>5 days</td>
</tr>
<tr>
<td>Comment</td>
<td>Exponential decrease of infiltration rate induced by preceding rain</td>
<td>Validation of exfiltration measurements</td>
<td>Long time investigation including investigation of exfiltration</td>
<td>Long time series of flow measurements, probe was not calibrated</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

To obtain good results, the following prerequisites must be fulfilled for the measurements:
- good conditions such as power supply and support by the local authorities reduce the otherwise very high effort to maintain the probe at the measuring station.
- well trained staff on site for reading out data and sampling and in the laboratory for sample analysis is highly recommended.
- a sampling campaign is mandatory to assure a suitable calibration of the on-line instruments. A huge deviation was recorded in Gladbeck comparing the results for a calibrated and a non-calibrated probe.
- small alterations in the hydrograph or pollutograph might already indicate measurement errors.
4.2 APPLICATIONS IN LYON, FRANCE

The results presented in this section have been obtained in the Ecully experimental catchment, located in the west suburbs of Lyon.

4.2.1 Exfiltration

After preliminary tests had been carried out between April 2003 and June 2003 to train the team and to get adequate information on the site, the QUEST protocol was adapted to the specificity of the investigation reach. The utilized solution of tracer had a concentration in NaCl between 170 and 180 g/L. The volumes of solution used for the indicator pulse and the reference pulse were respectively 25 L and 4 L. The time interval between two indicator pulses was 20 minutes and the first reference was injected 10 minutes after the first indicator, in order to obtain fully separate peaks. Every measurement campaign had a duration of two hours and was performed with 4 reference and 3 indicator pulses. An example of conductivity time series obtained during an experiment is illustrated in Figure 9. The reference and indicator signals present peaks of conductivity in about the same concentration range, between 3500 and 4200 µS/cm. The conductivity baseline is stable during the experiment at 700 µS/cm. The exfiltration ratio (%) was calculated for each indicator pulse by taking into account the two references peaks directly before and after its passage. An average exfiltration ratio with an associated uncertainty was calculated for each of the six measurements campaign realised between July 2003 and July 2004.

The uncertainty calculations were performed with a preliminary version of the data analysis routine, in which the uncertainty in the conductivity baseline was not considered. However, as this baseline was rather stable during the experiments, the associated uncertainty is assumed here to be low and negligible compared to the other uncertainties. The uncertainty calculation considers the analytical uncertainty on one single value of exfiltration ratio closed to 1 % and the uncertainty linked to the distribution of the values of exfiltration ratio which depends essentially on the variability of the wastewater flow rate during the experiment.

![Figure 9 : Example of experimental results during a measurement campaign of exfiltration (July 2003).](image)

The results obtained during the six measurements campaigns carried out in Ecully are illustrated in Figure 10. The exfiltration ratio varies between 2 and 16 %. It can be observed that the rate of exfiltration exhibits some variability. Our hypotheses is that it i) decreases in periods of high ground water table (GWT) and ii) increases in periods of low GWT. In July 2004, an event-based variability is assumed. After a strong rainfall event (40 mm), the exfiltration ratio decreased from 15 to 10 %. Therefore, one can argue that exfiltration follows a temporal pattern which is similar to that of infiltration and that the variability of exfiltration is also influenced by rainfall frequency and intensity. Additional experiments will be necessary to confirm these hypotheses in order to develop a better understanding of these phenomena.
4.2.2 Infiltration

The most frequently used method in Ecully was the stable isotopes method, as favourable local conditions where observed, with clearly distinct origins of drinking water and of groundwater with different isotopic signatures. Numerous measurement campaigns have been carried out, at hourly, daily and seasonal scales.

Usually, infiltration flow is assumed to be constant at daily scale while strict wastewater flow is characterised by strong variations due to human activities with two peaks in the morning and in the evening. The daily cycle of strict wastewater (i.e. drinking water) can be observed in Figure 11 with a strong decrease of the contributions during the night period and flow peaks during the evening (20:00) and the morning (07:00-09:00). This figure also reveals that the infiltration rate (in m$^3$/h) present important hourly variations, which are partly explained by private groundwater pumping installations used by some inhabitants in Ecully.

A measurement campaign was carried out in December 2003, with several samples in order to study the spatial variability of the reference values for drinking water and for groundwater and to estimate the impact of uncertainty linked to the spatial variability in infiltration ratios. The main results are given in Table 3. The infiltration flow (in m$^3$/h) appears more important in the day period than in night period during which the infiltration ratio (in % of total dry weather flow) is higher. These values are due to the fact that what is named here infiltration is both real infiltration (i.e. groundwater entering the sewer pipes through defects) and groundwater discharged into the sewers by the private groundwater pumping installations.
Table 3: Estimated values of infiltration and their 95% confidence interval (i.e. 2σ) uncertainties on 17/12/2003.

<table>
<thead>
<tr>
<th></th>
<th>infiltration ratio (% of dry weather flow)</th>
<th>total dry weather flow QT (m³/h)</th>
<th>infiltration flow QINF (m³/h)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Day period</td>
<td>38.0 ± 12.4</td>
<td>167.2 ± 2.4</td>
<td>63.5 ± 20.8</td>
</tr>
<tr>
<td>Night period (02h00-06h00)</td>
<td>56.0 ± 15.4</td>
<td>74.6 ± 4.1</td>
<td>41.7 ± 11.7</td>
</tr>
</tbody>
</table>

### 4.3 Applications in Rome, Italy

The methods for assessing infiltrations and exfiltrations have been applied in two catchments in Rome, Infernetto and Torraccia. The first one is a residential area with a twenty years old separated sewer which geology is mainly characterized by sand and gravel; the second one is a residential area with a ten years old combined sewer which geology is mainly characterized by tuff.

#### 4.3.1 Exfiltration

In Torraccia, both QUEST and QUEST-C methods were applied, whereas in Infernetto only the QUEST method was applied.

**QUEST method**

The equipment used consisted of one submerged probe (Sigma 900max) for level and velocity measurements and three WTW conductivity probes (LF 197 device with sensor TetraCon 325). The conductivity data were recorded in a data-logger (GRANT SQ400) at a resolution of one second. The chemical tracer was NaCl with 97% purity.

Some preliminary measurements were carried out, particularly in order to determine the shape of the peak at the end of each investigated reach and then to evaluate the time interval between two successive dosages.

Among all the experiments carried out, five of them were selected in that:
- the peaks were non-overlapping and close enough each other;
- the mixing distance for the reference tracer dosed was correctly calculated;
- the natural conductivity of wastewater during the experiment was not too highly variable.

For these five experiments, the error propagation was performed using a routine written in R-script by Rieckermann (2004). The characteristics of the experiments are shown in Table 4. In conclusion, the accuracy of this method could be assumed between 2 - 3%.

Table 4: Characteristics of the QUEST experiments and results: T = Torraccia; I = Infernetto

<table>
<thead>
<tr>
<th>Experiment code</th>
<th>Investigated length [m]</th>
<th>Duration [s]</th>
<th>Flowrate [L s⁻¹]</th>
<th>Mean exfiltration ratio [%]</th>
<th>Coverage of the ground and backfilling</th>
</tr>
</thead>
<tbody>
<tr>
<td>030710T</td>
<td>800</td>
<td>2000</td>
<td>10.63 ± 0.50</td>
<td>7.11 ± 0.02</td>
<td>Grass, concrete backfilling</td>
</tr>
<tr>
<td>030724T</td>
<td>800</td>
<td>2000</td>
<td>12.50 ± 0.51</td>
<td>0.09 ± 0.19</td>
<td>Idem</td>
</tr>
<tr>
<td>030730T</td>
<td>800</td>
<td>2000</td>
<td>12.48 ± 0.45</td>
<td>5.94 ± 0.41</td>
<td>Idem</td>
</tr>
<tr>
<td>040616T</td>
<td>400</td>
<td>4000</td>
<td>8.48 ± 0.45</td>
<td>-1.73 ± 4.69</td>
<td>Idem</td>
</tr>
<tr>
<td>040408I</td>
<td>1500</td>
<td>9000</td>
<td>22.59 ± 0.45</td>
<td>1.66 ± 0.11</td>
<td>Road, gravel backfilling</td>
</tr>
</tbody>
</table>

**QUEST-C method**

The equipment used consisted of one submerged probe (Sigma 900max) for level and velocity measurements and two pumps (Velp, mod. SO311) used for dosing the tracers and two other pumps (Watson & Marlow, mod. SCIQ 323) used for taking the wastewater samples. The ion concentrations were determined by means of a
Dionex Dx100 (anion column AS14 and cation column CS12). The energy during the field activities was supplied by two generators.

Before the application of the QUEST-C method, a general uncertainty analysis was carried out in order to plan accurately the experiments and for investigating if the tests were feasible with the proposed model by means of those equipments and in that experimental area.

Three different reaches were investigated in the Torraccia catchment. In Table 5, the experiment settings and the exfiltration ratio percentage computed by the average concentration of the tracers are summarized.

Table 5 : QUEST-C experiment setting and exfiltration ratio percentage in the investigated reaches.

