Tracer-based hydrograph separation methods for sewer systems

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TRACER-BASED HYDROGRAPH SEPARATION METHODS FOR SEWER SYSTEMS

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Summary

Context
After decades of operation the sewer networks of many cities are in a state of disrepair. Under unfavourable conditions, the amount of groundwater infiltrating into the deteriorated pipes can even exceed 50% of the wastewater volume. This additional hydraulic load is particularly detrimental to the pollutant removal efficiency of wastewater treatment plants. Second, available storm event capacities are reduced and emptying times of retention tanks are extended. As a result, non-treated wastewater is discharged into the environment more frequently. In addition, there is increasing evidence that in many European cities groundwater levels are controlled by the serious drainage effect of permeable sewer systems. Sustainable strategies for the planning, operation and maintenance of urban drainage networks require adequate knowledge about their functional behaviour and interaction with the surrounding aquifers. However, nowadays methods for the quantification of these exchange processes are still subject to considerable uncertainties due to their underlying assumptions.

Research Questions and Objectives
Under dry weather conditions, wastewater typically has two sources of origin: (i) “real” foul sewage and (ii) extraneous water or “infiltration”. The term “infiltration” refers to groundwater that is unwantedly discharged into our sewer systems. Such non-polluted water enters the sewers through different kind of leaks like cracks, holes and open pipe joints. Whereas in the past infiltration was not necessarily unwanted (e.g., sometimes even brooks were intentionally led into the sewers for flushing purposes), the “parasitic water” is nowadays considered to be an unwanted interference factor that should be reduced or eliminated.

Larger punctual sewer leaks can be located relatively easily by closed-circuit television or walk-through inspections. Yet, sewer infiltration typically occurs through a vast number of smaller defects, which are rather "diffusively" distributed over widespread ranges of a catchment. Practitioners therefore often address this issue by calculating the “parasitic discharge” on the assumption that the night time minimum in the diurnal wastewater hydrograph is equal to the extraneous flows. In the context of today’s ever-growing metropolitan areas this practice is considered oversimplified and is likely to lead to erroneous results. In order to obtain more reliable information on extraneous discharges the use of tracers can be the optimal choice. Interestingly, to date little research has been devoted to this field.

Against this background, the primary goal of this thesis was to investigate the suitability of tracer based approaches to quantify infiltration at the catchment and subcatchment scale. This included (i) the elaboration of practice oriented experimental methods, (ii) algorithms for data processing and uncertainty analysis, and (iii) a guiding framework for experimental planning and design. The work was embedded in an international research project (APUSS), as a part
of which it was also envisaged that our partners would test the methods in several European cities under different operational and environmental conditions.

**New Infiltration Measurement Methods**

The *Stable Isotope Method* uses the stable isotopes compositions ($\delta^{18}O$, $\delta^{2}H$) of mains water and local groundwater as suitable proxies for the isotopic compositions of the foul sewage and the infiltrating water respectively. It therewith allows for the direct calculation of infiltration ratios. The method is suited to quantify infiltration, when useable differences in the oxygen or hydrogen isotope ratios of the drinking water and the infiltrating water exist. This is possible (but not necessarily granted) where drinking water originates from a distant hydrological regime, whilst infiltrating water stems from local precipitation.

The *Pollutant Time Series Method* does not require investigations of the drinking water or the origins of infiltration. Instead, the fraction of extraneous water is estimated by means of a combined analysis of measured time series of pollutant concentrations and wastewater discharge. This requires a certain dynamic of the wastewater hydrograph and pollutograph, which is usually suited by the diurnal variations of the wastewater discharge. The main innovations of this method are (i) the use of automatically operating in-line devices, which makes it possible to record time series of pollutant concentrations with a high temporal resolution, and (ii) advanced modelling procedures to identify even time-variant infiltration patterns. Measurements (concentration and discharge) are carried out at a single point in the sewer system without recourse to any additional external information.

**Conclusions**

Two generic infiltration methods have been developed, which will enable system operators and consulting engineers to substantially improve the knowledge about extraneous discharges and the hydraulic processes connected therewith. A significant effort has been devoted to the evaluation of uncertainties and errors both in the field measurements and in the data processing phase. For both methods, experimental protocols, uncertainty analysis procedures and data processing codes have been established and are available. The methods were evaluated and tested under field conditions and have been validated in the course of a comparative experimental study.

Successful implementation of the tracer methods depends on the local conditions and the skills and experience of the workforce. In practice it will thus be required to assess the confidence in the obtained results on an individual case basis. However, based on our experience we are confident that under suitable conditions a careful experimenter can estimate infiltration ratios with an accuracy of better than $\pm 4\%$ (95% confidence level related to the total wastewater flow). Basically, the new methods do therewith highly satisfy the requirements for practical applications in the assessment of sewer networks.
Zusammenfassung

Themenrahmen


Forschungsfragen und Ziele der Arbeit


Fremdwasserzutritte treten nur selten in Form grosser punktueller Defekte auf, die verhältnismässig einfach lokalisiert werden könnten. Typischerweise sind die vielen Einzel schäden eher „diffus“ über grosse Bereiche des Kanalnetzes verteilt. Infiltration von Fremdwasser wird deshalb in der Regel erst dann als ein Problem wahrgenommen, wenn die Summe der Schäden bereits beträchtliche Ausmasse angenommen hat. Die zuverlässige Kenntnis über das Ausmass und die Entwicklung solcher Fremdwasserzutritte ist eine wichtige Grundlage zur Beurteilung des baulichen Zustandes eines Kanalnetzes. Bisherige Messpraktiken lieferten hierzu allerdings nur bedingt aussagekräftige Informationen. So ist beispielsweise die in der
Schweiz übliche Interpretation des nächtlichen Abflussminimums im Kontext der gewachsenen Ballungsräume nicht mehr eindeutig.

Ziel dieser Arbeit war die Entwicklung neuer Messmethoden, um basierend auf natürlichen Tracersignalen eine zuverlässigere Bestimmung der Fremdwasserzutritte zu ermöglichen. Zentrale Punkte waren dabei die Erarbeitung von (i) praxisorientierten Versuchsmethoden, (ii) numerischen Algorithmen zur Datenauswertung und statistischen Analyse und (iii) Leitlinien für die Versuchsplanung. Die Zusammenarbeit in einem internationalen Forschungsprojekt (APUSS) ermöglichte zusätzlich, die Methoden in mehreren Europäischen Städten und unter verschiedenen lokalen Gegebenheiten der jeweiligen Forschungspartner zu testen.

**Neue Methoden zur Fremdwasserbestimmung**


**Schlussfolgerungen**

Die neuen Messansätze ermöglichen die zuverlässige Bestimmung des in einem Kanalnetz anfallenden Fremdwassers. Im Vergleich zu bisherigen Verfahren wird durch die Verwendung natürlicher Tracersignale eine zuverlässigere Bestimmung der auftretenden Infiltrationsmengen erzielt. Viele Fehlerquellen herkömmlicher Methoden schliessen sich dabei grundsätzlich aus. Damit können die neuen Tracermethoden dem Anwender vor allem im Hinblick auf eine objektivierte Planung der stets mit erheblichem wirtschaftlichem und technischem Aufwand verbundenen Kanalisationssanierungen wertvolle Informationen liefern.


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Most of the information now plotted in black and white against time or probability in the following pages, was at first hidden in numerous batches of (often not that fresh) water. It had to be recovered from wells and sewer pipes, processed in the lab and evaluated in the computer. For my co-workers this required the willingness to work under sometimes challenging conditions. Thank you Michael Federer, Markus Gresch, Sebastian Kempke Vinzenz Kroner, Qusay Sarhan, Inga Strohmann and Martin Wahr, who contributed to this work with either their internship work or diploma thesis. Thanks are also due to Jack Eugster and his laboratory team.

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Introduction
Introduction

1 Problem Setting

Aging sewer pipes were long considered "out of sight, out of mind", making sewer repairs a low fiscal priority when measured against other domains of public infrastructure. After decades of operation the sewer networks of many cities are in a state of disrepair, often as a result of inadequate preventative maintenance and insufficiently planned rehabilitation, or simply because of having reached the end of their useful design lives (Bishop et al., 1998; Davies et al., 2001; Ellis et al., 2003).

Capital resources are required to keep the quality of wastewater infrastructure up to date. However, often little information exists on the rates of deterioration of sewer networks, which are needed for a robust cost-benefit analysis to support decisions on sewer rehabilitation policies. The German Association for Water, Wastewater and Waste (DWA) claims that about 20% of the public sewer system in Germany requires short to mid-term rehabilitation, which would incur costs of around $70 billion (Berger and Lohaus, 2005). The US EPA estimates necessary expenditures of $390 billion to replace and upgrade existing wastewater infrastructure systems (including renewal of wastewater processing) in the United States over the next two decades (American Society of Civil Engineers, 2005). Furthermore, new legislative demands require enhanced ecologic performance criteria to be met by wastewater collection and treatment procedures. In the European Union, the full implementation of the Water Framework Directive 2000/60/CE will require substantial investments to amend and renew the existing wastewater technology. In this context, it is not conceivable to increase the wastewater treatment efficiency without increasing the efficiency of wastewater collection, i.e. the quality of sewer networks. In addition, the adverse effects of future climate change might exacerbate the need of investments in sewer infrastructure.

Sustainable strategies for the planning, operation and maintenance of urban drainage networks require adequate knowledge about their functional behaviour and hydraulic interaction with the surrounding aquifers. Nowadays, undesirable infiltration of groundwater into sewers can contribute to over 50% of the total discharge and is particularly detrimental to the pollutant removal efficiency of wastewater treatment plants. Nevertheless, conventional methods for the quantification of sewer infiltration are still subject to considerable uncertainties due to their underlying assumptions. As a consequence, knowledge about the water-tightness of a sewer system is often poor and investments are frequently based on uncertain information and limited data.
2 Sewer Infiltration

Under dry weather conditions, wastewater typically has two sources of origin: (i) drinking water the properties of which have changed through use in households, trade and industry (i.e. “real” foul sewage) and (ii) extraneous water from groundwater infiltration, public fountains, drainage pipes etc. (i.e. infiltration):

\[ Q_{\text{Wastewater}} = Q_{\text{Foul Sewage}} + Q_{\text{Infiltration}} \]  

(Eq.1)

In order to obtain comparative values, the amount of infiltration is commonly expressed as a fraction of the total wastewater discharge (infiltration ratio):

\[ \chi_{\text{Infiltration}} = \frac{Q_{\text{Infiltration}}}{Q_{\text{Wastewater}}} \]  

(Eq.2)

Sewer infiltration can enter the wastewater collection system through a variety of means such as defective pipes (cracks and fissures), pipe joints, couplings, manholes and house connections (Figure 1). Infiltration does not include, and is distinguished from, inflow. Inflow is stormwater that enters separate foul sewers from sources such as misconnected yard drains, roof downpipes, manhole covers, cross connections between storm sewers and sanitary sewers etc. (Butler and Davies, 2004).

![Figure 1. Sources of sewer infiltration (exemplary illustration, not meant to be exhaustive).](image)

Reported infiltration rates range widely between 0.01 to 1.0 m³/day/mm pipe diameter/km sewer length (Metcalf & Eddy INC., 1991). In the UK, infiltration ratios have been found to range from 15% to 50% of average dry weather flow (White et al., 1997). A nation wide census in Germany revealed an average infiltration ratio of 30% based on estimates provided
by the treatment plant operators surveyed (Statistisches Bundesamt, 2003). About 10% of the German treatment plants may suffer infiltration ingresses exceeding 50% of average dry weather flow (ATV-DVWK Arbeitsgruppe Fremdwasser, 2003). With regard to annual discharge patterns, Weiß et al. (2002) state that for a combined sewer system the typical relation between sewage, stormwater and infiltration inflow at the wastewater treatment plant may equal 0.30/0.35/0.35, thus more than 2/3 of the water passing through the plant was originally non- or little-polluted.

The rehabilitation of such defective systems has a great importance for a long-term optimisation of operational procedures and the protection of receiving waters: the additional hydraulic load transferred to the wastewater treatment plant is particularly detrimental to its pollutant removal efficiency (Decker, 1998). Adding to this, the treatment plant’s available storm event capacity is reduced and emptying times of retention tanks are extended. As a result, non-treated wastewater is discharged into the environment more frequently. Infiltration also adversely affects the regular operation of sewer systems and overflow regulatory facilities (WEF and ASCE Joint Task Force, 1994) and causes an increase in energy consumption and operating costs of pumping stations and treatment plant infrastructures. In addition, there is increasing evidence that in many European cities groundwater levels are controlled by the serious drainage effect of permeable sewer systems.

Probable factors leading to the deterioration of sewer pipes and joints depend on the material (e.g. vitreous clay, concrete, PVC, historical remains like brick lined sewers), the local ground conditions and the quality and type of construction work employed (Jones, 1984). A poor structural state of a sewer network is basically the consequence of (i) ageing of pipes (some parts of European sewer systems 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 and (iv) lack of appropriate investment and rehabilitation strategies (Bertrand-Krajewski et al., 2005). Fenner (1990) points out that several sewer surveys have shown that most types of sewer defects are not linearly related to age, but rather reflect the changes in design and construction used in the different historical periods.

Davies et al. (2001) describes a general process that can finally lead to extensive structural failure of rigid sewer pipes: Deterioration is often triggered by initial minor defects such as leakage through defective joints (Fenner, 1990). The entrainment of soil into the sewer through the initial defect can in turn promote the loss of support from the surrounding underground material, followed by significant deflections of the sewer pipes, causing cracking and failure. This further worsens the movement of soil into the sewer, accelerating the cavitation process (De Silva et al., 2001). In extreme cases, this process may finally result in a collapse of the deteriorated pipe (Davies et al. 2001 and references therein).
3 Conventional Evaluation of Sewer Infiltration

The applicability of various infiltration quantification techniques clearly depends on the scope of a survey. Today, larger punctual sewer leaks can be located relatively easily by closed-circuit television or walk-through inspections. Yet, sewer infiltration typically occurs through a vast number of smaller defects, which are rather diffusively distributed over widespread areas of a sewer system. Likewise the direct metering of infiltration by physically isolating a certain sewer length (i.e. by means of plugs inserted to the sewer pipes) is only feasible for comparable small subsections and over short time periods. Most conventional infiltration / inflow studies therefore address this issue by different types of local water balances or calculate the “extraneous discharge” based on flow measurements that are casually combined with more or less complex statistical methods (Dennis et al., 1981; Murray, 1987). In many cases the data analysis is simply based on the assumption that the night-time minimum of the diurnal wastewater hydrograph equals the extraneous flows. In the context of today’s ever-growing metropolitan areas these practices are considered oversimplified and are likely to lead to erroneous results. However, to date little research has been devoted to the development of alternative practices.

Conventional infiltration measurements can be categorized into:

1. Water balances
2. Hydrograph based approaches
3. Chemical methods

The following section gives a brief overview on these principles.