<table>
<thead>
<tr>
<th></th>
<th>Reach 1</th>
<th>Reach 2</th>
<th>Reach 3</th>
</tr>
</thead>
<tbody>
<tr>
<td>cLi [gr/L]</td>
<td>18.04</td>
<td>20.10</td>
<td>24.22</td>
</tr>
<tr>
<td>cBr [gr/L]</td>
<td>50.35</td>
<td>58.86</td>
<td>50.98</td>
</tr>
<tr>
<td>qLi [mL/min]</td>
<td>62.61</td>
<td>45.43</td>
<td>28.44</td>
</tr>
<tr>
<td>qBr [mL/min]</td>
<td>62.91</td>
<td>36.32</td>
<td>23.75</td>
</tr>
<tr>
<td>Exfiltration [%]</td>
<td>-8.69 ± 0.62</td>
<td>-1.73 ± 4.69</td>
<td>20.80 ± 4.46</td>
</tr>
</tbody>
</table>

4.3.2 Infiltration

In the Infernetto catchment, both isotopic and pollutant time series methods were applied, whereas in Torraccia only the isotopic method was used.

Isotopic method

The equipment used consisted of one submerged probe (Sigma 900max) for level and velocity measurements and one automatic sampler (SIGMA 900MAX) using for taking wastewater sampler from the sewer stream. All water samples were analyzed in laboratory for the δ18O abundance determination after Epstein and Mayeda (1953) on a Finnigan MAT mass-spectrometer. The wastewater samples were filtered using Whatman filters with a porosity of 0.45 µm.

The isotopic method applied in Rome was based on a two component hydrograph separation method; the two components (i.e. the sources of water drained by the investigated sewer systems) were identified as groundwater and drinking water. No springs or other sources were found by hydrogeological surveys in the investigated areas.

In order to quantify the uncertainty in the application of the isotopic method in Rome, an accurate error analysis was carried out by identification and quantification the principal sources of error.

Results from experimental surveys show that the infiltration ratio changes during the 24 hours experiments: it increases during the night and decreases during the day. The daily average infiltration percentage in Torraccia catchment has been estimated of 14.20 % ± 1.80 %, with precision ± 21 %. The daily average infiltration percentage in Infernetto catchment has been estimated to 49.90 % ± 11.90 % with precision ± 20 %.

Pollutant time series method

The equipment used consisted of one submerged probe (Sigma 900max) for level and velocity measurements and one on-line spectroliser s::can. The measured data were stored into a computer (con::vert, s::can). The wastewater samples were analyzed for laboratory calibration with the Merck kits and the spectroliser.

Preliminary campaigns in Infernetto catchment were carried out from 24 March to 15 May 2004. COD and TSS concentrations were measured in order to check possible troubles or drift and to identify the best location of the probe and the period of cleaning.

Then a calibration campaign was carried out from 10 October to 20 November 2004. Infiltration ratio and infiltration rate calculated by the experimental calibration curve are summarised in Table 6.

Table 6 : Infiltration ratio [-] and rate [L s⁻¹] calculated in Infernetto.

<table>
<thead>
<tr>
<th></th>
<th>Min</th>
<th>Mean</th>
<th>Max</th>
</tr>
</thead>
<tbody>
<tr>
<td>Infiltration ratio [-]</td>
<td>0.3629</td>
<td>0.5227</td>
<td>0.7198</td>
</tr>
<tr>
<td>Infiltration rate [L s⁻¹]</td>
<td>58.62</td>
<td>84.42</td>
<td>116.26</td>
</tr>
</tbody>
</table>
4.4 APPLICATIONS IN THE UK

4.4.1 Exfiltration

Field testing of the QUEST and QUEST-C methods (see Section 3.1) has been undertaken in collaboration with Thames Water PLC on a 660 m length of sewer in Potters Bar, S Hertfordshire on the outskirts of Greater London. The final 150 m of this sewer run had been recently replaced and was therefore ideal for use as the watertight reference section. The field tests employed three types of tracer (Rhodamine WT, lithium and bromide) with the use of the Rhodamine WT dye for the pulse dosing experiments providing a complement to other APUSS exfiltration studies which primarily used sodium chloride. A high level of temporal resolution is possible using this QUEST pulse dosing technique utilising a continuously monitoring in-sewer fluorimeter. Turbidity interference can be limited by comparing reference and indicator peaks which are as close as possible to each other without actually overlapping (Figure 12). Exfiltration rates calculated from tracer loadings rather than concentrations were preferred in order to eliminate the effect of flow variations on peak size.

A ratio of 2.5 between the Rhodamine WT tracer loadings for the indicator and the reference dosings was found to be appropriate for the field experiments. The simultaneous injection of multi-tracers under QUEST-C continuous dosing conditions facilitates realistic calculations of exfiltration rates for sewer travel times of about 20 minutes and these were determined as being typically less than 1.5 % for the test reach under investigation. Comparisons of the dosed amounts of Rhodamine WT and lithium with the loadings measured at the downstream monitoring station indicated nearly 100 % recoveries. This outcome for Rhodamine WT was surprising given the results of the batch laboratory testing (see Section 3.3.1) which suggested adsorption of the dye over extended time periods.

4.4.2 Infiltration

The application of the submersible UV-VIS spectrophotometer for the determination of COD time series has enabled the pollutant time series method to be successfully employed at the Thames Water sewage treatment works at Berkhamsted, W Hertfordshire to estimate sewer infiltration rates. This 8 km² mixed land use catchment (25 000 population) has a strongly reticulated combined sewer system comprised of circular ceramic pipes ranging in diameter from 0.15 m to 0.525 m with more than 60 % of the network being greater than 100 years old. The catchment suffers from severe infiltration with previously estimated average winter season infiltration rates of about 300 % based on the recorded flow time series. Figure 13 displays the graphical output for a 4 day period, as derived directly from the R-Script procedure, of the modelled discharge and the data fit COD time series, with the latter comprising the measured COD-equivalents (in black) and the calculated model COD times series (in red). The series provides a good reproduction of the measured data and satisfactorily explains the major elements of the wastewater pollutograph. The central diagram illustrates the constant baseflow separation and the diurnal wastewater variations. There is no evidence in the time series data for any
interflow component over the time period of this test run. The baseflow separation was calculated by fitting the modelled time series to the measurements by non-linear regression and the accumulated infiltration results are listed in Table 7. The infiltration ratio relates the volumes of infiltration to the total amount of wastewater discharged within the considered time span and the hydrograph separation predicts an infiltration rate of approximately 36 L/s under non-rainfall conditions. The methodology appears to be reasonably robust and enables a quantification of (non-polluted) infiltrating groundwater to be made based on the measured time series of wastewater flow and pollutant equivalent concentrations and without recourse to any additional external information.

![Graphs of Total Amount of Infiltration, Average Infiltration Ratio, Hydrograph Separation, and Data Fit](image)

Figure 13 : R-Script graphical output for infiltration at the Berkhamsted Sewage Treatment Works.

Table 7 : Accumulated Results (for 95% error band for baseflow separation).

<table>
<thead>
<tr>
<th>Time span</th>
<th>Σ Total infiltration</th>
<th>Infiltration ratio</th>
</tr>
</thead>
<tbody>
<tr>
<td>19.09 to 23.09</td>
<td>Min: 4.2 x 10^3 m³</td>
<td>Min: 14.3 %</td>
</tr>
<tr>
<td></td>
<td>Median: 10.9 x 10^3 m³</td>
<td>Median: 37.1 %</td>
</tr>
<tr>
<td></td>
<td>Mean: 11.1 x 10^3 m³</td>
<td>Mean: 37.4 %</td>
</tr>
<tr>
<td></td>
<td>Max: 18.5 x 10^3 m³</td>
<td>Max: 62.5 %</td>
</tr>
</tbody>
</table>
5. ASSOCIATED MODELS AND TOOLS

Measurements of infiltration and exfiltration rates in sewer pipes and house connections provide data and results which need to be integrated and adequately displayed in order to give end-users a clear representation and understanding of the phenomena affecting their sewer systems. Moreover, field data obtained by application of the methods developed in work areas 1 and 2 (see Chapters 3 and 4) shall be used in close relation to models, for calibration purposes and comparison of simulation results to field measurements. The complementary models and software tools developed in the work area 3 are briefly presented in this chapter: i) the similarity approach (section 5.1), ii) the leakage approach (section 5.2), iii) the AquaBase software package (section 5.3) and the large scale application example (section 1).

5.1 THE SIMILARITY APPROACH

5.1.1 The approach

Infiltration and exfiltration are basically driven by water head and leakage of the pipe. However, the knowledge about cause-effect relationships between independent pipe characteristics and dependent infiltration rates is fairly limited. There is a significant lack of data due to a high number of relevant processes and influencing factors (Davies et al., 2001) as well as due to the general data situation of large sewer systems. Thus, both deterministic (e.g. Dupasquier, 1999) and empirical models (e.g. Gustafsson et al., 1991) are difficult to apply and depend strongly on extensive measurements. Due to the high personnel and financial expenditures, an optimisation of measurement campaigns regarding cost efficiency and information yield is essential. Data-driven models were developed using the similarity approach “similar pipe characteristics lead to similar infiltration rates”, i.e. similarities in and between groups of reaches are used to improve the information yield and to transfer available information.