3.1 Water balances

At the first glance, it seems to suggest itself that sewer infiltration may simply be calculated from the volumetric differences between wastewater discharge and drinking water usage, either on a short or long term timescale. However, this procedure is basically hindered because of (i) the spatial incongruence of drinking water supply and wastewater collection networks, (ii) the uncertainties bound to the measurement of drinking water consumption and wastewater discharge, and (iii) the imprecise information about water losses from drinking water networks. Direct water balances might be suitable in exceptional cases, but have seldom been reported yet (Kamm, 1984). Nevertheless, many semi-statistical infiltration/inflow-approaches do implicitly include a simple water balance to derive an estimate for the infiltration part of the extraneous flows. For example, the “triangle-approach” (Haller and Weiß, 2001) is based on a graphical evaluation scheme to delineate the annual dry weather discharge (originally published by Annen (1980)), which is then compared to the average drinking water consumption.
3.2 Hydrograph based methods

Hydrograph based methods mostly imply that the hydrograph’s diurnal night-time minimum equals the amount of extraneous discharge. The categorical weak point of these minimum night flow based infiltration estimates is that even little relocation of wastewater discharge from daytime to night can cause a substantial bias in the infiltration estimates. To illustrate this point, consider the following thought experiment: We assume that the night-time troughs of a hydrograph would actually represent the current quantity of infiltrating water (Figure 2, left). Note that this example hydrograph is derived from a smoothed reproduction of real discharge measurements in one of our experimental catchments (trunk sewer, about 5'400 residents connected; receding infiltration pattern is caused by some rainfall preceding the dry weather flow situation shown). Integration of $Q_{\text{infiltration}}$ over the displayed 6 days period yields a total of 4'277 m$^3$. For a future scenario simulation, we now relocate 7.5% of the domestic water consumption from daytime to night (Figure 2, right). Obviously, the corresponding infiltration estimate based on the night-time minima of the relocated water use scenario is heavily biased, amounting to a total of 6'454 m$^3$ of apparent infiltration discharge. The total wastewater discharge is 10'498 m$^3$ in both the original and the altered version of the hydrograph. Whilst the correct infiltration ratio is 41%, the biased estimate in the future scenario erroneously yields $X_{\text{infiltration}} = 61\%$.

In our scenario, comparably little relocation of the daily drinking water consumption caused a substantial bias in the minimum night flow based infiltration estimates. Example given for an order of magnitude, the Swiss professional association for gas and water (Schweizerischer Verein des Gas- und Wasserfaches; http://www.svgw.ch/) states that the average Swiss household at present uses 18.6% of the total domestic drinking water consumption of around

![Figure 2](image-url)

**Figure 2.** Schematic representation of a wastewater hydrograph for a small size catchment in which night time minimum discharge is supposed to equal the amount of extraneous flows (left). Simulated alteration of the hydrograph pattern by relocating 7.5% of the domestic water consumption from daytime to nighttime (right).
162 l/day/person for laundry. Our exemplary scenario might therefore easily eventuate, if only 40% of the domestic laundry activities were relocated to night-time (which is not far-fetched considering i.e. possible new water price regulations, less strict rules on keeping peace at night in rented flats, use of washing machines with clock-timers etc.).

The technical literature contains some information on commonly used correction factors to account for domestic night residual sewage flow (Abwassertechnische Vereinigung, 1982; Fischer, 1990; Triebel, 1973). However, these procedures generally involve a great degree of subjectivity. The correction factors suggested were supposedly not even intended for actual measurement purpose in the original text sources, but rather aimed to provide planning guidelines or schematic control of environmental regulations. Furthermore, the question arises if the commonly used correction parameters are still up to date in view of today’s catchments sizes (long flow distances) and wastewater production patterns (24-hours usage of drinking water).

A more rational justified and promising future direction is essentially taken by hydrological approaches which combine the modelling of wastewater discharge and infiltration / inflow processes with a model calibration based on time series of an outer disturbance variable. Conceivable outer influence variables to be coupled with a hydrological sewer model are for example the spatial and temporal variation of rainfall (Abdel-Latif and El-Hosseiny, 1995; Grum et al., 2005) and/or groundwater tables (Karpf and Krebs, 2004).

### 3.3 Chemical methods

In order to obtain reliable information on the material fluxes and to monitor the chemical and hydraulic interactions within urban drainage networks, the use of artificial or natural tracers can be the optimal choice. Chemical hydrograph separation is based on a tracer mass balance approach to differentiate discharge components from two or more sources. In a binary mixing system, the infiltration ratio \( X_{\text{Infiltration}} \) is given by:

\[
X_{\text{Infiltration}} = \frac{C_{\text{Foul Sewage}} - C_{\text{Wastewater}}}{C_{\text{Foul Sewage}} - C_{\text{Infiltration}}} \tag{Eq.3}
\]

where \( C_i \) are the tracer concentrations. Whilst \( C_{\text{Wastewater}} \) can be measured in the sewer, the crucial problem is the estimation of \( C_{\text{Infiltration}} \) and \( C_{\text{Foul Sewage}} \). These parameters are basically not directly accessible at the catchment or even subcatchment scale. Furthermore, this problem can not be solved by using artificial tracers, as the artificial labelling of drinking water is prohibited (health and legal concerns) and the whole groundwater system can not be labelled either (environmental concerns and practicalities).
Two different types of “chemical” approaches may be distinguished:

a) Combining observed wastewater concentrations with assumptions about a specific per capita pollutant loading and the daily amount of foul sewage generated per inhabitant.

b) Pollutograph based quantifications, which analyse the dynamics of diurnal wastewater composition changes observed without recurring to external information.

a) Specific per capita pollutant loads

One of the first mentions of chemical parameters used to quantify extraneous flows is found in Hörler (1966), who suggests the use of chloride concentrations. Yet this early text does not specify any further details. The ATV task force on sewer infiltration (ATV-DVWK Arbeitsgruppe Fremdwasser, 2004) lists critical mean wastewater threshold values for chemical oxygen demand (\(\text{COD} < 400 \text{ mg/l}\)), total Kjedahl nitrogen (\(\text{TKN} < 35 \text{ mg/l}\)) and phosphorus (\(\text{TP} < 6 \text{ mg/l}\)) that may indicate elevated rates of sewer infiltration. System-dependently it is difficult to find suitable markers for precise quantifications, since sewage chemistry typically exhibits substantial background fluctuations that obscure the natural tracer signal. In this context, the estimation of unit per capita pollutant loads becomes a question at issue (Bonomo et al., 1994; Verbanck et al., 1989). In a more detailed investigation, Verbanck (1993) demonstrates the use of borates, for which a per capita load is derived from a quantitative assumption on the use of washing powder in the catchment. However, in general case the required accurate knowledge about the average per capita substance loadings and the average per capita foul sewage discharge is considered to be rather unavailable.

b) Pollutograph based quantification

A first comprehensible documentation of a pollutograph based infiltration method was presented within the scope of a series of research projects initiated by the Swiss Federal Office for Environmental Protection (BUS) in the early 1980s (Schweizer Bundesamt für Umweltschutz, 1983a, 1983b, 1984). Hager et al. (1984; 1985) derived an algebraic equation by which \(X_{\text{Infiltration}}\) is calculated from the mean and minimum discharge and the mean and minimum pollutant concentration of a diurnal time series. However, despite reports on applications of the “chemical method” often refer to these publications, practitioners indeed seem to prefer a somewhat different (graphically based) evaluation scheme (e.g. suggestions in the guidelines for the preparation of the SWISS Master Plan of Urban Drainage (GEP) (Verband Schweizerischer Abwasserfachleute, 1989)).

The following paragraph aims to provide a short summary of the basic principles of conventional pollutograph based quantification schemes:
The pollutograph based quantification of extraneous flows considers the diurnal variation of pollutant concentration \( (C) \) and discharge \( (Q) \). The pollutant load in the wastewater flow is obtained from the measurements by:

\[
L_{\text{Wastewater}}(t) = Q_{\text{Wastewater}}(t) \cdot C_{\text{Wastewater}}(t)
\]  
(Eq.4)

All hitherto used quantification schemes are commonly based on the simplified assumption of

- time invariant infiltration discharge \( Q_{\text{Infiltration}} = \text{const.} \)
- time invariant foul sewage concentration \( C_{\text{Foul Sewage}} = \text{const.} \)
- negligible pollutant concentration in the extraneous waters \( C_{\text{Infiltration}} = 0 \)

Thus

\[
L_{\text{Wastewater}}(t) = Q_{\text{Foul Sewage}}(t) \cdot C_{\text{Foul Sewage}} + Q_{\text{Infiltration}}(t) \cdot C_{\text{Infiltration}}
\]  
(Eq.5)

reduces to:

\[
L_{\text{Wastewater}}(t) = Q_{\text{Foul Sewage}}(t) \cdot C_{\text{Foul Sewage}}
\]  
(Eq.6)

which can be converted to:

\[
L_{\text{Wastewater}}(t) = \left( Q_{\text{Wastewater}}(t) - Q_{\text{Infiltration}} \right) \cdot C_{\text{Foul Sewage}}
\]  
(Eq.7)

From two pairs of measured discharge and concentration, one obtains

\[
L_{\text{Wastewater}}(t_1) = \left( Q_{\text{Wastewater}}(t_1) - Q_{\text{Infiltration}} \right) \cdot C_{\text{Foul Sewage}}
\]  
(Eq.8)

and

\[
L_{\text{Wastewater}}(t_2) = \left( Q_{\text{Wastewater}}(t_2) - Q_{\text{Infiltration}} \right) \cdot C_{\text{Foul Sewage}}
\]  
(Eq.9)

which can then be solved for \( Q_{\text{Infiltration}} \). In the case of three or more pairs of measured discharge and concentration values, the system of equations is overdetermined and can be solved by linear regression techniques. Practitioners do typically use a simplified graphical evaluation scheme as shown in Figure 3.
Figure 3. Simplified graphical evaluation of combined discharge and pollutant concentration time series data (raw wastewater at a treatment plant influent). Measured values (left) are transformed into a pollutant-load versus discharge plot (right). The intersection of the linear regression line with the discharge-axis specifies the infiltration estimate (approximately 430 m$^3$/h in this example).

The graphical representation of the wastewater data can support at least a certain intuitive check on if the data set complies with the requirements of the method. However, note that in Figure 3 the visual impression of having obtained a somehow acceptable model fit quality can be misleading, as it is largely induced by the fact that the dependent variable COD-load is incorporating the independent variable $Q$ as a multiplicative factor. In unfavourable situations (strong departure from the underlying assumption that $Q_{\text{Infiltration}}$ and $C_{\text{Foul Sewage}}$ are constant) inconsiderate application of the purely algebraic evaluation scheme of Hager et al. (1984; 1985) might be even more misleading, as it does not comprise any checks on the plausibility of the results.

4 Short Overview of Stable Isotope Hydrology

In the last decades the application of natural stable and radioactive isotopes has evolved into an established discipline in earth- and environmental-sciences (Faure et al., 2005; Fritz and Fontes, 1980; Hoefs, 2004). Isotope tracers are also frequently used in different biological systems, making this instrument increasingly important to ecological research (Fry, 2006; Kendall, 1998; Kohen, 2005). Natural abundance isotope techniques exerted significant influence in many areas of hydrology and hydrogeology and have become a widespread tool for the investigation of surface- and groundwater behaviour and exploration (Aggarwal et al., 2005; Clark et al., 1982; Cook, 2000; Gat and Gonfiantini, 1981; Mook, 2006). The subject area was in particular promoted by different programmes of the International Atomic Energy Agency, which has recently published a comprehensive review on isotope hydrology methodologies and their practical applications (Gat et al., 2001; Mebus, 2001; Mook, 2001; Rozanski et al., 2001; Seiler, 2001; Yurtsever, 2001). However, corresponding implementa-
tions in the field of urban hydrology are still rare (Butler and Verhagen, 1997; Harris et al., 1999; Sidle, 1998). The following section gives a short overview of basic terminologies and principles used in stable isotope hydrology.

4.1 The stable isotopes composition of water

Natural water ($H_2O$) is composed of the three stable isotopes of oxygen ($^{16}O$, $^{17}O$ and $^{18}O$), the two stable isotopes of hydrogen ($^1H$ and $^2H$ or $D$ = deuterium) and the unstable isotope Tritium ($^3H$). Tritium decays by beta decay. Because of their higher abundance, usually only the molecules $H_2^{16}O$, $H_2^{18}O$ and $HD^{16}O$ are investigated. The average ratio of $HD^{16}O$ to $H_2^{16}O$ in natural waters is about 1:6400, the ratio of $H_2^{18}O$ to $H_2^{16}O$ is about 1:500 respectively. These ratios can be estimated precisely by mass spectrometric techniques and have gained a wide range of applications in hydrology.

In general practice stable isotope data are cited in the $\delta$-notation in reference to an international accepted standard (Equations 10a and 10b). For the investigation of water the standard VSMOW (Vienna Standard Mean Ocean Water) has been established, that is distributed by the International Atomic Energy Agency (IAEA, 1995).

$$
\delta^{2H}_{Sample} = \frac{(^{2H}/^{1H})_{Sample} - (^{2H}/^{1H})_{SMOW}}{(^{2H}/^{1H})_{SMOW}} \cdot 1000 \ [\text{‰}] \tag{Eq.10a}
$$

$$
\delta^{18O}_{Sample} = \frac{(^{18O}/^{16O})_{Sample} - (^{18O}/^{16O})_{SMOW}}{(^{18O}/^{16O})_{SMOW}} \cdot 1000 \ [\text{‰}] \tag{Eq.10b}
$$

The $\delta$-value describes the relative deviation of an isotope ratio in a sample to the isotope ratio in the standard. More negative $\delta$-values indicate a “lighter”, more positive $\delta$-values indicate a “heavier” isotopic composition. This notation has been established for technical measurement reasons. It generates no problems for our mixing calculations and can be used analogous to a concentration value.

The absolute difference in isotopic composition between two samples is conventionally called the “isotopic separation” and is denoted as:

$$
\Delta^{18}O = \delta^{18}O_{Sample 1} - \delta^{18}O_{Sample 2} \quad \text{and} \quad \Delta^{2H} = \delta^{2H}_{Sample 1} - \delta^{2H}_{Sample 2} \tag{Eq.11a,b}
$$

In the following text this notation is used to describe the magnitude of the available tracer signal, which is in our case the isotopic separation between foul sewage and infiltrating water:

$$
\Delta^{18}O = \delta^{18}O_{Foul Sewage} - \delta^{18}O_{Infiltration} \quad \text{and} \quad \Delta^{2H} = \delta^{2H}_{Foul Sewage} - \delta^{2H}_{Infiltration} \tag{Eq.12a,b}
$$

12
4.2 Isotopic fractionation

Isotopes of the same element can be partitioned or separated in chemical reactions as well during phase transitions due to differences in the conversion rates for the different molecular species. The partitioning of isotopes between two substances or two phases of the same substance is called isotopic fractionation. Fractionation processes can occur under equilibrium or non-equilibrium (kinetic) conditions. As a third type of processes, fractionation can also occur as a result of molecular diffusion (Criss, 1999).

The fractionation factor \( \alpha \) between two compounds \( A \) and \( B \) is defined as the isotope ratio in \( A \) divided by the corresponding isotope ratio in \( B \):

\[
\alpha_{A-B} = \frac{R_A}{R_B} 
\]

(Eq.13)

For example, the fractionation associated with the isotopic exchange between water molecules during evaporation and condensation (\( H_2O_{\text{liquid}} \leftrightarrow H_2O_{\text{vapour}} \)) is defined as follows:

\[
\alpha_{\text{vapour-liquid, } O} = \frac{^{18}O^{16}O_{\text{vapour}}}{^{18}O^{16}O_{\text{liquid}}} = \frac{\delta^{18}O_{\text{vapour}} + 1000}{\delta^{18}O_{\text{liquid}} + 1000} 
\]

(Eq.14)

\[
\alpha_{\text{vapour-liquid, } H} = \frac{^{2}H^{1}H_{\text{vapour}}}{^{2}H^{1}H_{\text{liquid}}} = \frac{\delta^{2}H_{\text{vapour}} + 1000}{\delta^{2}H_{\text{liquid}} + 1000} 
\]

(Eq.15)

4.3 Stable isotope processes in the water cycle

Within the global water cycle, the stable isotope composition of water is predominantly controlled by large-scale meteorological evaporation and precipitation processes: A sequence of phase transformations causes a successive change of the isotope composition. Evaporation from the oceans is the major source of atmospheric water masses. The isotopic composition of modern seawater is close to VSMOW. When water evaporates from the ocean, the water vapour is depleted in \( D \) and \( ^{18}O \) (Craig, 1961; Craig and Gordon, 1965). This is due to the higher vapour pressure of \( H_2^{16}O \) compared to that of \( H_2^{18}O \) and \( HD^{16}O \), as well as to kinetic effects that are linked to diffusion processes. As clouds move overland, the heavy water molecules containing \( D \) and \( ^{18}O \) will preferentially rainout first in coastal areas and the remaining water vapour is further depleted in these isotopes. When the clouds continue to move inland and more water condenses into rain, the isotope ratio of the liquid water becomes more and more negative or “depleted” in the heavy isotopes (rainout). Consequently, precipitation in the inner parts of the continents is isotopically lighter than rain that
precipitates in the coastal regions (Rozanski et al., 1993). These relationships are referred to as “continental effect” (Figure 4).

Topographic elevations as for example the Alps will force the vapour masses to rise in altitude. The clouds cool adiabatically (expansion) and rainout. Equilibrium fractionation between vapour and the condensing phases preferentially partitions $D$ and $^{18}O$ into the rain or snow. Again, the vapour becomes progressively depleted in $D$ and $^{18}O$. Additionally, the extent of isotopic equilibrium fractionation between water and vapour is negatively correlated to the temperature: the fractionation is larger at lower temperatures. Figure 5 exemplifies this evolution of the isotopic composition in precipitation with results obtained from a simplified numerical model. The specific progress of isotopic composition changes of a finite vapour mass that is caused by the proceeding rainout under equilibrium fractionation between vapour and the condensing water phases is also referred to as a “Rayleigh distillation” type process.

**Figure 4.** Broad scale distribution of mean isotopic compositions in precipitation in Europe (IAEA, 2001): In a first order approximation this spatial distribution can be ascribed to the continental effect.
Figure 5. Isotopic composition of precipitation from a cooling vapor mass (numerical simulation).

The computation is based on a simplified model, by using the Magnus formula to derive a relationship between saturation vapor pressure and temperature (i.e. Magnus parameters given in Deutscher Wetterdienst (1987)). Temperature dependent isotopic fractionation factors for the phase transitions liquid / vapor and ice / vapor were taken from Majoube (1971) and O'Neil (1968) respectively.

In average, tropospheric temperatures decrease by about 0.6° C per 100-m rise in altitude, thus causing a proceeding rainout and isotopic depletion of the vapor masses. As a consequence of these so called “altitude effects” we can distinguish natural water reservoirs that are recharged from precipitation at high altitudes from those recharged at low altitudes. Based on a broader data set of groundwater and surface watercourses from the Swiss Alps and the Swiss Plateau Siegenthaler et al. (1983) infer an average decrease of 0.22‰ in $\delta^{18}O$ per 100 m rise of the contributing recharge area (Figure 6). Analogously a decrease of 0.26‰ / 100m was stated based on precipitation samples (Siegenthaler and Oeschger, 1980). More recent precipitation data of the Swiss National Network for Isotopes in the Water Cycle show an altitude effect of 0.2‰ / 100m for $\delta^{18}O$ and 1.2‰ / 100m for $\delta^2H$ respectively (Schürch et al., 2003). In a global overview, reported data on $\delta^{18}O$ altitude gradients generally fall into a range between 0.10‰ and 0.50‰ per 100 m rise (Bortolami et al., 1979; Clark and Fritz, 1997; Clark et al., 1982; Dubois and Flück, 1984; Fontes and Olivry, 1977; Moser and Stichler, 1970).
4.4 Mass spectrometric measurements

Analyses of water samples for both $^2\text{H}/^1\text{H}$ and $^{18}\text{O}/^{16}\text{O}$ ratios are routinely carried out with gas isotope ratio mass spectrometers (IRMS). Both dual inlet as well as continuous flow inlet systems are used nowadays. Against the background of analytical methods being continually developed and improved, only some general hints should be given here.

Conventionally, $\delta^{18}\text{O}$ analyses are performed indirectly on carbon dioxide that is previously equilibrated with the water sample at a constant temperature (i.e. 25°C). Due to an isotope exchange reaction $\delta^{18}\text{O}_{\text{water}}$ is linked to $\delta^{18}\text{O}_{\text{carbon dioxide}}$ by a well documented relationship (Epstein and Mayeda, 1953). $\delta^2\text{H}$ analyses are performed directly on hydrogen gas obtained on-line or off-line through high temperature reduction of water on a metal, like specially prepared zinc, chromium or, seldom used, uranium (Gehre et al., 1996a; Gehre et al., 1996b; Kendall and Coplen, 1985). An alternative method is based on an isotope equilibration technique that involves the exchange of hydrogen gas with the water sample in the presence of platinum powder as a catalyst (Coplen et al., 1991). The internal laboratory precisions of these analytical methods are typically within a range of $\pm 0.05$ to $\pm 0.2\%$ ($\delta^{18}\text{O}$) and $\pm 0.5$ to $\pm 1.5\%$o ($\delta^2\text{H}$) respectively (1σ-level).

Novel preparation systems based on an anaerobic thermal combustion (pyrolysis) of the sample have recently been introduced (thermal combustion elemental analyser - TC/EA). They can also be used for isotope measurements on water, which is then transferred to carbon monoxide and hydrogen gas. The method becomes more and more popular, as it is capable of
a high sample throughput rate. However, the attainable analytical precision is still inferior compared to conventional analyses and, from the actual point of view (year 2006), considered not to be sufficient for our purpose yet.

It should be emphasized that the interlaboratory precision (obtained when the same sample is analyzed by different laboratories) is naturally of a poorer quality than the internal laboratory precision mentioned above (repeated analysis of the same sample in the same lab). This becomes relevant when measurement results obtained from different laboratories should be analysed jointly within a study (Gröning et al., 2003; Lippmann et al., 1999). Lippmann et al. state that the apparent interlaboratory precision (1σ-level) derived from the whole pool of 87 laboratories participating at the “2nd interlaboratory comparison for deuterium and oxygen-18 analysis of water samples” (organized by the International Atomic Energy Agency, Vienna) was in the order of 0.11‰ for δ¹⁸O and 1.3‰ for δ²H. This inter-laboratory precision potentially accumulates with the internal laboratory precision when samples are distributed over several laboratories.

**Figure 7.** Application of natural abundance isotope tracers in sewer networks. The stable isotopes composition of natural ground and surface waters is predominantly determined by the topographic elevation and continental setting of the contributing recharge area (altitude effect and continental effect in precipitation). For application to sewer infiltration, convenient isotopic separation can exist where drinking water originates from a distant hydrological regime but groundwater stems from local precipitation.
4.5 Water stable isotopes as tracers in sewer systems

The decomposition of hydrographs by natural isotope tracers has become relatively standard in studies on catchment and streamflow hydrology and did as well lead to detailed discussion on the underlying runoff generation models (Butler and Verhagen, 1997; Buttle, 1994; Chanat and Hornberger, 2003; Sklash et al., 1975). However, the usage in wastewater engineering is still uncommon. In particular, the application of isotope tracers for the separation of diurnal wastewater hydrographs in sanitary or combined sewers has not yet been demonstrated.

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. Under dry weather conditions, wastewater hydrograph separation by stable isotopes requires a sufficient isotopic separation between local drinking water and infiltrating groundwater. Suitable isotopic differences can exist, where drinking water originates from a watershed that is situated in a different hydrological regime than the urbanized area, whereas sewer infiltration originates from groundwater that is recharged by local precipitation (Figure 7). Beyond this, it is critical to verify a low spatial variability of the isotopic composition within the ambient aquifer.

5 Context and Research Objectives

The work presented in this thesis was embedded in the European research project APUSS (Assessing infiltration and exfiltration on the Performance of Urban Sewer Systems (Bertrand-Krajewski et al., 2005)). The APUSS project examined the potentials for a more problem oriented management of existing wastewater collection infrastructures, i.e. if profound knowledge about the structural quality and functional efficiency of a sewer system can be used as the basis for efficient planning and investment strategies. As sewer infiltration and wastewater losses are fundamental issues when evaluating the environmental impacts caused by defective sewers, it was intended to develop methods and tools for the quantitative assessment of infiltration and exfiltration on a more scientific and sound basis than before (Appendix A).

The primary goal of this thesis is to investigate the suitability of tracer based approaches to quantify infiltration at the catchment and subcatchment scale. This includes the elaboration of practice oriented experimental methods, corresponding algorithms for data processing and uncertainty analysis, and a guiding framework for experimental planning and design.
In specific, the project involved the following main research tasks:

- Identify suitable tracer systems (substances and analytics) for the quantification of infiltration in sewer systems.
- Elaborate appropriate experimental procedures and suitable methods for data analysis.
- Demonstrate and field-test the applicability of the novel methods in the course of full-scale experiments.
- Develop an appropriate framework for the assessment of uncertainties.
- Validate the obtained experimental results.
- Provide instructions for implementing optimal operational conditions and experimental layouts.
- An appraisement of the degree of accuracy that is anticipated to be achievable in routine applications.

In addition, guidelines on the preparation and performance of the tracer experiments as well as detailed protocols on the data analysis were required to be provided to the APUSS partners.

With respect to the foreseen dissemination of the novel methods to practicing engineers and wastewater systems operators, the project stipulated that no artificial labelling of drinking water (health and legal concerns) or groundwater (environmental concerns and practicalities) should be used.

6 Outline of the Thesis

Two innovative tracer methods for the quantification of parasitic discharges in sewers have been developed. The **Pollutant Time Series Method** estimates the fraction of infiltrating water from a combined analysis of continuous in situ measured wastewater discharge and pollutant concentration time series. It requires a certain dynamics of the wastewater hydrograph and pollutograph, which is usually guaranteed by the diurnal variations of the wastewater discharge. The **Stable Isotope Method** uses the different isotopic signatures of drinking 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 methods were evaluated and tested under field conditions and have been validated in the course of a comparative experimental study.

**Chapter 1** introduces the concepts of wastewater hydrograph separation using the **Pollutant Time Series Method**. It details on applicable mixing models and gives emphasis on the
practical identifiability of the corresponding model parameters as well as on the influence that measurement errors take on the final infiltration results. The application is subsequently demonstrated with two case studies.

**Chapter 2** addresses the decomposition of wastewater hydrographs by using the stable isotope composition of water as an intrinsic tracer. The proper use of the *Stable Isotope Method* requires thorough hydrological investigation to define the principal interactions and pathways of different waters in a catchment. The performance of corresponding measurement campaigns is demonstrated based on a comprehensive field study. The chapter then focuses on the statistical analysis of field data and introduces suitable Monte Carlo simulation based evaluation methods. Furthermore, a detailed discussion is devoted to certain hydrodynamic transport phenomena: Apparent intraday fluctuations of the infiltration discharge observed in our data suggest the existence of larger cumulative backwater zone volumes that have not yet been accounted for.

**Chapter 3** concentrates on the evaluation of suitable experimental layouts (sampling schemes) for the *Stable Isotope Method*. It introduces a Monte Carlo simulation based meta-evaluation of different field-sampling strategies and their effect on the quality and informative value of the infiltration results obtained.

**Chapter 4** reviews the state of development achieved in the project. The validity of the tracer methods is evaluated in terms of a cross validation, where both the Pollutant Time Series Method and the Stable Isotope Method have been applied in parallel within a full scale experimental campaign. Validity is furthermore examined based on a Monte Carlo simulation based statistical survey of the acquired data material. Subsequently it is discussed how the suitability of local boundary conditions for the application of the methods in a catchment of interest can be assessed.

The thesis ends with a summary of the major findings in the **Conclusions** section.

Supporting information about the context and scope of the APUSS project are summarized in **Appendix A**.

Quantitative knowledge about infiltration is an expedient problem-oriented benchmark criterion for sewer rehabilitation where sewer pipes are situated below the groundwater table. However, potential losses of wastewater can occur in areas with low groundwater tables. Against this background, **Appendix B** supplies details on the QUEST-C method, which has been introduced as a suitable monitoring tool for sewer exfiltration.

Referencing information on the documentation of operation procedures for the performance of field experiments and software for data analysis are given in **Appendix C**.

This thesis is structured as a paper-dissertation. It is indicated which of the chapters have been submitted for publication.
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Quantification of Infiltration into Sewers based on Time Series of Pollutant Loads

Oliver Kracht and Willi Gujer

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Quantification of Infiltration into Sewers based on Time Series of Pollutant Loads

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Abstract
We introduce the concepts of a novel approach that allows for the quantification of infiltrating non-polluted waters by a combined analysis of time series of pollutant concentrations and discharged wastewater volume. The methodology is based on the use of automated sensors for the recording of the pollutant concentrations. This provides time series in a high temporal resolution that are suitable for a detailed data analysis and discussion on the underlying assumptions. The procedure is demonstrated on two examples from recent measurement campaigns in Switzerland.

Keywords: Infiltration; inflow; natural tracers; pollutants; sanitary sewers; time series

1 Introduction
Most conventional Infiltration / Inflow studies are based on different types of local water balances, or simply calculate the “parasitic discharge” on the assumption that the night-time minimum of the diurnal wastewater hydrograph equals the extraneous flows. These methods are subject to considerable uncertainties due to the underlying assumptions. A favourable tool would in contrast be a chemical tracer that directly differentiates the origin of water in the sewer (i.e. “real” foul sewage or infiltrating water). Yet, the artificial labelling of drinking water is prohibited (health and legal concerns) and the infiltrating water cannot be labelled either (environmental concerns and practicalities).

As artificial tracers are not feasible for measuring infiltration, specific inherent characteristics of the wastewater flow have to be used as natural indicators of the mixing processes. The presence of infiltration principally causes a dilution of pollutant concentrations in the wastewater. Some former studies therefore aimed to estimate infiltration ratios by combining observed average concentrations of organic matter or nutrients in the wastewater with assumptions about the daily amount of foul sewage generated per inhabitant. In more detailed investigations for instance Verbanck (1993) proposed the use of specific dissolved components like borates. However, the required accurate knowledge about the average per capita substance loadings and the average per capita foul sewage discharge must generally presumed to be rather unknown. These difficulties can principally be overcome by using the cyclic diurnal variations of the wastewater composition as an intrinsic tracer. Hager et al. (1984) suggested an iterative mathematical procedure to calculate the amount of infiltration based on pollutant concentrations of grab samples and flow measurements, typically taken in a series
over a 24 hours period. Some similar approaches that use a graphical analysis of the pollutant concentrations are occasionally used in practice. All these procedures principally require the hypothesis of a constant pollutant concentration in the foul sewage and of a constant amount of infiltration over the day. However, as the amount of information about the diurnal variation of pollutant concentrations that can be gained with grab samples is rather limited, it was yet difficult to test the validity of these underlying assumptions.

In this paper we present a novel concept for the quantification of infiltrating non-polluted waters. The fraction of infiltrating water is determined from a combined analysis of time series of suitable pollutant concentrations and discharged wastewater. The data analysis is based on the use of a mixing model that describes the concentration of pollutants in the wastewater in dependency on the quantity of wastewater flow ($Q$) and time ($t$). The employed parameter set contains variables for a time dependent description of the infiltration rate and the pollutant concentration in the foul sewage. The parameter values can be estimated by fitting a modelled time series of pollutant concentrations to the measured wastewater data.