In order to verify the approach, statistical methods were applied on comprehensive data sets of the city of Dresden and of the cities of Bottrop and Gladbeck, all in Germany. The data sets contain in total 27 subcatchments with their network topology and a characterisation of the individual reaches by their attributes. For every subcatchment, the infiltration rate was estimated by measurements, while the corresponding groundwater table was known in some subcatchments. The characteristics of the subcatchments were compared with analysis-of-variance methods (Müller, 1991). With these methods it can be investigated whether several grouped samples, i.e. reach groups, belong to one basic population. Significant dissimilarities were detected between subcatchments and reach populations with different infiltration rates. Due to the wide range of observed infiltration rates (min÷max = 0.18÷2.95 L/[m².h] = 1:16), it seemed to be probable that the dissimilarities between the reach populations are linked in some way to the infiltration rates. The relationship between the reach attributes and the infiltration rates was analysed with multidimensional scaling (MDS) (Borg and Groenen, 1997). This method transforms efficiently a high dimensional arrangement of objects, so that a low-dimensional and interpretable configuration with the optimum approximation of the observed pattern is reached. Thus, any number of independent parameters can be represented by one or two figures. A relation between the measured infiltration rates and the MDS results of the Dresden data set without groundwater information could not be found. But, a relation between sewer attributes and the urban district type (e.g. inner centre, broader centre, suburb) was identified. Due to this link between city history and network development it can be concluded that in principle a larger network consists of identifiable homogenous areas. The MDS results of the Dresden data set with groundwater information are shown in Figure 14. The parameter set was reduced to one dimension dim. 1. A good correlation between dim. 1 and the reach-surface-specific infiltration rate \( q_f \) was found. A comparison with the time weighted head shows that this correlation, i.e. the specific infiltration rate, is not dominated by the groundwater head. The analysis of a combination of the Dresden and the Bottrop/Gladbeck data set led to similar results. Because of the reasonable correlation between dim. 1, standing for the reach attributes, and the infiltration rate, it can be concluded i) that there is a recognisable relationship between the independent parameters and the infiltration rate and ii) that the parameters - or a part of them - are sufficient to describe infiltration. Thus, the similarity approach is applicable.
5.1.2 Application 1: method to identify optimum location of measurements

An optimally defined measurement gauge is located at the outlet of a more or less homogeneous subcatchment. Thus, the reach-specific error of the measured infiltration rate is minimised and the information content gained is high. The developed method consists of two steps: i) describing the homogeneity of the upstream subcatchment with a similarity figure \( \phi \) for every potential placement and ii) determining the most homogenous subcatchments with an optimisation algorithm.

The calculation of the similarity figure was adapted from cluster analysis (Kaufman and Rousseeuw, 1990). The input is a data set containing all relevant reaches with their attributes. Due to the scale invariance of distance measures, the parameters must be standardised. The parameters can be weighted by means of their relevance for infiltration. The distance or dissimilarity between two reaches represented by their parameter vectors is calculated with the Euclidean and the Mahalanobis distance (Fahrmeir et al., 1996). The distances are aggregated on subcatchment level to determine one characteristic value with the mean distance between all reaches and the mean distance of all reaches to the groups centroid. These aggregations are based on the idea of a more or less equal distribution of extraneous water in sewers. To determine the optimal placement, an optimisation algorithm is run. With an iterative procedure, the catchment is divided until a given number of subcatchments, i.e. measurement gauges, is reached. The conditions for separating manholes are:

- the similarity figure \( \phi \) is minimal: a minimal \( \phi \) stands for a maximal possible homogeneity.
- sum of the size measures of relevant reaches > critical value: the size measure can be the sewer length, the reach surface, the connected area, connected inhabitants etc. This constraint results from measurement requirements like minimal flow or length.
- the manhole is not within a mesh: an infiltration rate measured in a mesh cannot be allocated to reaches.

The method was tested on six subcatchments. Reach related infiltration rates were modelled with a conceived infiltration model based on real independent parameters and a constant groundwater table on an arbitrary level. For every subcatchment, 50 random distributions of five gauges were generated and compared with the optimised positioning. The comparison shows an information improvement and error reduction, respectively, of in mean 14 % to maximum 35 %. The main reason for these results which seem to be mediocre is the fact that the procedure is not a classification of independent objects. The network topology, especially number and size of meshes, as well as the spatial distribution of attributes as a result of the building history, have a strong influence on the results. With a better knowledge about in- and exfiltration processes, the optimisation results might be higher. Furthermore, the method was tested in three smaller catchments with relatively detailed information about infiltration. The application of the method results in good coherence between the optimised arrangement of gauges and detected changes of measured infiltration rates.

The proposed optimisation method is not based on typical input/output functions. It compares and classifies states. Therefore, the results are to be considered within the boundaries conditions of the given data set. They
will always have a significant uncertainty. Information about groundwater has an overwhelming relevance for their quality. Advantageously, a definite parameter set is not necessary. The method can be adapted to nearly every data situation.

5.1.3 Application 2: integrated model to assign and merge infiltration rates of subcatchments

With models which allow to transfer the results of I/E measurements from subcatchments to other ones, it is possible to aggregate the available information on catchment scale. Two models based on statistical methods were developed.

The method based on analysis of variance treats subcatchments – represented by means and variances – as one coherent unit. It determines parameter similarities between subcatchments with unknown I/E class and subcatchments with known I/E class. In cases of high similarities, the I/E class is transferred. The parameters are analysed with post-hoc tests, e.g. the Fisher Least Significant Difference test (Winer et al., 1991), and multiple comparison tests (Siegel and Castellan, 1988) respectively. The number of not rejected tests (i.e. two subcatchments are determined as belonging to one population for the considered parameter) is summarised. Pairs of subcatchments with a high sum can be assumed as similar and should have a similar I/E class.

Figure 15 shows a matrix of the similarities between paired subcatchments with modelled infiltration classes. As five parameters were used for the infiltration model, five is the maximum number of similarities. Due to their distribution, the minimum number of similarities to be investigated was set to three or 60 %. From the 15 % of pairs identified to be similar, only 50 % were correctly assigned (crosses in Figure 15), i.e. were similar with regard to all 5 parameters.

The transfer-method based discriminating analysis (Fahrmeir et al., 1996) considers single reaches. The method assigns the reaches of a subcatchment with unknown I/E class to subcatchments with known I/E class based on classification probabilities. The resulting I/E class of the investigated subcatchment is calculated as a weighted mean.

The results of the above mentioned example lead to an assignment rate of 50 % assuming a suitable assignment error of 0.66 class width (equivalent to one third of the range). Assuming a softer error of 1.0 class width, 75 % of the subcatchments were correctly assigned. Furthermore, the method was applied to the Dresden data set with groundwater information. Assuming a suitable assignment error of 0.66 class width, 1 of the 5 subcatchments was correctly assigned. Assuming a softer error of 1.0 class width, 4 of 5 subcatchments were correctly assigned.

With the transfer methods, it is possible to improve the information about the I/E status of complete sewer networks. With a suitable error defined as acceptable the transfer of infiltration classes was successful for approximately half of the subcatchments. Keeping in mind that only few data were available, the positive verification of the methods is indicated but not completely satisfactory.
5.2 THE LEAKAGE APPROACH

5.2.1 Background

The leakage approach can be traced back to the Darcy equation. It is widely used for modelling hydrological interactions between aquifer and surface water and is related to the wetted area of the river bed, the difference between groundwater and surface water level, and a specific leakage factor representing the ability of the reach to infiltrate groundwater or exfiltrate river water. As an integrative parameter the leakage factor describes various attributes of the soil layer of the river bed and thereby the potential of exchange between the compartments ground- and surface water. The approach is applicable both for in- and exfiltration.

According to Gustafsson (2000) and Karpf and Krebs (2004a) the leakage approach can be modified for the simulation of groundwater infiltration into sewer systems:

\[ Q_{\text{infiltration}} = k_L A_S (h_G - h_S) \]  

**requirement:** \( h_G > h_S \)

where:

- \( Q_{\text{infiltration}} \): infiltration of groundwater (m\(^3\)/s)
- \( A_S \): groundwater-influenced pipe surface (m\(^2\))
- \( h_S \): water level in sewer pipes (m)
- \( h_G \): groundwater level (m)
- \( k_L \): leakage factor (s\(^{-1}\)).

An extended model, which takes into consideration regional differences and house connections was proposed for further development and implementation:

\[ Q_{\text{in}} = (h_{GWL} - h_w) P_{w1} L K_I K_R + q_{0\text{in}} + q_{in\text{HC}} N_{HC} \]  

**requirement:** \( h_{GWL} > h_w \)

where:

- \( Q_{\text{in}} \): infiltration flow (m\(^3\)/d)
- \( h_w \): water level in the pipe (m)
- \( h_{GWL} \): groundwater level around the pipe (m)
- \( P_{w1} \): external wet perimeter (m)
- \( L \): length of the pipe (m)
- \( K_I \): local coefficient (d\(^{-1}\))
- \( K_R \): regional coefficient (-)
- \( q_{0\text{in}} \): infiltration flow from other infiltration sources (m\(^3\)/d)
- \( q_{in\text{HC}} \): mean infiltration flow for a single house connection (m\(^3\)/d)
- \( N_{HC} \): number of house connections to the pipe.

The achievable spatio-temporal resolution of the approach is relatively high but depends on the data situation. It is, at least theoretically, possible to determine infiltration at single pipe level and at daily scale. But, only little information about processes can be expected. The quality of modelling and predicting infiltration rates is high for larger catchments (Karpf and Krebs, 2004b).

5.2.2 Application

**Data needs**

The data needs for the application are listed in Table 8.

For the calibration of the model coefficients, time series of groundwater levels and infiltration rates \( Q_{\text{in}} \) at the end of the catchment, i.e. at the WWTP, must be available. To cover various groundwater conditions, a long time period and a high temporal resolution of data is necessary. Monthly values over a period of several years are recommended.
The infiltration rates can be balanced by calculating the difference of wastewater flow and the average consumption of drinking water. Due to uncertainties associated to balanced drainage rates (Karpf and Krebs, 2003), the variation of infiltration rates may be smoothened.

Table 8: Data needs for the leakage model approach.