## 2 Methodology

The chemical hydrograph separation of wastewater discharge requires a suitable natural tracer and an appropriately parameterized mixing model allowing for the identification of the different flow components.

A suitable bulk concentration parameter for this application is as an example the chemical oxygen demand (COD) of the wastewater, as in most cases its concentration in the parasitic water can assumed to be negligibly low. This avoids certain difficulties that would otherwise arise from the need to accurately characterize the mass contribution of tracer substance that is 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). Such use of an appropriate automatic measuring device provides time series in a high temporal resolution, which significantly improves the documentation of the wastewater composition dynamics compared to conventional discontinuous laboratory analysis. This is a basic requirement for our suggested data analysis.

Our brief definition of a mixing model starts from the general assumption that the amount of discharged wastewater is composed of a variable volume of real foul sewage and a certain volume of parasitic infiltration.

$$Q_{\text{wastewater}} = Q_{\text{foul sewage}} + Q_{\text{infiltration}}$$  \hspace{1cm} (Eq.1)
When observation intervals are broadened, the amount of infiltration usually reveals considerable temporal variations and seasonal trends. In particular, larger rain events cause a rise of the groundwater table and an increased degree of saturation in the vadose zone. Depending on the hydraulic retention capacity of the underground, this provokes a temporarily retarded additional amount of infiltration. The infiltrating water is therefore conceptually divided into a constant baseflow and an exponentially receding interflow component:

\[ Q_{\text{infiltration}} = Q_{\text{baseflow}} + Q_{\text{interflow}} = Q_{\text{baseflow}} + Q_{0,\text{interflow}} e^{-k_{\text{rec}} (t - t_{0,\text{interflow}})} \]  

(Eq.2)

where \( Q_{0,\text{interflow}} \) is the initial magnitude of interflow at the time \( t_{0,\text{interflow}} \) and \( k_{\text{rec}} \) is the associated recession constant.

Based on the recorded time series of \( Q_{\text{wastewater}} \) a modelled time series \( COD_{\text{wastewater,model}} \) is calculated according to Equation 3. By fitting \( COD_{\text{wastewater,model}} \) to the measured time series \( COD_{\text{wastewater}} \), the unknown parameters of \( Q_{\text{infiltration}} \) can now be estimated.

\[ COD_{\text{wastewater,model}} = \frac{(Q_{\text{wastewater}} - Q_{\text{infiltration}}) \cdot COD_{\text{foul-sewage}}}{Q_{\text{wastewater}}} \]  

(Eq.3)

As \( t_{0,\text{interflow}} \) and \( Q_{0,\text{interflow}} \) are not independently identifiable, one of these two parameters must be set to an appropriate fixed value beforehand. Since the occurrence of a rain event is usually well documented, this can most easily be done for \( t_{0,\text{interflow}} \). The corresponding value of \( Q_{0,\text{interflow}} \) can then be estimated by the optimization algorithm. This necessity is simply due to the characteristics of the exponential receding function and does not conflict the applicability of our approach: The estimated \( Q_{0,\text{interflow}} \) at a point in time \( t_{0,\text{interflow}} \) does not necessarily need to occur in the span of time of the real time series, but might even be positioned to a virtual point outside this range. The only demand on these two parameters is to allow for an adequate description of the interflow receding curve according to Equation 2 within the investigated span of time.

\( COD_{\text{foul-sewage}} \) can basically be introduced into the model as a constant value. However, it appeared to be advantageous to account for a time dependent variation of the pollutant concentration in the foul sewage. Favourable time dependent terms are in the first instance descriptions of diurnal or weekly periodicities. To further extend the flexibility, an additional dependency of \( COD_{\text{foul-sewage}} \) on the amount of currently discharged foul sewage \( Q_{\text{foul-sewage}} \) can also be considered. These extensions lead to the following equations:

\[ COD_{\text{foul-sewage}} = f(t, Q_{\text{foul-sewage}}) \]  

(Eq.4a)

For the analysis of the two demonstrated application studies (next chapter) we specified:

\[ COD_{\text{foul-sewage}} = f(t) + f(Q_{\text{foul-sewage}}) \]  

(Eq.4b)
In these examples the diurnal variation was sufficiently described by a simple periodic function with a periodicity of one day:

\[ f(t) = A \cdot \sin(freq \cdot 2 \cdot \pi \cdot (t - \text{phase})) \]  
\[ \text{(Eq.5)} \]

where \( A \) is the amplitude and \( freq \) is the frequency of the diurnal variation. \( freq \) was fixed to the value 1 day\(^{-1} \) in our examples. \( \text{phase} \) accounts for the constant phase shift of this periodicity relative to the daily 24 hours cycle. A weekly periodicity was not considered.

To describe the dependency of the foul sewage concentration on \( Q_{\text{foul-sewage}} \) we chose a conceptual polynomial formulation with constant coefficients \( a \), \( b \) and \( c \) in the form of

\[ f(Q_{\text{foul-sewage}}) = a + b \cdot Q_{\text{foul-sewage}} + c \cdot (Q_{\text{foul-sewage}})^2 \]  
\[ \text{(Eq.6)} \]

Due to the complicated temporal dynamics during rain events, we found it not viable to add on definitions for a flow component stemming from surface runoff. It must therefore be emphasized, that our concept is restricted to dry weather flow conditions.

3 Field Measurements

The application of our approach will be discussed based on two exemplary sets of field measurements. Figure 1a shows a cut out from a measurement campaign that was realised in spring 2003 in the influent of the Uster municipal wastewater treatment plant (CH). The plant serves a catchment with mixed infrastructure and approximately 35'000 inhabitants connected. The second campaign was conducted in autumn 2003 in a combined sewer (Figure 1b). This trunk sewer connects the village Rümlang (CH, approximately 5’600 inhabitants) to the regional treatment plant.

\[ \text{Figure 1. Field measurements at the two investigation sites Uster and Rümlang.} \]
COD-equivalents were measured in 2 minutes intervals with a submersible UV-VIS spectrometer (type spectro::lyser, s::can Messtechnik GmbH, Vienna) by means of light absorption spectra in the 200-750 nm range. In Uster the probe was installed in the influent channel downstream of the screening chamber. In Rümlang it was directly mounted inside the combined sewer pipe. UV-VIS absorbance is only an indirect method for the determination of COD-equivalents. The instrument was therefore ‘in-situ’ calibrated with a series of grab samples taken for laboratory analyses from the sewage stream passing the sensor. The wastewater flow was recorded with a temporal resolution of 1 minute with standard area-velocity flow meters (Doppler ultrasonic velocity and ultrasonic respectively air bubbler level measurement).

4 Data Analyses

The flow recordings from Uster had to be preprocessed with a moving average filter in order to remove higher frequency fluctuations overlaying the wastewater discharge curve. These modulations were caused by periodic backwater effects into the incoming sewer due to the treatment plant’s screening activity. No pre-treatment was required for the COD time series from Uster and both the flow and COD time series from Rümlang.

The mixing model according to Equations 1 to 6 was implemented using the simulation and data analysis tool AQUASIM (Reichert 1994). The parameter values were then estimated by minimizing the sum of weighted squares of the deviations between the measured COD-equivalents and the calculated model time series $COD_{wastewater, model}$ (Figures 2a and b).

![Figure 2. Comparison between the measured and modelled time series of COD-concentrations.](image-url)
This optimisation was performed with the constrained minimisation algorithm of AQUASIM (Reichert 1998), that is based on the secant method of Ralston and Jennrich (1978). In order to prevent biased estimates, two minor rain events occurring in Uster at 25.05. and 29.05.2003 have been excluded from the parameter fit. The quadratic term of Equation 6 was finally not used in the analyses of the two presented case studies, since it provided only negligible improvements of the fitting quality. The estimated parameter values, error approximates and correlation coefficients are assembled in tables 1 and 2.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Uster (WWTP)</th>
<th>Value</th>
<th>Standard Error</th>
<th>Rümlang (trunk sewer)</th>
<th>Value</th>
<th>Standard Error</th>
</tr>
</thead>
<tbody>
<tr>
<td>Q_{baseflow} [l·s^{-1}]</td>
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<td>2</td>
<td>5.3</td>
<td>0.8</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Q_{0,interflow} [l·s^{-1}]</td>
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<td>1</td>
<td>4.3</td>
<td>0.7</td>
<td></td>
<td></td>
</tr>
<tr>
<td>k_{rec} [day^{-1}]</td>
<td>0.47</td>
<td>0.02</td>
<td>0.12</td>
<td>0.04</td>
<td></td>
<td></td>
</tr>
<tr>
<td>l_{0,interflow}</td>
<td>fixed to 23.05.2003 12:00</td>
<td>fixed to 18.11.2003 00:00</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>a [mg·l^{-1}]</td>
<td>452</td>
<td>38</td>
<td>728</td>
<td>22</td>
<td></td>
<td></td>
</tr>
<tr>
<td>b [mg·s·l^{-2}]</td>
<td>3.1</td>
<td>0.3</td>
<td>3.7</td>
<td>1.0</td>
<td></td>
<td></td>
</tr>
<tr>
<td>c [mg·s^{-2}·l^{-1}]</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td></td>
<td></td>
</tr>
<tr>
<td>A [mg·l^{-1}]</td>
<td>148</td>
<td>6</td>
<td>100</td>
<td>5</td>
<td></td>
<td></td>
</tr>
<tr>
<td>phase [day]</td>
<td>0.28</td>
<td>0.01</td>
<td>0.29</td>
<td>0.01</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Table 1. Estimated parameter values from the two example series

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Uster</th>
<th>Value</th>
<th>Standard Error</th>
<th>Rümlang</th>
<th>Value</th>
<th>Standard Error</th>
</tr>
</thead>
<tbody>
<tr>
<td>Q_{baseflow}</td>
<td>1</td>
<td>0.06</td>
<td>0.31</td>
<td>0.88</td>
<td>-0.70</td>
<td>0.45</td>
</tr>
<tr>
<td>Q_{0,interflow}</td>
<td>0.06</td>
<td>1</td>
<td>0.32</td>
<td>0.15</td>
<td>-0.18</td>
<td>0.14</td>
</tr>
<tr>
<td>k_{rec}</td>
<td>0.31</td>
<td>0.32</td>
<td>1</td>
<td>0.06</td>
<td>-0.08</td>
<td>0.07</td>
</tr>
<tr>
<td>a</td>
<td>0.88</td>
<td>0.15</td>
<td>0.06</td>
<td>1</td>
<td>-0.93</td>
<td>0.56</td>
</tr>
<tr>
<td>b</td>
<td>-0.70</td>
<td>-0.18</td>
<td>-0.08</td>
<td>-0.93</td>
<td>1</td>
<td>-0.62</td>
</tr>
<tr>
<td>A</td>
<td>0.45</td>
<td>0.14</td>
<td>0.07</td>
<td>0.56</td>
<td>-0.62</td>
<td>1</td>
</tr>
<tr>
<td>phase</td>
<td>-0.20</td>
<td>0.03</td>
<td>0.10</td>
<td>-0.28</td>
<td>0.26</td>
<td>0.001</td>
</tr>
</tbody>
</table>

Table 2. Estimated correlation matrices

Standard errors and correlation coefficients have been calculated by means of a finite difference approximation for the derivatives of the parameter estimation with respect to the accumulated residuals. This simplification is supposed to be reasonable for the two demonstrated data sets. However, as the employed mixing model contains distinct nonlinear
structures, it must be emphasized that such linearization can cause inaccurate error estimates in cases of larger uncertainty ranges of the parameters. The error approximation is yet furthermore possibly affected by the occurrence of serially correlated residuals between measured and modelled data. This is assumed to be implicitly due to the high temporal resolution of the automated in-line measurements, but can lead to a certain underestimation of the standard errors.

5 Results and Discussion

The modelled time series for COD provides a good reproduction of the measured data and can satisfactorily explain the major elements of the wastewater pollutograph. The parameters describing $Q_{\text{infiltration}}$ were successfully identifiable from the measured time series $Q_{\text{wastewater}}$ and $COD_{\text{wastewater}}$ without requiring additional external information.

Figures 3a and 3b illustrate the resulting separation of the wastewater discharge into the three flow components parasitic baseflow (constant), parasitic interflow (exponentially receding) and real foul sewage (diurnal variation). In both examples strong rainfalls (not shown) before 23.05.2003 (Uster) and 18.11.2003 (Rümlang) clearly cause an interflow component of infiltration that is persisting over a considerable number of days.

The partition was calculated from Equation 2 by means of a Monte Carlo Simulation assuming normally distributed parameter estimates and standard errors for $Q_{\text{baseflow}}$, $Q_{\text{interflow}}$ and $k_{\text{rec}}$ as given in Table 1. The correlations according to table 2 had to be considered in the applied sampling routine. Shaded areas show the mean values. Full lines indicate the 2.5th and 97.5th percentile of the resulting cumulative probability density functions derived from 1000 simulation runs. It is remarkable that the separation between total infiltration and foul
sewage is apparently more precise than the separation between baseflow and interflow. For exemplification, the total volumes of infiltration that were discharged within the span of a week are listed in Table 3. The infiltration ratio \( X_{\text{infiltration}} \) relates the volumes of infiltration to the total amount of wastewater discharge within the considered span of time.

### Table 3. Accumulated results (2.5th to 97.5th percentile ranges)

<table>
<thead>
<tr>
<th>Considered time span</th>
<th>( \sum Q_{\text{infiltration}} )</th>
<th>average ( X_{\text{infiltration}} )</th>
</tr>
</thead>
<tbody>
<tr>
<td>Uster (WWTP)</td>
<td>24.05. to 30.05.2003</td>
<td>66 - 70 ( \cdot 10^3 ) m(^3)</td>
</tr>
<tr>
<td>Rümlang (trunk sewer)</td>
<td>20.11. to 26.11.2003</td>
<td>4.4 - 4.9 ( \cdot 10^3 ) m(^3)</td>
</tr>
</tbody>
</table>

It should be pointed out that the approximated standard errors given in Table 1 and the uncertainty ranges specified in Figure 3 and Table 3 solely account for the effect of random errors that are superimposing the measured time series. However, the validity of our parameter estimates might rather be influenced by possible systematic measurement errors that are affecting the accuracy of the measured data series \( Q_{\text{wastewater}} \) and \( \text{COD}_{\text{wastewater}} \). The practical relevance of such uncertainties that are stemming from possible systematic errors embedded in the measurements of \( Q_{\text{wastewater}} \) and \( \text{COD}_{\text{wastewater}} \) will also depend on the intended use of the examination outcomes. As an example, the over- or underestimation of \( Q_{\text{wastewater}} \) will of course lead to an over- or underestimation of \( Q_{\text{infiltration}} \). However, if the demanded result of the examination is not the absolute value of \( Q_{\text{infiltration}} \), but rather the infiltration ratio \( X_{\text{infiltration}} \), the larger part of the error will in turn be crossed out. Nevertheless, even in this case some part of error contribution will remain, due to the partly nonlinear behaviour of the suggested mixing model.