<table>
<thead>
<tr>
<th>Type</th>
<th>Data</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>structural data</td>
<td>L</td>
<td>The structural data are contained in the sewer data base. Profile shape, diameter and slope are necessary to calculate PW1.</td>
</tr>
<tr>
<td></td>
<td>N_HC</td>
<td></td>
</tr>
<tr>
<td></td>
<td>profile shape</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Diameter</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Slope</td>
<td></td>
</tr>
<tr>
<td>water levels</td>
<td>h_w</td>
<td>The water level in the sewer pipe can be modelled. However, it is also feasible to estimate it by measurements of water levels in the system at known conditions.</td>
</tr>
<tr>
<td></td>
<td>h_GWL</td>
<td>The estimation of groundwater level at each pipe is based on the interpolation of groundwater measurements.</td>
</tr>
<tr>
<td>coefficients</td>
<td>K_i</td>
<td>The coefficients have to be calibrated.</td>
</tr>
<tr>
<td></td>
<td>K_r</td>
<td></td>
</tr>
<tr>
<td>specific infiltration rates</td>
<td>Q_i,in</td>
<td>The constants must be determined by separate investigations (e.g. by balancing). At least ( q_{in\ HC} ) seems to be not essential for a successful application.</td>
</tr>
<tr>
<td></td>
<td>Q_i,GHC</td>
<td></td>
</tr>
</tbody>
</table>

Calibration

The regional coefficient \( K_r \) is determined for a group of pipes according to field measurements representing regional peculiarities. Unless external inputs from measurements are available, it should be set to a default value equal to 1.

The calibration of the local coefficient \( K_i \) is based on the following equation:

\[
K_{i,T} = \frac{Q_{i,in,T}}{\sum_{i=1}^{g} [ h_{GWL,i,T} - h_{w,i,T} ] \cdot A_{i,T}} \quad \text{eq. 13}
\]

requirement: for all reaches with \( h_{GWL,T} > h_{w,T} \)

where:

\( K_{i,T} \) integral leakage factor at time \( T \) (d\(^{-1}\))
\( Q_{i,in,T} \) balanced infiltration in the catchment at time \( T \), without \( Q_{ind\ HC} \) and \( Q_{in} \) (m\(^3\)/d)
\( h_{w,i,T} \) water level in the sewer pipe \( i \) at time \( T \) (m)
\( h_{GWL,i,T} \) groundwater level at the sewer pipe \( i \) at time \( T \) (m)
\( A_{i,T} \) groundwater-influenced pipe surface of pipe \( i \) at time \( T \) (m\(^2\)).

The calculated leakage factor represents an integral parameter for all groundwater-influenced pipes at a certain time \( T \). In order to refer individual leakage factors to pipes, the calculation has to be carried out for a number of time spots. Thereby, the leakage factor of each groundwater-influenced pipe can be estimated to a weighted average of all calibration cases:

\[
k_{i,T} = \frac{\sum_{i} K_{i,T}}{n_{i}} \quad \text{eq. 14}
\]

requirement: equidistant time steps

where:

\( K_{i,T} \) calibrated leakage factor for pipe \( i \)
\( K_{i,T} \) integral leakage factor at time \( T \)
\( T_{i} \) time spot when \( h_{GWL,T} > h_{w,T} \)
\( n_{i} \) number of time spots when \( h_{GWL,T} > h_{w,T} \).

A simplified example for the calibration procedure is shown in Figure 16. For every time \( T \) (March and October) one leakage factor was calculated. The weighting for every pipe is done by averaging these \( K_{i,T} \)-values. Thus, the calibrated leakage factor of pipe 1, which is not groundwater-influenced in March, is equal to \( K_{1,\text{October}} \), the factor of pipe 2 is equal to the average of \( K_{2,\text{October}} \) and \( K_{2,\text{March}} \).
5.3 THE AQUABASE PACKAGE

The development of computerised data based management tool provides additional value through the provision of relevant data directory and visual display menus valuable to end users. The development of the conceptual model (see Section 5.2) and the integration approach result in software development. The software allows the estimation of I/E in sewers and integration over the catchment scale. This database software holds all necessary data of a particular catchment and it is able to apply main APUSS project results to sewer system described by mentioned data as well.

Figure 16: Example for model calibration.

Figure 17: The AquaBase environment.
It was decided to make use of the existing software tool AquaBase with inner structure corresponding to all demands on sewer system data description and with possibility of additional development in order to implement new features defined in the APUSS project (Figure 17).

The AquaBase program represents a very useful tool for the easy to use asset data management in water utility companies. The AquaBase functions in principle as an urban drainage database with GIS functionalities, user friendly interface and number of specialised functions related to the sewerage system data management. The tool is developed as a client-server architecture based on WinBase database engine. There are no particular hardware requirements related to the use of this application.

The AquaBase has been developed in a course of several years and the current version profits from numerous practical comments and suggestions from the side of water utility specialists. The tool is very flexible and gives the user very profound insight into the asset (or model) data he/she is manipulating. Specialised queries can detect random as well as systematic errors in the data and propose the way of correcting these data. Very profound system of flagging and selections makes the user possible to work with subsets of the whole data set and to focus areas of his/her priority. Very fast graphics will be appreciated by standard GIS users.

The AquaBase was chosen as a user friendly software tool, which is enabled to incorporate field data as well as develop internal tool and routines requested for the APUSS project. The AquaBase data structure has been extended to be able to consider new data inputs, variables and coefficients used for the calculation of the infiltration and exfiltration rates. System was modified to support the graphical presentation (using colour thematic maps) of the input and result data, connected to pipes and polygons (catchments).

5.3.1 The models implemented

Exfiltration is measured at the sewer reach scale, while infiltration is measured at subcatchment scale. Accordingly, conceptual models respectively at sewer reach and subcatchment scales for exfiltration and infiltration have been proposed. The involved key state variables are water levels in pipes $h_W$, surrounding groundwater levels $h_{GWL}$ and I/E wetted perimeters $P_W$ (Figure 18). The model parameters at reach and catchment scales reflect the structural state of the sewer pipes and need to be calibrated by means of field experiments and results (see e.g. Section 5.2).

![Key state variables used in the I/E models.](Figure 18)

5.3.2 The package functionalities

An integrated tool has been developed in the frame of the AquaBase software, which displays the map of the sewer system and the dry weather flow rates, the interpolated groundwater levels, and which includes data and time series containing experimental I/E results. Automatic calibration routines have also been developed in order to calibrate the models’ parameters according to the data and time series. Models and tools have been established and tested in the frame of a large scale example of general application of methods and models, with an end-user application perspective (see also Section 5.4).

The following main features have been implemented into AquaBase:
- development of interface for input variables
- implementation of ground water depth computation
- implementation of DWF water level computation
- elaboration of a communication centre for separate tools (I/E data processing)
- implementation of infiltration and exfiltration models
- calibration of infiltration and exfiltration models.
- performance indicators calculation, analyses and links to other PI platforms (Figure 19)
- uncertainty analysis of obtained results (Figure 19).

A couple of multi-parametric formulas describing infiltration and exfiltration process in sewer pipe is implemented in the AquaBase, giving the user the possibility to calculate the rough estimates based on sensitivity analysis on distinct parameters. IE can be then calibrated based on new methods of measurements to reach better accuracy of the final estimate. In order to cope with the uncertainty of the interested phenomena, an external module of AquaBase was designed and implemented, based on the Monte Carlo method. This module develops a large spectrum of potential results which are then statistically post processed and the uncertainty in the selected range of confidence interval defined. This functionality brings an extra value to the AquaBase and the I/E calculation, and propagates also to the values of performance indicators (PI), which are generated at the end of the calculation. PI can be then used by practitioners for the planning of reconstruction and investment in the urban drainage system (see Sections 6.1 and 6.2).

The AquaBase with the I/E extension is developed as a clear product of the APUSS project. As such it can be freely used by both scientists as well as practitioners in urban drainage domain. It is expected that this tool can be found useful for water utility companies in terms of I/E assessment, WWTP functions as well as sewer system reconstruction. The AquaBase is delivered on installation CD, and is accessible through the APUSS website.

5.4 LARGE SCALE TEST APPLICATION

A large scale area application covered the final testing of the APUSS methodology. The Prague sewer system catchment was selected to perform this task.

5.4.1 Data collection

The main source for data was the Sewer System Master Plan of Prague processed and elaborated in 1999-2001. Number of activities that should be performed by APUSS data collectors has been already processed during that
Despite this, it is necessary to stress the data collection was the “uphill struggle” to carry out, especially from the point of human resources. To describe the selected system, following necessary data had to be collected:

- sewer system topology data,
- data of catchments,
- ground water level maps,
- monitoring campaign data.

**Sewer system topology** model covers main trunk sewer system only. However, basic sewer system data consist of more than 7000 nodes with a pipe length of approximately 190 km.

Each node is described by set of information:
- X and Y coordinates of node centre,
- bottom level,
- level of terrain,
- type of node (manhole, outlet, object – CSO, pump).

Each pipe is described by a set of data:
- upstream (from) and downstream (to) node,
- material (to recognise roughness),
- shape (circle, egg …),
- level of downstream connection.

**Catchment data**: the overall drained area exceeding 200 000 hectares was divided into almost 1000 catchments described in detail.

Each catchment is described by a set of information:
- node to which the catchment is connected,
- catchment area,
- hydraulic length,
- slope,
- number of connected inhabitants,
- percentage of impervious area.

**Ground water level maps** were digitized in detail. The result were more than 10 Gbytes data in over 120 files, i.e. approximately 200 000 points (Figure 20). Such amount of XYZ point was too much to handle. Therefore, the data was diluted to grid 100 x 100 m by a program specifically developed to perform only this task. With the resulting approximately 50 000 points, it was possible to work.

Each ground water level point is defined by:
- node of artificial type,
- water level value in node time series table.

**Monitoring campaign.** In the scope of the Prague Urban Drainage Master Plan, the monitoring campaign was divided into 3 phases in a period of time. Overall amount of monitoring profiles approaches almost 100. Finally, for APUSS implementation, 45 calibration profiles monitoring discharge have been selected (see left part of Figure 21). In each calibration profile, there were several discharge time series measured. All the discharge time series have been elaborated in order to find relevant minimum discharge value.

Each calibration profile is defined by:
- pipe of monitoring profile location,
- I/E calibration discharge value.
5.4.2 I/E computation

Processing of the APUSS features implemented in AquaBase has been carried out according to elaborated training notes. All tasks were performed according to released procedure. Huge data set, however, called for performance and logical implementation changes in AquaBase. After changes were implemented, all calculation worked securely.

Figure 21 - Thematic maps of sewer system (left = I/E groups definition, right = flood by ground water level).

The changes were caused by:
- ground water level computation. Huge number of water level points caused the software was not able to process the computation. A number of 50 000 was finally enough to describe water levels on the area of Prague and the software showed proper performance in calculation process.
- I/E group definition. From 45 calibration profiles, 45 I/E groups had to be created. This was probably the main problem in data elaboration. The sewer system is linked and switched through and it was necessary to find proper location of group borders in order not to get system looped. Auto-calibration process of In/Exfiltration calculation was not stable and fast enough. After implementation of changes in logical structure of data the calculation performance increased.

Data collection and elaboration for purposes of I/E modelling in a large scale is time consuming. Moreover, calibration of I/E is not easy and fast process despite developed enhanced functions and robust hardware. Users have to be on one side advanced enough to understand computation background and meaning of the results values (engineering point of view) and, on the other side, users have to be advanced enough to know how, where and what to change in software settings to get expected results.
6. ECONOMIC AND OPERATIONAL ASPECTS

Municipalities and/or sewer operators have to decide investment strategies to rehabilitate and upgrade efficiency and quality of their sewer systems. Frequently, such decisions are based on limited information and do not consider simultaneously the sewer pipes, the wastewater treatment plant and the environment, especially groundwater. In order to provide some elements to help operators in making decisions, work area 4 included three topics: i) definition of a set of I/E specific performance indicators (section 6.1), ii) proposition of a generic multi-criteria methodology to compare investment strategies (section 6.2) and iii) analysis of costs structure of sewer systems and costs of the new methods (section 6.3). Detailed reports and publications are available on the APUSS website.

6.1 I/E PERFORMANCE INDICATORS

6.1.1 Objectives

A set of specific performance indicators (PI) has been developed to assess the impact of I/E on sewer systems and applied to three cases studies. PI are a means of aggregating information on system characteristics and experimental data from monitoring or modelling, and translate it into performance values. PI have been calculated from sewer systems characteristics and I/E rates measured with the methods developed in work area 1 (see Chapters 3 and 4). Once calculated, PI can be classified in relation to good or bad performance, which allows a standardized and objective comparison of the performance of different sewer systems and constitutes a means to technically support the establishment of priorities for rehabilitation and/or construction investments, taking into account I/E impacts (Cardoso et al., 2002, 2005). Links have been established between the AquaBase platform and the PI calculation software (see Section 5.3).

6.1.2 Methodology

The principal requirements considered for the performance assessment methodology are: i) flexibility, in order to accommodate with ease the different sensitivities, interpretations or objectives of the analysis; ii) a certain degree of standardisation in order to facilitate a multi-disciplinary approach, where the various aspects to be considered may be brought down to the same quantified basis; and iii) a quantitative, numerical base - the envisaged tool should be translatable computationally in order to afford intensive use, either from within or as post-processor to the current modelling techniques or monitoring data (Cardoso et al., 1999).

The methodology consists in the selection and development of three components for each aspect of performance analysed (Cardoso and Coelho, 2004): i) the numerical value of a sewer network property or state variable, which is expressive of the particular aspect being scrutinized (I/E), i.e. the Performance Indicator (PI) definition; ii) a classification of the PI values scoring them in relation to good or bad performance; iii) an operator, which allows the performance values at element level to be aggregated across the system or parts of it to obtain an overall system performance figure. The performance assessment scale depends on the scale of available data. In cases where data are available at element level, there are i) a global value that is achieved through a particular operator in order to represent the performance assessment of the network, e.g. a simple or weighted average, and conversely ii) a population of elementary values, which leads itself to a basic statistical treatment, giving information on the PI values dispersion on the system through, e.g., percentile bands. In cases where data are only available at catchment level, only a global PI value for that catchment is achieved.

6.1.3 Performance Indicators for infiltration and exfiltration

Seven PI for infiltration were established for the assessment of impacts of infiltration in sewer systems (Cardoso and Coelho, 2004) and are presented in Table 9. Five PI for exfiltration were established for performance assessment of impacts of exfiltration in sewer systems (Cardoso and Coelho, 2004) and are presented in Table 10.

These PI were applied to four experimental catchments that have been studied in the project: two in Italy, Rome (Infernetto and Torraccia) and two in France, near Lyon (Yzeron and Ecully). Torraccia is about 85 ha, whose 55 ha are residential and 30 ha are rural. The sewer network is an egg-shaped combined system (1x1.2 m, 1.5x1.8 m and 1.5x2.1 m) laid about 4 to 9 m under the ground level. It is dated 14 years and pipe material is concrete. Infernetto is about 5.5 km² and it is characterized by residential houses with private gardens. The groundwater
level changes between 6 to 7 m under the ground in the north-east part and 1.5 m under the ground in the south-west one. The system is separated and sewer cross sections are circular (0.3 m, 0.6 m, 0.8 m and 1 m) laid about at 2 to 6 m under the ground level. It is dated about 19 years and pipe material is concrete (size 1.0 m), ceramic (sizes 0.6 m and 0.8 m) and PVC (size 0.3 m). Infernetto and Torraccia have been chosen because they have very different geological, hydrogeological and sewer characteristics and can be considered representative of almost all urban areas in Rome. Infiltration and exfiltration rates have been assessed by the application of the APUSS methods.

The Yzeron catchment has an area of 150 km$^2$ with increasing urbanization and imperviousness since 20 years. Its drainage system is combined, with a total length of 294 km and sewer sizes ranging from 200 mm circular pipes to 2 m egg-shaped pipes. In 1992, a diagnostic study revealed high infiltration rates linked to a degraded state of the sewer system. The Ecullly catchment is smaller, with an area of 245 ha. The habitat is mostly residential. The imperviousness is evaluated to 46 % (De Bénédictis, 2004). Its drainage system is combined, with a total length of 15 km, a mean slope of 0.027 m/m, and sewer sizes ranging from 200 mm circular pipes to 1.8 m egg-shaped pipes. Infiltration and exfiltration rates have been measured using two of the APUSS methods.

### Table 9: Infiltration performance indicators.

<table>
<thead>
<tr>
<th>Definition</th>
<th>Unit</th>
<th>Designation</th>
<th>Concept/Explanation</th>
</tr>
</thead>
<tbody>
<tr>
<td>$\frac{Q_{\text{inf}}}{Q_{\text{full}}}$ (%)</td>
<td>PI$_{11}$ Infiltration full capacity utilization</td>
<td>Proportion of the sewer full section flow capacity used by the infiltration flow. This indicator supplies information on the hydraulic performance of the analysed sewer(s), but gives no indication about the net infiltration flow. White et al. (1997) refer that, for design purposes, in UK 10% of sewer capacity should be allocated to infiltration in separate domestic systems according to the Water Authorities Association (WAA, 1989).</td>
<td></td>
</tr>
<tr>
<td>$\frac{Q_{\text{inf}}}{Q_{\text{avdwf}}}$ (%)</td>
<td>PI$_{12}$ Infiltration proportion of dry weather flow</td>
<td>Infiltration flow expressed as a percentage of the daily mean dry weather flow. A drawback of this indicator is its dependency on the values of the dry weather flow. This indicator gives an estimate of the weight of infiltration flow on treatment costs. It may be expressed either as percentage of volume or cost. Stevens (1998) presents a study in New York where about 50% of the total flow reaching WWTP was infiltration flow; Belhadj et al. (1995) report values of 42% of dry weather flow.</td>
<td></td>
</tr>
<tr>
<td>$\frac{Q_{\text{inf}}}{Q_{\text{avdwf}} - Q_{\text{inf}}}$ (%)</td>
<td>PI$_{13}$ Infiltration proportion of domestic wastewater</td>
<td>Infiltration flow expressed as a percentage of the daily mean domestic wastewater. A drawback of this indicator is its dependency on the values of the dry weather flow. The German guidelines ATV-A118E (ATV, 1999) considers an infiltration/inflow flow equal to 100% of domestic flow, for design purposes of separate domestic systems.</td>
<td></td>
</tr>
<tr>
<td>$\frac{Q_{\text{inf}}}{\text{sewer longitudinal wall area}}$ (m$^3$/day/(cm.km))</td>
<td>PI$_{14}$ Infiltration flow per unit sewer wall area</td>
<td>Infiltration flow per unit area of sewer wall. This indicator is related to the sewer wall area that is potentially subject to infiltration. It can be the wall area below groundwater level. In this case it was considered the total sewer wall area. EPA (2001) refers to that in USA this value can vary from 0.05 to 1.39 m$^3$/day/(cm.km). Tchobanoglous et al. (2003) refer that these values can range from 0.1 to 10 m$^3$/day/(cm.km).</td>
<td></td>
</tr>
<tr>
<td>$\frac{Q_{\text{inf}}}{\text{sewer length}}$ (m$^3$/s/km)</td>
<td>PI$_{15}$ Infiltration flow per unit sewer length</td>
<td>Mean infiltration flow per unit length of sewer. This indicator is related to the sewer length that is potentially subject to infiltration. This indicator is relevant in systems where infiltration takes place predominantly along the sewers.</td>
<td></td>
</tr>
<tr>
<td>$\frac{Q_{\text{inf}}}{n^{#}\text{manholes}}$ (m$^3$/s)</td>
<td>PI$_{16}$ Infiltration flow per manhole unit. Manholes are an important source of infiltration. This indicator gives an idea of the influence of the number of manholes on total infiltration.</td>
<td>Mean infiltration flow per manhole unit. Manholes are an important source of infiltration. This indicator gives an idea of the influence of the number of manholes on total infiltration.</td>
<td></td>
</tr>
<tr>
<td>$\frac{Q_{\text{inf}}}{n^{#}\text{service connections}}$ (m$^3$/s)</td>
<td>PI$_{17}$ Infiltration flow per service connection</td>
<td>Mean infiltration flow per service connection. Service connections are an important source of infiltration. This indicator gives an idea of the influence of the number of service connections on total infiltration. Ellis (2001) refers that in UK house connections contribution represents 30% to 40% infiltration in sewers.</td>
<td></td>
</tr>
</tbody>
</table>
Table 10: Exfiltration performance indicators.