To conduct a rough check of the robustness of our estimates with respect to possible systematic errors embedded in the measured variables, a brief set of error scenarios was calculated. For both measured variables a hypothetical constant offset error \( \alpha \) and a relative error \( \beta \) according to Equations 7a and 7b were assumed. The parameter estimation was repeated and the infiltration totals and average infiltration ratios for the time span of a week were recalculated based on the new estimates.

\[
\text{COD}_{\text{wastewater,measured}} = \alpha + \beta \cdot \text{COD}_{\text{wastewater,real}} \quad \text{(Eq.7a)}
\]

\[
Q_{\text{wastewater,measured}} = \alpha + \beta \cdot Q_{\text{wastewater,real}} \quad \text{(Eq.7b)}
\]

A hypothetical offset error of 45 mg/l for \( \text{COD}_{\text{wastewater}} \) equals 20% of the average COD concentration measured in Uster and 10% of the average value in Rümlang respectively. Offset errors for \( Q_{\text{wastewater}} \) were chosen as 15% of the minimum value occurring in the time
Series. Table 4 summarizes the observed effects on the calculated mean values for the final results.

Systematic relative errors embedded in COD\textsubscript{wastewater} apparently do not affect either $\sum Q_{\text{infiltration}}$ or $X_{\text{infiltration}}$. A relative error on $Q_{\text{wastewater}}$ causes a recognisable effect on $\sum Q_{\text{infiltration}}$ only. More critical attention has to be spent to possible systematic offset errors in the concentration and flow measurements, as these can considerably influence all estimates. However, the assumed magnitudes for systematic measurement errors given in Table 4 are supposed to reflect a rather pessimistic hypothesis of the achievable measurement quality, especially with regard to the concentration measurements.

**Table 4.** Potential influence of systematic measurement errors (scenarios with minimal and maximal means)

<table>
<thead>
<tr>
<th>Scenario</th>
<th>Description</th>
<th>Uster (WWTP)</th>
<th>Rümlang (trunk sewer)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>24.05. to 30.05.2003</td>
<td>20.11. to 26.11.2003</td>
</tr>
<tr>
<td>1</td>
<td>without systematic measurement errors</td>
<td>$\sum Q_{\text{infiltration}}$</td>
<td>68.5 ·10$^3$ m$^3$</td>
</tr>
<tr>
<td></td>
<td></td>
<td>average $X_{\text{infiltration}}$</td>
<td>64 %</td>
</tr>
<tr>
<td>2</td>
<td>$\alpha = +/- 45$ mg/l offset error on COD\textsubscript{wastewater}</td>
<td>59.3 / 77.2</td>
<td>4.2 / 4.9</td>
</tr>
<tr>
<td></td>
<td></td>
<td>55 / 72</td>
<td>33 / 38</td>
</tr>
<tr>
<td>3</td>
<td>$\beta = +/- 15%$ relative error on COD\textsubscript{wastewater}</td>
<td>68.5 / 68.5</td>
<td>4.5 / 4.5</td>
</tr>
<tr>
<td></td>
<td></td>
<td>64 / 64</td>
<td>36 / 36</td>
</tr>
<tr>
<td>4</td>
<td>$\alpha = +/- 15%$ of min. value offset error on $Q_{\text{wastewater}}$</td>
<td>59.3 / 77.6</td>
<td>3.9 / 5.2</td>
</tr>
<tr>
<td></td>
<td></td>
<td>60 / 67</td>
<td>32 / 39</td>
</tr>
<tr>
<td>5</td>
<td>$\beta = +/- 15%$ relative error on $Q_{\text{wastewater}}$</td>
<td>58.2 / 78.7</td>
<td>3.9 / 5.2</td>
</tr>
<tr>
<td></td>
<td></td>
<td>64 / 63</td>
<td>36 / 36</td>
</tr>
<tr>
<td>6</td>
<td>worst combination of scenarios 2 to 5</td>
<td>42.1 / 99.3</td>
<td>3.0 / 6.4</td>
</tr>
<tr>
<td></td>
<td></td>
<td>51 / 75</td>
<td>30 / 42</td>
</tr>
</tbody>
</table>

6 Conclusions

A novel concept for the chemical separation of wastewater hydrographs was introduced. The methodology enables the quantification of infiltrating non-polluted waters based on measured time series of wastewater flow and pollutant concentrations without requiring additional external information. The procedure was demonstrated with two case studies that were based on the use of a submersible spectrometer probe for the recording of COD-equivalents.

The method is very flexible and supports our understanding of the dynamics of infiltration processes in urban sewer networks. Compared with earlier tracer based approaches it does not require the hypothesis of a stationary behaviour of the diurnal hydrograph and pollutograph.

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The capability to consider natural storage and interflow phenomena significantly broadens its practical applicability. It particularly simplifies the realisation of measurements in rainy seasons, when infiltration rates are typically increased due to elevated groundwater tables. Compared to solely hydrograph based approaches, our method avoids external assumptions like that the hydrograph’s night-time minimum equals the “parasitic discharge”. Such simplifications will become more and more problematic in the future, as technical progress along with energy and water consumption policies progressively effectuates water consuming activities to be carried out during night time. Furthermore, sewage systems are growing with expanding agglomerations, which will cause an increasingly phase shifted superimposition of the hydrographs of wastewater arriving from different parts of a city.

We therefore expect our approach to be of a high interest for future applications. The most critical benchmark for a general applicability will be the attainable minimisation of the overall uncertainty in the infiltration estimates. As demonstrated, our next development steps will therefore have to be focussed on the evaluation and possible detection and prevention of systematic measurement errors on the employed input variables.

Acknowledgements

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References


CHAPTER 2

A Stable Isotope Approach for the Quantification of Sewer Infiltration

Oliver Kracht, Markus Gresch and Willi Gujer

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A Stable Isotope Approach for the Quantification of Sewer Infiltration

Oliver Kracht, Markus Gresch and Willi Gujer

Abstract
Extraneous flows in wastewater collection systems are conventionally evaluated solely on the consideration of discharge hydrographs, which often involves a great degree of subjectivity and oversimplification. In order to obtain reliable information on the material fluxes within the urban environment, the use of intrinsic tracers can be the optimal choice. We demonstrate the successful use of naturally occurring stable isotopes of water ($^{18}$O/$^{16}$O and D/H) to accurately quantify extraneous discharge of groundwater in a combined sewer network. Fresh water supply from a distant hydrological regime provided usable isotopic separations between drinking water (proxy for “real” foul sewage) and local groundwater (proxy for sewer infiltration) of 1.8‰ in $\delta^{18}$O and 11.7‰ in $\delta^2$H. Diurnal variation of wastewater isotopic composition reflected both the varying rates of foul sewage production and irregular dispersion effects in the pipe network. The latter suggest the existence of larger cumulative backwater zone volumes, which have not yet been attended to. Infiltrating groundwater contributed 39% (95%-confidence interval = ± 2.5%) of the total daily dry weather wastewater discharge. The paper discusses all major aspects relevant for practical application of the method. It presents a comprehensive framework for uncertainty analysis and details on the detection and discrimination of possibly interfering effects.

1 Introduction
Sustainable strategies for the planning, operation and maintenance of urban drainage networks require adequate knowledge about their functional behaviour and interaction with the surrounding aquifers. However, knowledge about the water-tightness of a sewer system is often poor and investments are frequently based on uncertain information and limited data. Whilst larger punctual sewer leaks can be located relatively easily by closed-circuit television or walk-through inspections, sewer infiltration typically occurs through a vast number of smaller defects, which are rather "diffusively" distributed throughout larger areas of a catchment. Practitioners therefore often address this issue by calculating the “parasitic discharge” on the assumption that the night time minimum in the diurnal wastewater hydrograph is equal to the extraneous flows. In the context of nowadays growing agglomerations this practice is considered oversimplified and likely leads to erroneous results. However, to date little research has been devoted to the use of direct natural tracers as an alternative for the quantification of extraneous discharges (Hager et al., 1984; Hager et al., 1985; Kracht and Gujer, 2005; Verbanck, 1993).
2 Outline of Methodology

Under dry weather conditions, wastewater typically has two sources of origin: (i) drinking water the properties of which have changed through use in households, trade and industry (i.e. “real” foul sewage) and (ii) extraneous water from groundwater infiltration, public fountains and drainage pipes:

\[ Q_{\text{Wastewater}} = Q_{\text{Foul Sewage}} + Q_{\text{Infiltration}} \]  

(Eq.1)

where \( Q \) denotes discharge. As this study is limited to dry weather conditions, the inflow of surface runoff during wet weather is not discussed herein.

The branched structure of sewer networks impedes direct chemical characterization of the mixing endmembers at the catchment or even subcatchment scale: When they enter the sewer system, foul sewage and infiltration are mixed and any separate analysis is hindered. Thus, endmember characterization must be performed indirectly, i.e. by examining drinking water and local groundwater. As a second constraint, sewer infiltration cannot be measured using artificial tracers: The artificial labeling of drinking water is prohibited (health and legal concerns) and homogenous distribution of tracers throughout an entire aquifer is neither feasible nor desirable for environmental reasons. Instead, specific natural water characteristics have to be used. However, in sewer systems it is typically difficult to find suitable inherent markers, since sewage chemistry usually exhibits substantial background fluctuations that obscure the natural tracer signal.

In this study we use the stable isotope composition of mains water (proxy for the foul sewage) and local groundwater (proxy for the infiltrating water) as direct natural tracers for the quantification of infiltration into a sewer system. On larger scales, the isotopic composition of water is widely controlled by the continental setting and topographic elevation of the region where ground water and surface water are recharged by precipitation (Clark and Fritz, 1997; Craig and Gordon, 1965; Fritz and Fontes, 1980; Gat and Gonfiantini, 1981; Gat et al., 2001; Hoefs, 2004; Mebus, 2001; Rozanski et al., 1993; Rozanski et al., 2001). In reference to our application, suitable isotopic separations can exist, where drinking water originates from a distant hydrological regime, whereas sewer infiltration stems from groundwater that is recharged by local precipitation. Assuming a binary mixing system, the fraction of wastewater that stems from infiltration (infiltration ratio \( X_{\text{Infiltration}} \)) and the corresponding amount of infiltration discharge are given by:

\[ X_{\text{Infiltration}}(t) = \frac{\delta_{\text{Wastewater}}(t) - \delta_{\text{Drinking Water}}(t)}{\delta_{\text{Groundwater}}(t) - \delta_{\text{Drinking Water}}(t)} \]  

(Eq.2)  

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

(Eq.3)

where \( \delta_i \) are the respective isotopic compositions.
An infiltration study based on isotopic tracers typically consists of the following three steps: (i) A survey on the stable isotope composition of potentially infiltrating waters, which is essentially based on detailed hydrological investigations that define the principal hydraulic interactions and pathways in the catchment. (ii) A survey on the stable isotope composition of the local drinking water. This should include suitable measures to continuously control the homogeneity of mains water composition. (iii) A series of wastewater samples that covers one or more diurnal cycles at each measuring point. Wastewater discharge must be measured in parallel to calculate infiltration totals and to correctly weight infiltration ratios over the desired observation time (e.g. 24 h):

\[ Q_{\text{infiltration, 24h-total}} = \int_{24h} X_{\text{infiltration}}(t) \cdot Q_{\text{wastewater}}(t) \cdot dt \]  

(Eq.4) \[ X_{\text{infiltration, 24h-total}} = \frac{Q_{\text{infiltration, 24h-total}}}{\int_{24h} Q_{\text{wastewater}}(t) \cdot dt} \]  

(Eq.5)

3 Site Description and Hydrological Setting

Our study was carried out in the fall of 2003 in Rümlang (CH), a community of about 5'400 inhabitants located in the Glatt valley on the north-eastern boarder of Zurich. Rümlang has a mixed infrastructure with no predominant type of industry. The built-up area amounts to 292 ha, the immediate surrounding is mainly used for agriculture. The public sewer system has a total length of 23.1 km. The structural state of the sewer network is considered to be representative for average conditions in Switzerland. Its specific setting makes the catchment suitable for the use of the stable isotope approach: The village is predominantly served with drinking water from Lake Zurich (via public pipeline systems), a large fraction of which stems from precipitation in the Alps. In contrast, local groundwater originates from precipitation at a moderate altitude and therefore shows a distinctly different isotopic composition.

The deeper underground of Rümlang is formed of tertiary marls and sandstones from the Upper Freshwater Molasse. These molasses rocks crop out in the western regions of the terrain and form the headwaters for several small creeks crossing the broader village area. Due to its proximity to the Alps, the area was repeatedly ice covered during the Pleistocene. The Glatt valley was thereby principally formed by glacial erosion during the second last glacial period (Riss). Meltwater deposits of the advancing and retreating glaciers from the last glacial period (Würm) cover the main part of the area under investigation (Haldimann, 1978; Jäckli, 1962). This unlithified glacial unit is primarily composed of gravels and sands. It forms a local aquifer, which is partly covered and intersected by poorly permeable moraine sediments towards the bottom of the valley. The moraine deposits tend to function as a hydraulic barrier at the lower boundary of the aquifer, causing an artesian overflow that was observable up until the beginning of the 20th century (Kempf et al., 1986). However, the build up of artificial drainages and the productive use of groundwater by two pumping wells caused the
hydraulic heads to drop down. The aquifer is unconfined nowadays. Both terrain and bedrock surface are principally inclined towards the north east in the direction of the valley bottom. Consequently the river Glatt forms the natural receiving water for all surface runoff and subsurface discharge in the area. This topographic constellation prevents infiltration of river water from influencing the local groundwater in the relevant areas of the sewer network. This ensures that all relevant shallow groundwater in Rümlang entirely stems from local precipitation.

Regional climate is temperate with annual precipitation averaging around 1100 mm in the long term records (MeteoSwiss stations Kloten and Reckenholz 1998-2003). 2003 was indeed rated as exceptionally warm and dry (only around 850 mm/a). However, strong rainfall in October 2003 exceeded the long-term average by more than 100%. Yet our principal experiment conducted in November 2003 was again preceded by three comparably dry weeks.

![Figure 1. Delineated watershed of the Lake Zurich catchment; Rümlang investigation site. Reproduction of GIS-data authorized by Swisstopo (BA068125).](image)

### 4 Pre-Investigations and Pilot Studies

Based on the 25 m grid Swiss digital elevation model DHM25, the mean altitude of Lake Zurich’s hydrological catchment was calculated to be around 1220 m asl (Figure 1). In contrast, Rümlang’s residential area and sewer system is situated in a moderate altitude, predominantly between 430 to 460 m asl. Highest elevations of the village’s natural hydrological catchment do not exceed 500 m asl. Due to Switzerland’s large topographic variations, local
altitude effects for $\delta^{18}$O and $\delta^2$H values in precipitation can vary significantly (Schotterer et al., 1995). However, based on a broader data set of groundwater and surface water from the Swiss Alps and the Swiss Plateau Siegenthaler et al. (1983) inferred an average decrease of 0.22‰ per 100 m rise for $\delta^{18}$O. Similarly, a decrease of 0.26‰ / 100m was stated based on precipitation samples (Siegenthaler and Oeschger, 1980). More recent precipitation data of the Swiss National Network for Isotopes in the Water Cycle show an approximate decrease of 0.2‰ / 100m for $\delta^{18}$O and 1.2‰ / 100m for $\delta^2$H respectively (Schürch et al., 2003). Under this assumption, a suitable isotopic separation between drinking water and infiltrating groundwater between 1.6 to 2.0‰ in $\delta^{18}$O and around 10‰ in $\delta^2$H was expected for our experimental catchment.

A pilot survey conducted in fall 2002 revealed a mean isotopic composition of the drinking water of -11.6‰ ($\delta^{18}$O) and -81.5‰ ($\delta^2$H). Mean groundwater composition was estimated to be -9.7‰ ($\delta^{18}$O) and -69‰ ($\delta^2$H), reflecting the comparably heavier isotopic signature of local precipitation. These results positively confirmed our forecast on the usable isotopic differences. However, as a drawback we observed that the quality of the tracer signal in the water mains was considerably affected by varying amounts of local groundwater supplied to the drinking water network from a municipal well and spring water tappings.