<table>
<thead>
<tr>
<th>Definition</th>
<th>Unit</th>
<th>Designation</th>
<th>Concept/Explanation</th>
</tr>
</thead>
<tbody>
<tr>
<td>$\frac{Q_{\text{exf}}}{Q_{\text{avdwf}}}$</td>
<td>(%)</td>
<td>$\text{PI}_{E1}$</td>
<td>Exfiltration proportion of dry weather flow. Exfiltration flow expressed as a percentage of the daily mean dry weather flow. A drawback of this indicator is its dependency on the values of the dry weather flow. Amick and Burgess (2000) present values of experimental studies with values between 11.9% and 49%. Ellis et al. (2002) refer that probably this value is not greater than 10%.</td>
</tr>
<tr>
<td>$\frac{Q_{\text{exf}}}{\text{sewer length}}$</td>
<td>(m³/s/km)</td>
<td>$\text{PI}_{E2}$</td>
<td>Mean exfiltration flow per unit sewer length. This indicator will give relevant results in systems where exfiltration takes place predominantly along the sewers. Literature values are between 0.000027 and 0.001 m³/day/m (Amick and Burgess, 2000).</td>
</tr>
<tr>
<td>$\frac{Q_{\text{exf}}}{\text{sewer longitudinal wall area}}$</td>
<td>(m³/day/(cm.km))</td>
<td>$\text{PI}_{E3}$</td>
<td>Mean exfiltration flow per unit area of sewer wall. This indicator is related with the sewer wall area that is potentially subject to exfiltration. Literature values are between 0.08 and 1.20 (Amick and Burgess, 2000).</td>
</tr>
<tr>
<td>$\frac{Q_{\text{exf}}}{\text{ nº manholes}}$</td>
<td>(m³/s)</td>
<td>$\text{PI}_{E4}$</td>
<td>Mean exfiltration flow per manhole unit. Manholes are an important source of exfiltration. This indicator gives an idea of the influence of the number of manholes on total exfiltration.</td>
</tr>
<tr>
<td>$\frac{Q_{\text{exf}}}{\text{ nº service connections}}$</td>
<td>(m³/s)</td>
<td>$\text{PI}_{E5}$</td>
<td>Mean exfiltration flow per service connection. Service connections are an important source of exfiltration. This indicator gives an idea of the influence of the number of service connections on total exfiltration.</td>
</tr>
</tbody>
</table>

6.1.4 Results of Performance Indicators for infiltration and exfiltration

Some results obtained for the application of infiltration and exfiltration PI to the case studies described, are presented in Figure 22 to Figure 25. In these figures the performance of the different systems can be compared in the same graph (Cardoso et al., 2004, 2005). In the case of the Yzeron catchment, infiltration PI were calculated for 8 subcatchments and also for the entire catchment as a simple mean. There was no data on exfiltration. Flow data used for calculation was the average daily infiltration or exfiltration flow and the average daily dry weather flow from the APUSS experimental measuring campaigns.

From the results it is possible to conclude that, for the global Ecully catchment, the contribution of infiltration is very high representing about 40% of the dry weather flow that will reach WWTP (PII₂). This fact has no impacts with regard to the hydraulic capacity of the system as PI₁ presents a value of 0.3 %. This means that the main impacts are in terms of operational costs and pollutant that will be transferred to the WWTP or discharged into the receiving water. For the global Yzeron catchment, the contribution of infiltration represents 55 % of the dry weather flow that will reach WWTP (PII₂). There are impacts with regard to the hydraulic capacity of the system as PI₁ presents a value of 13 %, but not very significant. This means that there are big impacts in terms of operational costs, as the wastewater volume to be transported and treated in the system, due to infiltration flow, is twice the dry weather flow. This reflects the fact that the Yzeron sewers are in a bad structural state as detected some years ago by the Greater Lyon Sewer Department, and rehabilitation projects are under preparation at the moment. For the global Torraccia catchment, the contribution of infiltration represents about 15 % of the dry weather flow that will reach WWTP (PII₂). There are no impacts with regard to the hydraulic capacity of the system as PI₁ presents a value of 0.06 %. This means that infiltration has no relevant impacts in the systems’ performance. For the Infernetto catchment, the contribution of infiltration represents about 48 % of the dry weather flow that will reach WWTP (PII₂). This means that there are impacts in terms of operational costs. Comparing now the performance of the different catchments/subcatchments, it is clear that Yzeron2 shows significant problems as a consequence of infiltration. In general PI for Yzeron2 are 22 times higher than for the Ecully and Torraccia. In spite of PI₂ is 88 %, the value for PI₁ is 8 %. This means that this catchment presents severe infiltration problems and that impacts are higher in terms of costs. It seems that there are no hydraulic problems. On contrary, Yzeron4 presents the values PI₁ of 45 % and PI₂ of 59 %. In this case, besides the impact on costs, there are also problems with hydraulic capacity. In general, Yzeron catchment presents significantly higher values for the PI than Ecully and Torraccia. Yzeron seems to have bigger problems related to infiltration occurrence than Ecully. Torraccia presents the lowest PI values and seems to have no problems related to infiltration while Infernetto seems to show impacts of infiltration, for the assessed PI.
Regarding now exfiltration, Ecully presents lower PI values than the other systems, with exception of PI E1 that represents 10% of dry weather flow lost, the highest value. This means that the unit rate of exfiltration along the sewers is not very high. However, if the sewer is long or the dry weather flow is low, this percentage increases. The average PI value of Torraccia presents higher values than Ecully, with exception for PI E1. In Torraccia this is about 5% meaning that this percentage of dry weather flow is lost, half of Ecully value. In Torraccia there is a significant difference between Reach1 and Reach2. PI for Reach1 are the highest, with exception of PI E1, meaning that this reach shows clearly problems of exfiltration. At Infernetto PI E1 and PI E2 show a behaviour similar to that of Ecully. Comparing now the performance of the different catchments/subcatchments, it is clear that Torraccia Reach1 has more problems regarding exfiltration impacts and thus should have rehabilitation priority.

6.1.5 Conclusions on using PI

The above methodology aims to support sewer systems rehabilitation by using infiltration and exfiltration performance indicators (PI), as a means of aggregating information on system characteristics and experimental data from monitoring or modelling, and translate it into performance values. The application of PI for infiltration and exfiltration depends on available data, which means that on one hand the approach can be applied at pipe or larger scales; on the other hand the quality and uncertainty of PI results depend on the quality an uncertainty of data used. Any assessment using PI must consider a set of PI and not only one, in order to give a global view with significant information and taking into account the different points of view that affect systems performance.

6.2 GENERIC METHODOLOGY TO COMPARE INVESTMENT STRATEGIES

When I/E problems are detected and evaluated, end-users usually have various investment and rehabilitation strategies to solve them (rehabilitation of the sewer system, adaptation of the downstream wastewater treatment plant (WWTP), storage tanks, or any combination of the above solutions). In most frequent cases, the comparison of the possible strategies is based on a traditional cost-benefit approach, which consists to minimise the costs of investment and operation for an expected given benefit. Two of the main limitations in such an approach are: i) the difficulty to evaluate all criteria in monetary units (e.g. what is the cost in Euros of any kg of...
COD discharged into a river by a WWTP overloaded by infiltration?, and ii) the ignorance of some criteria that can not be easily calculated, like environmental aspects.

It is obvious that the choice of any investment strategy regarding infiltration or exfiltration phenomena should include environmental, technical and financial aspects. Therefore, a general methodology based on an integrated and multi-criteria approach is proposed to compare and rank investment strategies to remedy infiltration/exfiltration problems.

6.2.1 Description of the methodology

The proposed methodology is represented in Figure 26.

Figure 26 : Methodology for definition, comparison and choice of investment strategies regarding infiltration/exfiltration based on a multi-criteria analysis.

Infiltration and exfiltration may have numerous impacts on the urban drainage system which have been summarized in a literature review carried out in the APUSS project (Neitzke, 2002). The inventory and the evaluation of these impacts (step 1) are fundamental in order to get an exhaustive view of the different aspects and to avoid neglecting of any of them. For any specific catchment, all impacts have to be checked (are they all pertinent in this particular catchment?) and evaluated (flows, volumes, transfer of pollutants, etc.) by means of appropriate measurements and modelling. Of course, the quality of this evaluation is a key element for the quality of the following steps in the proposed methodology. The necessary efforts shall be devoted to get and elaborate the requested knowledge of the phenomena and of the impacts.
The impacts need to be expressed through criteria that can be used for their evaluation and for the comparison of investment strategies (step 2). In the multi-criteria method Electre III (Roy, 1996), it is useless to express all criteria with a common unit. Quantitative criteria can be used but also qualitative criteria based on end-user’s expertise when data or models are not available.

Regarding the definition of the investment strategies (step 3), they may concern for infiltration sewer pipes (rehabilitation), the WWTP (higher treatment capacities) and CSO storage tanks (higher tank capacities). Regarding exfiltration, only rehabilitation of sewer pipes may solve the negative impacts induced by exfiltration. Each possible strategy should be defined in detail and evaluated (step 4). For the selection of rehabilitation technologies (RTs), many reviews exist in journals, conferences, textbooks and on the web. CLABSA s.a. and SINTEF s.a. within the FP5 project CARE-S developed both a database for RTs (Villanueva et al., 2004) and a cost tool (Freni et al., 2005) that helps the user to select appropriate RTs.

After the choice of possible additional criteria (step 5), all the quantitative criteria should be calculated and the qualitative criteria estimated (step 6). The analysis of the results and the ranking of the strategies are made by using a multi-criteria approach (step 7). Among the several multi-criteria methods which exist, the Electre III method (Roy, 1996) is widely used and accounts for uncertainties in the evaluation of the performance indicators or of criteria which is well appropriate for studies on environmental aspects, because of the high uncertainties in data and results. However, it should be clear that the weights and the thresholds involved in the use of the methods remain partly subjective and could be of great influence in the final results. In consequence, the proposed solutions have to be reviewed critically before making the final decision (step 8).

6.2.2 Description of the Ecully semi-virtual case study

The above methodology has been tested, for demonstration purposes, on a semi-virtual case study defined for the catchment of Ecully, in Lyon, France. It should to be clearly emphasized here that some specific simplifications and hypotheses have been introduced. Consequently, the conclusions drawn from this example should not be neither generalised nor extrapolated. Only the general methodology can be applied by any end-user with his/her own information and necessary adaptations.

Ecully is an urban catchment of 245 hectares located in the western suburbs of Lyon. The habitat is mostly residential and the imperviousness of the catchment is evaluated to 46 %. The drainage system is a combined sewer system with a 0.027 m/m mean slope. There is no wastewater treatment plant at the outlet of the catchment. For the purpose of this case study, a virtual WWTP has been introduced, designed and simulated by means of the ASIM software developed by EAWAG (Gujer and Larsen, 1995). In order to limit the impact of CSOs, a virtual CSO storage tank has also been introduced in the case study. All components are schematized in Figure 27, with some basic elements of information.

![Figure 27: Components of the Ecully semi-virtual case-study.](image-url)
Both the rainfall–runoff process over the catchment and the flow propagation in sewer network are simulated by initial and continuous rainfall losses functions and by means of a single linear reservoir. The model has been calibrated with 10 rainfall events and validated with 5 other events.

The WWTP is an activated sludge plant with biological nitrogen removal (nitrification-denitrification). The design has been calculated for a maximum inflow $Q_{\text{max}}$ equal to 3 times the dry weather flow DWF, for an influent temperature of 13 °C and for a maximum nitrogen concentration in the effluent equal to 10 mg N/L (i.e. 83 % removal) which leads to a COD removal rate equal to 95 %.

In case of overflow, the excess flow is temporarily stored in the CSO tank. If the CSO tank is full, the effluents are discharged directly into the receiving water. The water stored in the tank is then pumped towards the WWTP, provided its storage time has been less than 24 hours, in order to avoid septicity problems and related dysfunctioning in the WWTP. If the storage time is longer than 24 h, the water is discharged into the receiving water. These rules have been defined to simplify the case study. For real systems, other and more complex rules may be defined.

Seven simulation series have been carried out over a complete 15 years period for infiltration ratios IRs of 0, 25, 50, 75, 100, 150 and 200 % of the strict domestic dry weather flow. The reference simulation series (or scenario A) is the series calculated for IR = 0 % and with corresponding well designed WWTP. All simulations are carried out with a 6 minutes time step and results are then recalculated for daily time steps. The results for each simulation series are i) financial criteria, e.g. investment costs (Euros), operation and maintenance costs (Euro/year) and ii) environmental criteria: annual COD and total nitrogen loads discharged into the receiving water from both the WWTP and the CSO tank (kg/year). It was assumed that infiltration does not bring any pollutant into the sewer system and does not change the influent temperature which follows a sinusoidal function over one year.

### 6.2.3 Impacts of infiltration

Infiltration increases the hydraulic loads in the sewer system which leads to i) higher annual energy consumption for the pumping station and the WWTP operation and ii) more frequent discharges of non-treated pollutant loads due to more frequent combined sewer overflows. Infiltration also increases the annual pollutant loads discharged by the WWTP due to the decrease of WWTP efficiency regarding the carbon and nitrogen removal (Table 11).

Table 11 : Comparison of the situation without and with infiltration regarding the annual pollutant loads (kg/year) and the annual energy consumption (kWh/year) of the pumping station and the WWTP operation.

<table>
<thead>
<tr>
<th>Infiltration ratios IR (%)</th>
<th>0 %</th>
<th>25 %</th>
<th>50 %</th>
<th>75 %</th>
<th>100 %</th>
<th>150 %</th>
<th>200 %</th>
</tr>
</thead>
<tbody>
<tr>
<td>total COD load with infiltration / total COD load without infiltration (scenario A)</td>
<td>1</td>
<td>1.03</td>
<td>1.06</td>
<td>1.09</td>
<td>1.13</td>
<td>1.26</td>
<td>1.44</td>
</tr>
<tr>
<td>total NGL load with infiltration / total NGL load without infiltration (scenario A)</td>
<td>1</td>
<td>1.06</td>
<td>1.1</td>
<td>1.15</td>
<td>1.19</td>
<td>1.28</td>
<td>1.35</td>
</tr>
<tr>
<td>Total relative annual energy consumption (reference value = 1 for scenario A)</td>
<td>1</td>
<td>1.02</td>
<td>1.04</td>
<td>1.07</td>
<td>1.09</td>
<td>1.13</td>
<td>1.17</td>
</tr>
</tbody>
</table>

### 6.2.4 Definition of rehabilitation strategies

Three main investment strategies have been considered: i) investment in higher treatment plant capacities in order to account for infiltration, e.g. to have the same efficiency than without infiltration, ii) investment in rehabilitation of the sewer system, and iii) a mixed investment: renovation of 50 % of the pipes and investment in upgrading the WWTP.

Regarding the rehabilitation of the sewer system, three RTs were selected among repair, renovation and replacement techniques and average investment costs have been applied based on specialists’ experiences. Facing the lack of data regarding the structural state of sewer pipes in the Ecully catchment, the following hypotheses have been made: i) infiltration occurs only in the pipes located in the river aquifer, and ii) infiltration is diffuse and occurs along all pipes located in the aquifer. Another important hypothesis is that by rehabilitating 100 % of the sewer pipes subject to infiltration, 100 % of the infiltration problem will be solved. This optimistic
hypothesis should be considered with caution because groundwater may find other ways to enter the sewer systems (HC, manholes, etc.).

Regarding WWTP upgrading, the effects of infiltration can be compensated until IR = 75 % by increasing only the recirculation for better denitrification. From IR = 100 %, the maximum limit for recirculation flow is reached (4 times dry weather flow), and the size of the biological tank size has to be increased in order to fully account for infiltration (Table 12). The secondary clarifier is designed based on hydraulic constraints and its size increases as a function of the increasing IR. For IR higher than 100 %, the impacts of the WWTP are very significant.

<table>
<thead>
<tr>
<th>IR</th>
<th>0 %</th>
<th>25 %</th>
<th>50 %</th>
<th>75 %</th>
<th>100 %</th>
<th>150 %</th>
<th>200 %</th>
</tr>
</thead>
<tbody>
<tr>
<td>Biological tank volume</td>
<td>1.00</td>
<td>1.00</td>
<td>1.00</td>
<td>1.00</td>
<td>1.31</td>
<td>2.31</td>
<td>4.00</td>
</tr>
<tr>
<td>Secondary clarifier volume</td>
<td>1.00</td>
<td>1.21</td>
<td>1.53</td>
<td>1.83</td>
<td>1.98</td>
<td>2.27</td>
<td>2.56</td>
</tr>
</tbody>
</table>

6.2.5 Comparison of the strategies

None of the selected strategies appears as the best one for all criteria. Each scenario for IR ≥ 75 % has been analysed with the Electre III method using the different weights and thresholds. Of course, the choice of the weights and thresholds depends on each end user and on each case. An analysis of the robustness and of the sensitivity of these parameters should be carried out in order to check if the calculated ranking is stable or not. In the present case, the results appear quite stable. The application of the Electre III method leads to the results given in Table 13. Repair always appears as the best solution because it efficiently reduces the pollutant loads and is a cheap investment. With increasing IR, the upgrading of the WWTP becomes really expensive and not the most pertinent solution. “No investment” strategy is acceptable for low IR but is the worst solution for high infiltration due to the high pollutant loads discharged into the receiving water.