The temporal stability of groundwater isotopic composition was screened by monthly samples (July 2003 to April 2004) taken from two observation points. The observed variation falls within the range of measurement uncertainty and is considered to be negligible for the comparable short time scale of our actual experiment.

5 Experimental Section

Operation and monitoring of the tracer experiment

Optimization of the experimental boundary conditions required close cooperation with the local stakeholders as well as detailed investigations into the adjacent water supply systems. To improve the quality of the tracer signal, the entire water supply system of the catchment was temporarily changed such that only water from Lake Zurich was delivered. In parallel, homogeneity of the drinking water was continuously controlled by automatic conductivity measurements installed in the water mains. This was suitable because low mineralized drinking water exploited from Lake Zurich (300 µS/cm) was clearly distinguishable from local groundwater productions (about 800 µS/cm). The progression and effects of the measures undertaken can be followed in Figure 2: The graphs show conductivity time series recorded at the junctions of the local system with the neighbouring networks and at selected places within the local waterlines themselves. In routine operation, intermittent pumping from the local supplies caused substantial diurnal fluctuations in the mains water composition.
Therefore local production was stopped from 03.11.2003 to 23.11.2003. Maintaining uninterrupted drinking water supply (≈ 1640 m$^3$/day in total) required increasing the pipeline-based infeed of Zurich lake water by around 370 m$^3$/day. However, smaller compositional fluctuations remained visible until a cross connection within the supra-regional pipeline system (which was indeed unnoticed before) was closed on 14.11.2003 (charts a and b). Another critical task was the effective flushing of the local reservoir storage tank, which due to its comparably large volume (2900 m$^3$) significantly delayed the system’s adaptation to the modified operation (chart d). Sampling of drinking water and wastewater for isotopic analyses was performed under stabilized conditions from 20.11.2003 to 22.11.2003. To inspect if rising of groundwater tables (i.e. caused by the shut down of the municipal production well) had an influence on the amount of groundwater infiltrating into the sewers, an array of 16 piezometers was carefully monitored throughout the whole experiment.

Figure 2. Monitoring of electrical conductivity in the drinking water network. The catchment is supplied with drinking water from Lake Zurich (300 µS/cm) as well as from local groundwater productions (about 800 µS/cm). Boundary conditions were optimized for our measuring campaign, conducted from 20.11. to 22.11.2003 (marked by vertical arrow). a, b and c: Devices installed at conjunctions with regional pipelines and neighbouring drinking water systems. d: In-/outlet of the local reservoir-tank. e and f: Devices installed within the Rümlang local network.
Wastewater sampling and discharge metering

All wastewater sampling and discharge measurements were conducted in an intercepting sewer that connects the village to the regional wastewater treatment plant. A continuous series of 29 hourly samples were taken from 20.11.2003 22:00 to 22.11.2003 02:00. Water level (h) and flow velocity (v) were recorded in 1 minute intervals using a combined air-bubbler and ultrasonic Doppler probe. From these, wastewater discharge calculates to:

\[ Q = v \cdot \frac{1}{2} \left( \frac{d}{2} \right)^2 \cdot (\varphi \cdot \sin \varphi) \quad (\text{Eq.6}) \]

where \(d\) is the diameter of the circular sewer pipe and \(\varphi\) is derived from:

\[ \varphi = 4 \cdot \arcsin \left( \frac{h}{d} \right) \quad (\text{Eq.7}) \]

The average dry weather wastewater discharge in November 2003 was 2720 m\(^3\)/day for the whole village. For technical reasons, the wastewater data presented in this text were taken from a point in the interceptor that only covers around three quarters of the whole catchment (average discharge of 1870 m\(^3\)/day).

Groundwater and drinking water sampling

The groundwater sampling scheme for the principal experiment was aimed to cover the entire range of shallow groundwater that, from hydrological a priori knowledge, potentially represents sources of infiltration into the local sewer system. 32 samples taken from monitoring wells (various screened sections from 2 to 10 m below ground), two production wells (screened sections down to 17 m below ground), spring tappings, surface water sites (local creeks) and artificial draining systems were considered. Groundwater well samples were collected using a portable pump, following purging a minimum of three well casing volumes until electrical conductivity, pH, dissolved oxygen, and oxidation-reduction potential parameters stabilized. Drinking water samples were taken from seven continuously purged observation points in the mains network, each of which was sampled 3 times in the course of the experiment.

Analytical methods

Samples were filtrated through 0.7 \(\mu\)m glass fiber filters and stored at 5\(^\circ\) C. Stable isotopes analyses were conducted in the Laboratory of Atmospheric Chemistry at the Paul Scherrer Institut (CH). We used automated sample preparation devices directly coupled to the dual-inlet system of a Finnigan MAT Delta-S gas ratio mass spectrometer. \(\delta^{18}\)O values were measured using the conventional water-CO\(_2\) equilibration technique (Epstein and Mayeda, 1953). \(\delta^{2}\)H values were measured by reducing the water on chromium at 850\(^\circ\)C and direct
transfer of H$_2$ into the mass spectrometer (Gehre et al., 1996a; Gehre et al., 1996b). All values are cited in the δ-notation in reference to the international standard VSMOW (International Atomic Energy Agency, 1995). Calibrations were based on the regular analysis of standards distributed by the IAEA (IAEA OH5-8). The internal precision of the analytical methods was estimated by replicate analyses of in-house working standards along with the sample series and amounted to 0.08‰ (δ$^{18}$O) and 0.8‰ (δ$^2$H) respectively (1σ-level).

Computational methods

All statistical evaluations were performed with the R software environment for statistical computing (R Development Core Team, 2005; Ribeiro and Diggle, 2001) using in-house scripts written by the authors.

6 Results and Discussion

Progression of the tracer experiment

Interfering fluctuations of the drinking water composition were successfully suppressed by our operational adjustments applied to the water supply network (Figure 2). This enabled us to perform the sampling campaign under well stabilized conditions. The long term wastewater discharge record revealed no anomalies or discontinuities before, during or after our experiment. A smaller increase in groundwater levels was limited to the proximate surrounding of the shut down production well and did not exert influence on the sewer system. We therefore consider the general discharge situation to be unaffected by our experimental modifications. The wastewater sampling was preceded by three comparably dry weeks. Last rainfall occurred three days before the first sample was taken.

Suitability of the natural tracer signal

Our experiment primarily aimed to distinguish “real” foul sewage (the transfer product of drinking water used in households and industries) from parasitic sewer infiltration (basically stemming from shallow groundwater). Figure 3 gives an overview of δ$^{18}$O and δ$^2$H values of all samples collected during our principal experiment. The stable isotopes composition of both mixing endmembers assumed is sufficiently distinct, with mean drinking water values plotting at δ$^{18}$O = -11.31‰ / δ$^2$H = -80.3‰ and mean groundwater values plotting at δ$^{18}$O = -9.54‰ / δ$^2$H = -68.6‰ (a single groundwater outlier plotting at δ$^{18}$O = -8.90‰ and δ$^2$H = -64.1‰ is not shown here and excluded from further evaluation). Groundwater δ$^{18}$O and δ$^2$H values are in particular clearly correlated and plot close to the Global Meteoric Water Line (Rozanski et al., 1993), which generally reflects the meteoric origin of these waters (note that the meteoric water line is not shown in Figure 3).
All wastewater samples plot close to the mixing line between drinking water and groundwater. This positively confirms the applicability of our approach with respect to two important points: First, we observe no indication of possibly deviating sources of sewer infiltration or sewage (these would, however, be undetected when plotting on the mixing line). Second, the data indicate no discernible alteration by evaporative enrichment. For comparison, modeled trends for a possible evaporative enrichment of the drinking water endmember are shown. The scenarios were calculated by classical Craig-Gordon type evaporation models (Craig and Gordon, 1965; Gonfiantini, 1986), which take into account the effects of kinetic isotope fractionation caused by a diffusive sublayer at the air/water interface. Evaporative enrichment of the wastewater or one of its mixing endmembers would have caused isotopic values to clearly deviate from the actual mixing line. The critical question of possible evaporative enrichment was additionally addressed by a smaller set of in-household tests (data not shown here). Indeed, no significant effects were observed for major water consuming activities like bathing or showering. Evaporative enrichment was significant for cooking only, which, however, contributes only negligibly to the total amount of wastewater.

Figure 3. Isotopic characterization of ground water (proxy for sewer infiltration), drinking water (proxy for foul sewage) and wastewater. Large grey circles display the corresponding arithmetic means, for which a best fit straight mixing line was calculated by linear regression. Trends for a possible evaporative enrichment of the drinking water were modelled on the basis of classical “Craig Gordon” type evaporation models. The envelope is covering assumptions about the relative humidity of the free atmosphere from h = 0% (upper line) to h = 95% (lower line). Numbers refer to the modelled fraction of drinking water lost by evaporation.

**Diurnal patterns and hydrograph separation**

Municipal wastewater flows generally follow characteristic diurnal patterns that are primarily controlled by the actual production of domestic and industrial sewage. Wastewater isotopic
composition changes should therefore reflect the varying proportions of foul sewage and groundwater infiltration contributing to the total wastewater discharge. Figure 4 compares the time series of measured wastewater $\delta^{18}$O and $\delta^2$H values to the endmember ranges estimated for groundwater and drinking water (charts a and b). The isotopic composition of wastewater systematically varies within the boundaries spanned by the two mixing endmembers. As expected, wastewater is enriched in $^{18}$O and $^2$H at night, indicating a higher fraction of isotopically heavier infiltrating water being present in the sewer. During the day the wastewater shows depleted isotope values, due to the larger amount of isotopically lighter foul sewage being discharged into the sewer system by the consumers. Note that the 2.5th/97.5th percentiles for the empirically observed isotopic compositions of groundwater and drinking water samples are marked by dashed lines, whereas fully colored areas reflect the 95%-confidence intervals for the expected mean values of these mixing endmembers. The underlying statistical concept for this latter estimate will be discussed further below.

Applying Equations 2 and 3, the wastewater hydrograph is subdivided into its elementary components “foul sewage” and “infiltrating groundwater”. Figure 4c presents the resulting hydrograph separation. Integration of $Q_{\text{Infiltration}}$ over 24 hours gives a total discharge of parasitic infiltrations of 711 m$^3$/day. Related to the measured total wastewater discharge of 1834 m$^3$/day, this corresponds to an integrated infiltration ratio of $X_{\text{Infiltration}, 24h-total} = 0.39$.

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**Figure 4.** Time series of wastewater isotopic composition and resulting hydrograph separation. Open circles in chart a and b indicate the isotopic composition of individual wastewater samples. Bars specify the combined measurement uncertainty for a single sample (2σ-level). Coloured areas define the 95%-confidence intervals for the mean isotopic composition of drinking water and infiltrating groundwater derived by the Monte Carlo error propagation. For comparison, dashed lines mark the 2.5th and 97.5th percentiles of the empirical distribution of groundwater samples (n=31) and drinking water samples (n=21). Chart c presents the resulting decomposition of the wastewater hydrograph into foul sewage and infiltration. Diagonal shading marks the 95%-confidence range for the infiltration discharge.
**Statistical analysis**

The uncertainty of the infiltration results principally depends on: (i) the accuracy of our estimates for the mean isotopic composition of foul sewage and infiltrating water, (ii) the spatial positioning of infiltration inflows, (iii) the accuracy of our estimates for the isotopic composition of individual wastewater samples and (iv) possible systematic errors embedded in the discharge measurements.

The set of 21 samples taken from the drinking water network has a standard deviation of 0.06‰ (δ¹⁸O) and 0.6‰ (δ²H) respectively. This scatter lies within the uncertainty range of the laboratory methods applied. We infer that the intrinsic variability of drinking water in the network was negligible during our experiment. This assumption is supported by evaluating the mains water electrical conductivity time series: Considering the conductivity values and corresponding isotopic endmember compositions of lake water and local groundwater exploitations, the data suggest that mains water variability did not exceed standard deviations of 0.006‰ (δ¹⁸O) and 0.04‰ (δ²H).

In contrast, the set of 31 groundwater samples reveals a standard deviation of 0.17‰ (δ¹⁸O) and 1.4‰ (δ²H), indicating that natural variability is not negligible for this mixing endmember. Therefore, the accuracy by which the mean isotopic composition of infiltrating water can be estimated additionally depends on the nature of hydraulic interlinking between sewer pipes and aquifer. This begs the question, to what extent the observed groundwater variability averages out as a consequence of the multitude of sewer defects with active groundwater seepage and their spatial distribution relative to the aquifer. In the general case, the answer requires a suitable model describing how information on the spatial distribution of groundwater isotopic composition can be combined with information on the spatial distribution of relevant sewer damages.

The groundwater data were geostatistically analyzed to evaluate the spatial variability of isotopic compositions. Figure 5 displays an omnidirectional empirical variogram calculated for both the δ¹⁸O and δ²H data. To facilitate the comparability of both datasets, semivariance was scaled by the sample variance. The diagram exhibits random, pure nugget variance, thus indicating no spatial autocorrelations at the scale of observation that can be resolved with our data (the first lag class interval in Figure 5 corresponds to a mean spacing of 58.4 m). This finding was likewise confirmed by a sequence of directional variograms calculated to take into account possible anisotropic effects (not shown here). As a consequence, interpolation techniques can not aid in establishing a refined topological relationship for our datasets. With respect to further data processing, we conclude that the spatial distribution of groundwater isotopic compositions is best expressed as a random field that has no spatial component: The sample mean and variance of the regionalized variable observed here fully estimate every point in the random field. The lack of spatial structure in the groundwater isotope data implies
that the positioning of defects in the sewer lines is adequately described as a pure random variable, too. Hence, the accuracy of our estimate for the mean isotopic composition of infiltrating water is supposed to be basically controlled by the overall perforation state of the sewer pipes (i.e. infiltration points per kilometer of sewer line). These interrelationships will be discussed in more detail below.

![Figure 5. Omnidirectional empirical variograms for the isotopic composition of groundwater samples. The maximum sampling point spacing of 3500 m was subdivided in 30 uniform lag classes (bins). The diagram displays mean semivariance values calculated for each lag class.](image)

We assessed the combined measurement uncertainty of our final infiltration results by means of nonlinear error propagation using the following Monte Carlo simulation scheme (Figure 6):

In the first step, 10000 independent virtual data sets are generated to simulate stochastic realisations of the random variables underlying the observed measurement results. Table 1 summarizes all procedures and specifications used for stochastic sample generation:

Values for groundwater (n=31) and drinking water (n=21) isotopic compositions are derived by classical bootstrapping (i.e. randomly sampling n times, with replacement) from the original data points (Efron and Tibshirani, 1998). It has to be considered, that the δ¹⁸O and δ²H data are not independently distributed. In particular the δ¹⁸O and δ²H groundwater values are clearly correlated, which is basically due to local meteoric water line effects controlling their natural variability. We therefore implemented the bootstrap such that corresponding δ¹⁸O and δ²H values of the original groundwater samples remain paired. It should be noted that disregarding this pairing would lead to an underestimation of the total uncertainty of the final results.

The time series of wastewater isotopic composition and wastewater discharge can not be treated by classical bootstrap, as the original data consist of only one replicate per timestep.
We therefore use a parametric bootstrap approach to approximate the randomness of these variables. In each bootstrap run, both a $\delta^{18}O_{\text{Wastewater}}$ and a $\delta^{2}H_{\text{Wastewater}}$ time series is generated by independently scattering the original data points with normal distributed errors reflecting the corresponding measurement uncertainties.

We identified four relevant types of systematic errors potentially disturbing the discharge measurements. The simulation includes appropriate offset and / or relative error terms for the recorded water level ($h$) and flow velocity ($v$) time series as well as the measured sewer diameter ($d$). In each Monte Carlo replication exactly one realization of these error variables is drawn. Thus, the accuracy of the discharge measurements is affected such that the results of a whole time series replicate are always influenced in the same direction. We utilize normal distributions with adjusted standard deviations that reflect our practical experience of maximum error ranges in comparable measurement situations. In contrary to these systematic

Figure 6. Flowchart of the Monte Carlo simulation scheme applied for error propagation. Dashed blue borders highlight the outcomes for which probability distributions are graphically represented in Figures 4, 7 and 8.
errors, pure random scatter (affecting the individual minutely measurements) is inconsequential on the final results: In preliminary simulations it proved to be largely compensated by the comparatively high measurement frequency. Hence, to save computational time, corresponding error terms were not included into the final simulation scheme.

In the second step, a subset of groundwater samples is randomly drawn with replacement from the bootstrap sample to be considered as infiltration contributors. The size of this subset is preselectable and directly reflects an assumption about the total number of individual infiltration points. It thus enables different scenarios about the state of sewer perforation to be calculated. Subsequently, arithmetic means are calculated to obtain estimates for the mean isotopic composition of infiltrating groundwater (selected subset) and drinking water (unchanged bootstrap sample).

In the third step, time series of infiltration ratios and wastewater discharge are calculated from Equations 2 and 6. Infiltration ratios obtained from $\delta^{18}O$ and $\delta^2H$ data are merged by weighted averaging. Weighting factors for each timestep consider the reciprocal variance of infiltration ratios obtained from the individual $\delta^{18}O$ and $\delta^2H$ records. Subsequently, the diurnal infiltration hydrograph is derived by combining the infiltration ratio and wastewater discharge time series. In the final step, all resulting time series are integrated to 24-hour total values (10000 replicates).

### Table 1. Specifications for Random Sample Generation in the Monte Carlo Simulation

<table>
<thead>
<tr>
<th>Isotopic Composition</th>
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<tbody>
<tr>
<td>groundwater</td>
<td>nonparametric bootstrap on 31 original data points</td>
<td></td>
</tr>
<tr>
<td>drinking water</td>
<td>nonparametric bootstrap on 21 original data points</td>
<td></td>
</tr>
<tr>
<td>wastewater</td>
<td>parametric bootstrap using i normal distributions $\mu_i =$ original data points (hourly data) $\sigma(\delta^2H) =$ 0.8 ‰ ; $\sigma(\delta^{18}O) =$ 0.08 ‰</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Discharge Measurements</th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>water level (h)</td>
<td>offset error term: normal distribution $\mu = 0$ m ; $\sigma = 0.005/\sqrt{3}$ m</td>
<td></td>
</tr>
<tr>
<td></td>
<td>relative error term: normal distribution $\mu = 1$ ; $\sigma = 0.01/\sqrt{3}$</td>
<td></td>
</tr>
<tr>
<td>velocity (v)</td>
<td>relative error term: normal distribution $\mu = 1$ ; $\sigma = 0.05/\sqrt{3}$</td>
<td></td>
</tr>
<tr>
<td>pipe diameter (d)</td>
<td>absolute error term: normal distribution $\mu = 0$ m ; $\sigma = 0.02/\sqrt{3}$ m</td>
<td></td>
</tr>
</tbody>
</table>
The uncertainty of our estimates for $Q_{\text{Infiltration}, \text{24h-total}}$ and $X_{\text{Infiltration, \text{24h-total}}}$ is characterized by the spread of the corresponding bootstrap-derived probability distributions. Figure 7 demonstrates the dependency of the observed coefficient of variation (standard deviation of the 10000 bootstrap replicates normalized by the mean) on the hypothesis taken for the number of contributing infiltration points. As mentioned before, the degree of sewer perforation is a critical system property with respect to the accuracy of our quantifications. Unfortunately, information about the number of infiltration points can not be derived from the measured variables, but must be taken from external sources. Assuming that all infiltrating groundwater enters the sewer system through only one single defect (worst case scenario with respect to the precision of our estimates), the coefficient of variation of the 24h-totals amounts to 12.4% ($Q_{\text{Infiltration}}$) and 11.4% ($X_{\text{Infiltration}}$) respectively (1σ-level). However, it decreases rapidly as the hypothesized number of infiltration points is enlarged. In the case of 100 sewer defects, it amounts to just about 5.8% and 3.3% (for 1000 sewer defects: 5.6% and 3.1%).

With respect to our current application, the sensitivity of the combined measurement uncertainty to the assumption about the sewer perforation can be regarded as negligible for scenarios with more than 10 individual sewer defects. We believe that realistic orders of magnitude for the number of individual infiltration points in our catchment do exceed this threshold by far. Comparative values can be deduced from existing Closed Circuit Television (CCTV) records that are periodically performed by larger sewer operators. Example given, the database maintained for the sewer system of Zurich (Entsorgung & Recycling Zürich - ERZ; 655 km of combined sewers and 74 km of sanitary sewers; considered to be comparably well maintained) reveals an average of 2.6 (combined sewers) and 3.1 (sanitary sewers) active leaks per kilometer that were directly established visually. Furthermore, 9 (combined sewers) respectively 21 (sanitary sewers) potential leaks per kilometer were identified through indirect

![Figure 7](image_url) 

**Figure 7.** Uncertainty of 24h-integrated infiltration discharge ($Q_{\text{Infiltration}}$) and infiltration ratio ($X_{\text{Infiltration}}$) depending on the number of contributing infiltration points considered for mixing in the Monte Carlo simulation.
indications like visible calcifications. A considerable number of defects are assumed to be undiscovered.

Scaled down to the size of our investigation site, we consider a number of 50 active infiltration points to reflect a realistic value and to be a conservative assumption with regard to combined measurement uncertainty. Figure 8 displays the probability distributions for the 24h-totals of $Q_{\text{Infiltration}}$ and $X_{\text{Infiltration}}$ obtained by using this adjustment in the Monte Carlo simulation. The 2.5th/97.5th inter-percentile range of these distributions provide the bootstrapped 95%-confidence intervals of our final estimates, yielding 632 to 794 m$^3$/day for $Q_{\text{Infiltration, 24h-total}}$ and 0.36 to 0.41 for $X_{\text{Infiltration, 24h-total}}$. Normalized to the total wastewater discharge, this corresponds to relative 95%-confidence intervals of about ± 4.5% ($Q_{\text{Infiltration}}$) and ± 2.5% ($X_{\text{Infiltration}}$) respectively. We rate these narrow confidence ranges to reflect the comparably low combined measurement uncertainty of our approach, which thereby highly satisfies the requirements for practical applications in the assessment of sewer networks. Note that the infiltration ratio is estimated more precisely than the infiltration discharge, since flow measurement errors largely cancel out in the computation of $X_{\text{Infiltration}}$. Depending on the actual problem, $X_{\text{Infiltration}}$ might therefore offer a more robust benchmark value than $Q_{\text{Infiltration}}$.

**Figure 8.** 24h-integrated infiltration results from 10000 Monte Carlo runs. The simulation considers the existence of 50 infiltration points which are randomly distributed over the sewer network.

**Interpretation of diurnal infiltration variations**

Sewer infiltration can show substantial temporal variations induced by seasonal changes in groundwater levels or as a consequence of rain events (i.e. occurrence of exponentially receding interflow). However, from our basic understanding the actual flux of groundwater infiltrating into the sewer pipes under dry weather conditions should not be subject to intraday fluctuations. Thus it is remarkable that from the hydrograph separation given in Figure 4 the absolute infiltration discharge at our measurement point indeed reveals certain diurnal variations: The results suggest that $Q_{\text{Infiltration}}$ is higher during the day than at night. We can exclude this observation to be an artifact due to non-linear errors in the discharge metering,
since this was counterchecked by parallel measurements with other devices installed in the same sewer line.

Presuming that the actual infiltration flux at the groundwater to sewer boundary remains constant during a diurnal cycle, such fluctuations may indicate irregular dispersion effects probably caused by larger backwater zones. This hypothesis is supported by the observation that the measured wastewater $\delta^{18}O$ and $\delta^2H$ time series do not agree with those predicted from the direct mixing of the groundwater and foul sewage endmember (Figure 9a, c and e): Mixing proportions for charts (c) and (e) are computed according to the average infiltration discharge obtained from the tracer analysis and the measured wastewater discharge (chart a).

To reproduce realistic characteristics on these “direct mixing” time series, a Fischer's type routing procedure was applied to simulate shear augmented Taylor dispersion (Fischer, 1968; Fischer et al., 1979; Rutherford, 1994). We used a dispersion coefficient of 0.2 m$^2$/s (Rieckermann et al., 2005) and a sewer length of 1500 m (mean flow path length for our investigation site). Note that both measured series are clearly attenuated in amplitude and as well time shifted compared to these direct mixing model curves.

Purely hydrodynamic phenomena of non stationary flow (the difference of celerity and flow velocity) explain only a small fraction of these observations. To gain an approximate estimation of the effects caused by larger backwater volumes, we routed modeled time series of wastewater isotopic compositions through a serially connected combination of a completely mixed linear reservoir (variable volume) and a completely mixed fixed volume reservoir (Figure 9b, d and f). The input foul sewage hydrograph is described as a parameterized triple sinusoidal curve. The input infiltration flow is assumed to be constant (Figure 9b). Isotopic compositions for foul sewage and infiltration are fixed to the endmember values estimated in the experimental study. Input wastewater time series are generated by mixing both components and subsequently routed through the simulated backwater compartments. Model outputs are fitted to the measured time series of wastewater discharge and wastewater isotopic composition by weighted nonlinear regression. Figures 9d and 9f show that this lumped backwater zone model is principally capable of explaining the observed wastewater isotopic composition changes. It also explains how the constant input infiltration flow is transformed to an infiltration hydrograph that exhibits diurnal fluctuation properties which basically resemble those characteristics observed in our experiment. Table 2 summarizes the estimated parameter values and some further model features. Two principal conclusions can be drawn: (i) The estimated parameter value for the input infiltration discharge amounts to 708 m$^3$/day. This corresponds satisfactorily with the value of 711 m$^3$/day (confidence interval of 632 to 794 m$^3$/day) obtained by the direct hydrograph separation, discussed earlier. This is plausible as the simulated backwater effect influences the wastewater time series with a 24h-periodicity and thus does not affect the value of such quantities based on 24h-integrations. (ii) The estimated reservoir volumes amount to an
average of 152 m$^3$, which can be considered an approximate order of magnitude of total backwater zone volume being present in the real system.

Indeed we detected that one rainwater water retention tank (connected in main line to a major subunit of the network) was accidentally blocked and partly piled up to a volume of around 65 m$^3$ during November 2003. However, this tank only explains part of the identified backwater volume and we were not able to definitely ascribe further backwater quantities to other special technical structures. This suggests that the majority of effective backwater volume is most likely associated with the cumulative effect of numerous smaller pools and backwater zones distributed over the whole network. These can for example be caused by smaller structural deficiency and/or sediment deposits.

**Figure 9.** Diurnal variation of wastewater isotopic composition predicted from the simple direct mixing of groundwater and foul sewage (left side: a, c and e) compared to the modelled effect of backwater zones (right side: b, d and f). Full red lines: model outcomes; Open circles: measured values; Dashed lines: inputs to the backwater zone model.
**Table 2.** Parameter estimates obtained for the backwater zone model. Model outcomes were fitted to the measured wastewater isotopic composition and discharge.

<table>
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<th>Parameter</th>
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<td>δ¹⁸O</td>
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<td>(plus estimated initial values for δ¹⁸O and δ²H)</td>
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**Future prospects of the use of stable isotopes for sewer infiltration surveys**

The primary goal of our study was to evaluate the suitability of the stable isotopes composition of water as a direct natural tracer for the quantification of sewer infiltration. In our field study, both bootstrap estimated confidence intervals for the 24h-integrals of $Q_{\text{Infiltration}}$ and $X_{\text{Infiltration}}$ range clearly below ± 5% related to the total wastewater discharge. With this degree of accuracy, the method well overcomes the limits of the conventional discharge metering based approaches. The validity of our results is additionally confirmed by the outcomes of another tracer experiment conducted in parallel in the same catchment: In this independent study the fraction of extraneous water was estimated from high resolution pollutant concentration time series (Kracht and Gujer, 2005). We hence conclude that the demonstrated stable isotope approach holds promise for general application to urban catchments, where suitable isotopic separations between drinking water and potential infiltration sources exist. However, with regard to comparable future implementations we want to underline the following critical points: (i) Successful application of the tracer method will depend on the local boundary conditions. It is important to carefully verify appropriate homogeneity of drinking water and infiltration sources. Larger variations can obscure the natural tracer signal and remain crucial for the method. It is also inevitable to derive the isotopic composition of infiltration origins from direct investigations within the catchment. The simple use of regional background values is not appropriate, as local groundwater compositions may be altered by processes like drinking water losses or irrigation. (ii) A
generalized statement on the overall uncertainties is not yet possible. In practice it will be necessary to individually assess the confidence in the results obtained based on an accurate hydrological dataset, i.e. using the demonstrated Monte Carlo method. (iii) Though under dry weather conditions the process of groundwater infiltration is generally accepted as not subject to significant fluctuations within the course of a diurnal cycle, it was demonstrated that the amount of infiltration discharge observed at a certain point in the sewer network can reveal intraday variations due to hydrodynamic effects. This implies the need for appropriately time resolved sampling over at least 24 hours. Alternatively, flow proportional composite samples might be considered. (iv) The tracer method introduced in this paper is basically limited to sewer catchments where only two components (one drinking water source, one groundwater source) are interacting. The combination of $\delta^{18}O$ and $\delta^2H$ may theoretically enable application also in constellations where two isotopically different sources contribute to sewer infiltration. However, in relevant situations the discrimination between two different infiltration origins is likely to be impossible because of the high correlation between $\delta^{18}O$ and $\delta^2H$ observed in most natural waters (meteoric waterline). Second, the methodology would attain reduced precision and suffer the loss of redundant control with respect to potentially interfering evaporation effects. (v) It may stand to reason that such three-component mixing problems could be addressed by a multi-dimensional tracer study, i.e. by additionally considering the concentration or isotopic composition of specific dissolved species. However, we want to emphasize that the identification of suitable natural tracers for applications in sewer systems is not a trivial task, since a vast amount of dissolved species are added to the foul sewage by households and industry in varying amounts. It is a delicate work to predict these background fluctuations in sewage chemistry with an accuracy that suits for quantitative investigations. (vi) The pure nugget variances observed in our groundwater isotopic data indicate no spatial dependency only at the sampling scale we have used here. Consideration of properties at higher spatial resolution may have allowed links to be made between the isotopic composition of groundwater and the positioning of sewer defects. The same applies for investigations at larger scales (city wide experiments), in situations of significantly heterogeneous distribution of relevant sewer defects (as a consequence of structural conditions of the network or positioning of groundwater tables) and where groundwater isotopic composition reveals larger systematic variations. In such cases it might be necessary to individually adapt the statistical analysis by appropriately considering the spatial interrelation of groundwater and sewer perforation data.
Acknowledgements

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References


CHAPTER 3

Experimental Design Evaluation for the Stable Isotope Method

Manuscript in preparation
Experimental Design Evaluation for the Stable Isotope Method

1 Introduction

The accuracy of infiltration estimates obtained with the Stable Isotope Method mainly depends on two factors: (i) the existing natural and technical boundary conditions (isotopic separation and natural intrinsic variability of the mixing endmembers, wastewater discharge properties, sewer network perforation characteristics) and (ii) the layout and quality of the experimental investigations. A comprehensive data analysis procedure including Monte Carlo simulation based routines for uncertainty analysis was derived in chapter 2. A simplified classification scheme for the pre-exploration of local boundary conditions will be presented in chapter 4.