<table>
<thead>
<tr>
<th>IR scenario</th>
<th>Ranking results</th>
</tr>
</thead>
<tbody>
<tr>
<td>75 %</td>
<td>Repair → WWTP → Renovation → Replacement Without</td>
</tr>
<tr>
<td>100 %</td>
<td>Repair → Renovation → WWTP → Replacement → 50 % R Without</td>
</tr>
<tr>
<td>150 %</td>
<td>Repair → Renovation → Replacement → WWTP → Without 50 % R</td>
</tr>
<tr>
<td>200 %</td>
<td>Repair → Renovation → Replacement → WWTP → Without 50 % R</td>
</tr>
</tbody>
</table>

6.2.6 Conclusion

This methodology allows the end-user to account for different aspects (environmental, financial, etc.) of the impacts of infiltration (or exfiltration) on drainage systems, and to evaluate the impacts of possible selected investment strategies. The multi-criteria method Electre III is suggested to compare and to rank the various strategies, as this method has a high potential adaptability to diverse contexts and objectives. It is better than a simple traditional cost-benefit approach which over-promotes the economic aspects and introduces compensation of the heterogeneity between the different criteria.

The example of application reveals that the methodology can be used, but also that validated and site specific information (data, measurements, calibrated models, etc.) is necessary in order to carry out a valuable analysis. In many real practical cases, neither all impacts of infiltration and exfiltration nor the impacts of rehabilitation techniques can not be properly evaluated because of a lack of data, models and of knowledge on the involved phenomena. Consequently, many hypotheses will still be introduced in the analysis, with unknown bias in the
application of the methodology. Results should always be reviewed carefully. The final decision making process should take into account the context of the study and not forget the aspects that have not been modelled.

6.3 COST STRUCTURE AND COSTS OF APUSS METHODS

A questionnaire on sewer costs structures was developed and sent to selected APUSS associated end-users. The questionnaire was divided in two parts. Part 1 was about general aspects of in- and exfiltration and part 2 about technical and financial information about the sewer systems and environment. An infiltration problem is in most of the cities already known and often also as normally accepted, but only in a few cases the operator takes measures to reduce this problem. But, as expected, the technical data of the different sewer systems strew in a wide range. The transfer of information between different cities was not possible because of the large diversity of approaches and organisations.

However, a new cost-structure for sewer systems has been developed, dividing all arising costs in six groups. These are:
- costs for material
- wages and salaries
- miscellaneous operational costs
- disposal of waste
- depreciation
- calculated interest.

This classification into six groups had been successfully tested in different benchmarking projects for wastewater treatment plants in Germany (e.g. Schulz et al., 2002). All six groups are divided into sub-groups if necessary. For example, the group “Materials” is divided into eleven sub-groups. In Table 14, the new developed (reduced) commercial data entry form for a sewer system is displayed, which constitutes the basis for both cost-benefit analysis and benchmark studies.

<table>
<thead>
<tr>
<th>No.</th>
<th>Index</th>
<th>Overall sewer system</th>
<th>Private sewage system</th>
<th>Sewage collection</th>
<th>Stormwater treatment</th>
<th>Miscellaneous</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Total costs for the year: ___________ €</td>
<td>Total for cost group</td>
<td>Gully pots</td>
<td>Fat/light liquid separator</td>
<td>In house tank</td>
<td>gravity sewer</td>
</tr>
<tr>
<td>01</td>
<td>Materials</td>
<td>€</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>54</td>
<td>Power, water, natural gas, IP gas, transmitted heat</td>
<td>€</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>541</td>
<td>Other fuels (for motor vehicles)</td>
<td>€</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>544</td>
<td>Auxiliary andoperation fuels</td>
<td>€</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>547</td>
<td>Cost for purchases (services and materials)</td>
<td>€</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>5471</td>
<td>Of APUSS rehabilitation services, of APUSS rehabilitation materials</td>
<td>€</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>5472</td>
<td>Of APUSS maintenance services, of APUSS maintenance materials</td>
<td>€</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>551</td>
<td>Miscellaneous</td>
<td>€</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>5511</td>
<td>including other duties</td>
<td>€</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>552</td>
<td>Wages and salaries</td>
<td>€</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>5521</td>
<td>Of APUSS service and maintenance</td>
<td>€</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>5522</td>
<td>Of APUSS cleaning and inspection</td>
<td>€</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>553</td>
<td>Miscellaneous operational costs</td>
<td>€</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>554</td>
<td>Disposal of waste</td>
<td>€</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>555</td>
<td>Specification (standard)</td>
<td>€</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>556</td>
<td>Miscellaneous material</td>
<td>€</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>56</td>
<td>Total amount</td>
<td>€</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

The application of the cost benefit analysis has been illustrated by means of four examples (rehabilitation vs. replacement in different installation depths). The necessary detailed cost data for the replacement and rehabilitation of a sewer pipe came from different investment projects by Emschergenossenschaft and from literature. For comparison, yearly operation and investment costs were calculated, which revealed that the amortisation time of the planned measure was one of the main points in decision making.

For all measurement methods developed in the APUSS project, detailed ranges of costs (personal cost, cost for measuring equipment, and cost for consumption) have been evaluated by the APUSS partners (Table 15). They must be considered as costs for a "prototype" measurement which could be reduced when measurements will be applied on a routine basis. With regard to the planning phase, more experience would speed up preparatory analysis. With regard to the practical execution of the experiment, more experience leads to improved logistics and organizational aspects. This would reduce the overall personnel costs. Looking only at economical aspects, “classic” flow measurements appear more beneficial. However, the new measurement methods provide values of infiltration and exfiltration with better evaluation of uncertainty. The results obtained using these methods are more reliable than those obtained with the “classic" methods.
Table 15: Range of costs for experiment campaigns (estimation of each partner).

<table>
<thead>
<tr>
<th>Methods developed within the APUSS project</th>
<th>QUEST-C (exfiltration)</th>
<th>pollutant time series method (infiltration)</th>
<th>stable isotopes method (infiltration)</th>
<th>HC *) (in- and exfiltration)</th>
</tr>
</thead>
<tbody>
<tr>
<td>measuring equipment</td>
<td>~ € 10,700 to € 13,200</td>
<td>~ € 24,500 to € 28,300</td>
<td>~ € 17,200</td>
<td>~ € 300 **) to € 2,200</td>
</tr>
<tr>
<td>including</td>
<td>Flow measuring unit</td>
<td>Flow measuring unit s.: can probe</td>
<td>Flow measuring unit equipment for groundwater sampling</td>
<td>CCTV camera sealing balloons (only rental basis)</td>
</tr>
<tr>
<td>personnel cost</td>
<td>~ € 1,500 to € 2,000</td>
<td>~ € 1,500 to € 5,000</td>
<td>~ € 3,000</td>
<td>~ € 400 to € 800</td>
</tr>
<tr>
<td>calculated hours</td>
<td>~ 70 h – 80 h</td>
<td>~ 60 h – 110 h</td>
<td>~ 120 h</td>
<td>~ 80 h – 160 h</td>
</tr>
<tr>
<td>calculated Price per hour</td>
<td>€ 22 - € 27</td>
<td>€ 25 - € 45</td>
<td>€ 25</td>
<td>€ 5</td>
</tr>
<tr>
<td>number of experiments/samples</td>
<td>1 / 4 - 10</td>
<td>1 / 20</td>
<td>1 / 44</td>
<td>2-3 / -</td>
</tr>
<tr>
<td>cost for consumption</td>
<td>~ € 300 to € 500</td>
<td>~ € 500 to € 1,000</td>
<td>~ € 2,700</td>
<td>~ € 100 to € 200</td>
</tr>
<tr>
<td>total cost</td>
<td>~ € 13,200 to € 15,000</td>
<td>~ € 27,500 to € 30,400</td>
<td>~ € 24,900</td>
<td>~ € 1,200 to € 2,800</td>
</tr>
<tr>
<td>cost per experiment</td>
<td>~ € 1,800 to € 2,200</td>
<td>~ € 2,000 to € 5,500</td>
<td>~ € 5,700</td>
<td>~ € 600 to € 900</td>
</tr>
</tbody>
</table>

Cost information gathered from APUSS partners

*) HC: Measurement of house connection for in- and exfiltration

**) Price for the measuring equipment is only rental basis calculated
7. CONCLUSION AND PERSPECTIVES

During the period 2001-2004, the European research project APUSS (Assessing infiltration and exfiltration on the Performance of Urban Sewer Systems) was devoted to sewer infiltration and exfiltration questions.

New methods based on tracers experiments have been developed to measure infiltration and exfiltration in sewer systems: at subcatchment scale for infiltration and at sewer reach scale for exfiltration. For all methods, protocols, uncertainty analysis and data processing codes have been established and are available. The methods have been applied in various experimental sites with different contexts. Volumetric methods have also been tested to measure I/E in house connections.

Conceptual models to simulate infiltration and exfiltration at various time scales have been tested and applied. A statistical method based on the similarity approach has been developed to facilitate the identification of representative catchments and the extrapolation of experimental values to wider catchment at the whole city scale. As a natural complement, a software platform has been implemented to describe sewer systems, to store experimental data series on groundwater levels and I/E rates, to calibrate the models and to display all results. A user manual and case study application have been provided.

The cost structure of sewer systems has been analysed, and a cost benefit analysis has been carried out for replacement and rehabilitation of sewers. The range of costs of the new I/E measurement methods have been estimated. Specific I/E performance indicators have been established and applied to various case studies, and a multi-criteria methodology has been elaborated to compare and rank investment and rehabilitation strategies.

All above elements will contribute to help end users to get a better knowledge about I/E phenomena and to make decisions based on more accurate and elaborated information. All final reports, attached documents and published papers are publicly available on the APUSS website at http://www.insa-lyon.fr/Laboratoires/URGC-HU/apuss/.

8. ACKNOWLEDGEMENT

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9. REFERENCES