Provided that the general hydrological situation of a catchment has been identified to suit the principal requirement for an application of the Stable Isotope Method, best possible quality of the final results requires an optimal experimental layout to be used. However, it is possible that in some cases a less precise estimate would be sufficient, accompanied by savings in cost and effort. The experimenter’s scope to adjust the experimental layout principally applies to the definition of alternative sampling schemes for the survey of wastewater and its mixing components (i.e. local drinking water and groundwater being utilized as appropriate proxies). The number of samples taken determines both the required field work time and laboratory expenses. In general, it would be beneficial to have a formal framework for evaluating the effect of different sampling strategies to be applied.

We use a Monte Carlo simulation approach to investigate the influence of different sampling strategies on

1) the accuracy of the infiltration estimates

2) the reliability of the corresponding confidence interval estimates

that an experimenter would have computed based on the (limited) information available from his respective field campaign.

In the following, we will first give a brief description of the employed simulation scheme and then exemplarily explore the approach based on surrogate input data series that are generated by bootstrapping from the Rümlang study dataset (the dataset was discussed in detail in chapter 2). Note that the proper evaluation of different experimental design alternatives requires extended information about the hydrological properties of a catchment, which however is not available to us for the general case. The quantitative results drawn from the presented numerical example are thus considered to be basically transferable to analogue
catchment settings. However, adapted input data would be required for the evaluation of distinctly differing hydrological situations (spatial distribution of infiltration sources, shape of the wastewater hydrograph, diurnal water use pattern).

2 Numerical Simulations

In this chapter, the term experimental design basically refers to the number of samples taken for replicate analysis (groundwater and drinking water) and the schedule of sampling time intervals (wastewater time series). Simulated data sets are used to investigate the effects of different sampling schemes on the accuracy of the estimated infiltration quantities. A detailed flowchart of the employed Monte Carlo simulation scheme is shown in Figure 1.

Each Monte Carlo replicate consists of the following steps (numeration 1 to 4 as specified in Figure 1).

1) A set of surrogate input data is generated by bootstrapping from the original Rümlang study dataset (as it was presented in chapter 2). Ordinary bootstrapping is used for the groundwater and drinking water isotope values. For the time series based data (wastewater isotope values, wastewater discharge) a parametric bootstrap is applied by adding scatter to the original datapoints. The magnitude of scatter considers typical measurement uncertainties for these variables. Scatter for each isotopic value is generated independently, whereas scatter for the discharge measurements is applied in the sense of “systematic errors” that affect the results of a whole time series replicate in the same direction (analogous to the procedures detailed in chapter 2).

In the following, this surrogate dataset serves as a reference value that represents the “perfect” data points of a hypothetically “true” hydrological situation. Based on a simulated infiltration scenario (random sampling of 50 infiltration points), the surrogate data are subsequently analyzed for the resulting “true” values of $Q_{\text{infiltration, 24h-total}}$ and $X_{\text{infiltration, 24h-total}}$.

Remark: With regard to this first simulation step it might seem conceivable, to directly use the Rümlang study dataset (i.e. the measurement results obtained in our field campaign) as one fixed reference for all simulation runs, therewith omitting the bootstrapping herein. However, it has to be taken into account that our experimental data are indeed only one possible observation of reality. In other words, by effect of sampling subjectivity and measurement errors a multitude of different hydrological variants may have led to the same measured values. Thus, we consider the simple repetition of this data to be inappropriate (i.e. to rigid). In contrast, the population of surrogate datasets generated by multiple replication of simulation step 1) approxi-
mately represents the full set of all hypothetical “Rümlang realities” that are compatible with our measurement results.

2) This step simulates a **scenario about the field and laboratory investigations** being performed by a “virtual experimenter” based on differently specified types of sampling schemes (presets for the amount of groundwater, drinking water and wastewater samples to take):

   The **subjective selection of sampling points** is simulated by ordinary bootstrapping from the reference data derived in step 1. Options for wastewater sampling are limited to choose a series of 7, 13 or 25 samples taken in equally spaced time intervals over a period of 24 hours. The start time of this series is randomly selected in each simulation run. Thus, the following options exist:

   - **25 items wastewater series:**
     - samples taken at: 00, 01, 02, ... 23 and 24 o’clock
   - **13 items wastewater series:**
     - samples taken at: 00, 02, 04, ... 22 and 24 o’clock
     - 01, 03, 05, ... 23 and 01 o’clock
   - **7 items wastewater series:**
     - samples taken at: 00, 04, 08, ... 20 and 24 o’clock
     - 01, 05, 09, ... 21 and 01 o’clock
     - 02, 06, 10, ... 22 and 02 o’clock
     - 03, 07, 11, ... 22 and 03 o’clock

   Subsequently, the influence of **measurement errors** is simulated by scattering all values by Gaussian random numbers considering typical measurement uncertainties of the respective variables (selected uncertainty ranges correspond to the specifications in chapter 2). Again, scatter for each isotopic value is generated independently, whilst scatter for discharge measurements is generated as a “systematic error” type (q.v. remarks given for step 1).

3) It is now presumed, that the “virtual experimenter” would use the standard evaluation scheme presented in chapter 2 to analyze his data and (i) **compute the 24h based infiltration estimates** \( Q_{\text{Infiltration, 24h-total}} \) and \( X_{\text{Infiltration, 24h-total}} \) and (ii) **specify appropriate confidence intervals**.

4) Finally, the estimates obtained by the “virtual experimenter” are **compared** to the reference “true” value of the respective simulation run.
Remark: To preserve feasible computation times, it was necessary to reduce the number of internal iterations used by the “virtual experimenter” in simulation step 3) to $n.\text{MCS}_{\text{intern}} = 2'000$ (note that $n.\text{MCS}_{\text{intern}} = 10’000$ was used for the analogous evaluations in chapter 2). The results obtained with $n.\text{MCS}_{\text{intern}} = 2'000$ and $n.\text{MCS}_{\text{extern}} = 2'000$ are yet somewhat “rough”, but are considered to be sufficiently robust to fulfil the objectives of this evaluation. Increasing the number of Monte Carlo iterations (aiming to obtain smoother results) would have caused inexecutable long computation times.
Figure 1. Simulation scheme for the evaluation of different experimental layouts (sampling strategies) used for the Stable Isotope Method. Note that surrogate input data series are generated by bootstrapping from the Rümlang study dataset.
3 Results and Discussion

The simulation procedure was replicated $n_{\text{MCS}_{\text{extern}}} = 2'000$ times to derive:

1) Probability distributions for the deviation between the infiltration quantities stated by the “virtual experimenters” and the respective reference values. This is an alternative simulation based estimate of the combined measurement uncertainty of the infiltration results obtained with the Stable Isotope Method (first order uncertainty).

2) A comparison between the effective 95% confidence ranges derived in 1) and the predictions about 95% confidence intervals stated by the simulated “virtual experimenters”. This reflects a simulation based meta-evaluation of the quality of stated uncertainty ranges (second order uncertainty). In terms of a quantitative measure, we further counted the hit rates of the predicted 95% confidence intervals with respect to the respective “true” reference values (proportion of simulation runs, in which the predicted confidence interval actually contained the associated reference value).

Figure 2 illustrates the effect of the number of groundwater samples ($m_{\text{groundwater}}$) on the confidence ranges of the infiltration estimates (number of drinking water and wastewater samples being kept fixed). Naturally, uncertainty is reduced the more groundwater samples are considered for evaluation. However, up from a certain value an increase of sample size does not further contribute to a significant reduction of the observed uncertainty ranges (approximately $m_{\text{groundwater}} > 15$ for $Q_{\text{infiltration}}$ and $m_{\text{groundwater}} > 20$ for $X_{\text{infiltration}}$). This saturation is due to the overcompensation caused by the other sources of uncertainty (i.e. isotopic composition estimates for drinking water and wastewater and discharge metering).

Furthermore, Figure 2 compares these effectively “true” 95% confidence intervals to the predictions of the 95% confidence intervals, which the simulated “virtual experimenters” would have stated based on their respective on-hand information. The second order uncertainty is acceptably small for $m_{\text{groundwater}} > 15$. However, for $m_{\text{groundwater}} < 5$ the stability of the first order uncertainty evaluations becomes severely compromised. This means that an experimenter is endangered to considerably underestimate the uncertainty of his results. For $m_{\text{groundwater}} = 1$ the effect gets extreme by reason that the consideration of intrinsic groundwater variability is made impossible.

The lower extremes of Figure 2 might represent less realistic situations, as it is rather unlikely that an experimenter would perform an in depth investigation of groundwater and wastewater, but consider only two or three groundwater samples. Table 1 summarizes the results obtained for a series of different scenario types of practical relevance. Scenario 0 corresponds to the initial position of the full investigation presented in chapter 2. Scenarios 1 to 7 exemplify the sensitivity of the effective 95% confidence ranges with respect to a decrease in sample size.
used for the groundwater, drinking water or wastewater observation. Scenarios 8 to 11 represent typical combinations an experimenter might consider to choose. Table 2 summarizes the counted hit rates of internally predicted 95% confidence intervals with respect to the “true” reference values. In case of low second order uncertainty, the hit rates should equal 95%. Hit rates lower than 95% indicate that the simulated “virtual experimenters” were tending to underestimate the uncertainty of their results.

The scenario results indicate that in particular increasing the wastewater sampling interval causes a rapid increase of the combined measurement uncertainty. This deterioration of performance may be easily avoided, as increasing the frequency of wastewater sampling takes only minor efforts. Scenario 8 can be considered to represent an efficient combination of sample size and convenient measurement quality.

It was presumed that the quantitative results drawn from the presented numerical example are considered to be basically transferable to analogue catchment settings, but adapted input data are required for the evaluation of differing hydrological settings. However, it is yet difficult to predict if a certain catchment can be classified “analogue” to our example in this sense. It must be noted that experiences about the application of the stable isotope approach in different types of hydrological catchments is yet very sparse. Feedback data from further implementations of the method in different situations will be required to derive more profound empirical knowledge, on which robust recommendations about experimental layouts can be based on. We want to stress that the use of oversimplified sampling schemes may particularly provoke an increased risk of gross errors (i.e. application of the method in un-qualified hydrologic settings that are not recognized as such by the experimenter) which was in respect thereof not covered by the simulation approach demonstrated in this chapter.
Figure 2. Effective 95% confidence intervals of 24h integrated infiltration estimates obtained from simulated sampling campaign scenarios with different numbers of groundwater samples (full lines: differences between the 2000 infiltration estimates and the 2000 reference values of the respective simulation run). For comparison, dashed lines refer to the distribution of 2000 predictions about 95% confidence intervals that would have been stated based on the on-hand information available to the simulated “virtual experimenters” (thick dashed lines: mean; thin dashed lines: 2.5th/97.5th inter-percentile range).

Note that all values are expressed as percentage of the total wastewater discharge ($Q_{\text{wastewater}}$ reference value respectively internal $Q_{\text{wastewater estimate}}$). Sample size for the observation of drinking water and wastewater was kept fixed to $m_{\text{drinking water}} = 21$ and $m_{\text{wastewater}} = 25$ respectively. Supporting points for different scenarios about the layout of the groundwater sampling campaign were calculated at $m_{\text{groundwater}} = 1, 2, 3, 4, 5, 7, 9, 11, 13, 15, 20, 25$ and 30.
Table 1. Effective 95% confidence intervals observed in different scenario types (expressed as percentage of the total wastewater discharge).

<table>
<thead>
<tr>
<th>Scenario characteristics (number of samples)</th>
<th>95% confidence interval of 24h totals</th>
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</thead>
<tbody>
<tr>
<td>Groundwater</td>
<td>Drinking Water</td>
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<tr>
<td>0)</td>
<td>31</td>
</tr>
<tr>
<td>1)</td>
<td>21</td>
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<td>9)</td>
<td>14</td>
</tr>
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<td>10)</td>
<td>14</td>
</tr>
<tr>
<td>11)</td>
<td>7</td>
</tr>
</tbody>
</table>

Table 2. Hit rate of reference values with respect to the predicted 95% confidence intervals in 2000 MCS runs for different scenario types.

<table>
<thead>
<tr>
<th>Scenario characteristics (number of samples)</th>
<th>95% confidence interval of 24h totals</th>
</tr>
</thead>
<tbody>
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<td>Drinking Water</td>
</tr>
<tr>
<td>0)</td>
<td>95.2%</td>
</tr>
<tr>
<td>1)</td>
<td>94.2%</td>
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<tr>
<td>10)</td>
<td>93.0%</td>
</tr>
<tr>
<td>11)</td>
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</tbody>
</table>
CHAPTER 4

Innovative Tracer Methods for Infiltration Monitoring

Oliver Kracht, Markus Gresch and Willi Gujer

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Innovative Tracer Methods for Infiltration Monitoring

Oliver Kracht, Markus Gresch and Willi Gujer

Abstract
Two innovative tracer methods for the quantification of parasitic discharges in sewers have been proposed as routine applications within the scope of the European research project APUSS (Assessing Infiltration and Exfiltration on the Performance of Urban Sewer Systems). In order to improve the accuracy of wastewater hydrograph separation compared to conventional practice, the novel approaches are using intrinsic physicochemical characteristics (stable isotopes, pollutographs) of the wastewater as a natural tracer for the differentiation of its constituting components. The methods were evaluated and tested under field conditions and have been validated in the course of a comparative experimental study. We analyse the state of development achieved and demonstrate the application of the protocols with data from a reference experiment conducted in a 5'400 inhabitants combined sewer catchment. The article details the requirements for suitable boundary conditions and presents a comprehensive assessment of measurement uncertainties.

Keywords: Infiltration and Inflow; Sewer rehabilitation; Natural tracer; Stable isotopes; Pollutograph; APUSS

1 Introduction
Sewer systems constitute one of the most capital-intensive infrastructure systems in urbanized areas. Nevertheless, after decades of operation the sewer networks of many cities are in a state of disrepair, often as a result of inadequate preventative maintenance and insufficiently planned rehabilitation, or simply because of having reached the end of their useful design lives (Bishop et al., 1998; Davies et al., 2001; Ellis et al., 2003). The German Association for Water, Wastewater and Waste (DWA) claims that about 20% of the public sewer system in Germany requires short to mid-term rehabilitation, which would incur costs of around $70 billion (Berger and Lohaus, 2005). The US EPA estimates necessary expenditures of $390 billion to replace and upgrade existing wastewater infrastructure systems in the United States over the next two decades (costs include renewal of wastewater processing) (American Society of Civil Engineers, 2005). In the European Union, the full implementation of the Water Framework Directive 2000/60/CE will likewise require substantial investments to amend and renew the existing wastewater technology. In this context, it is hardly conceivable to increase the wastewater treatment efficiency without increasing the efficiency of wastewater collection, i.e. the quality of sewer networks.

The term “sewer infiltration” refers to extraneous groundwater that enters the sewer system through cracks and fissures in pipes and manhole walls, damaged house connections and
defective pipe joints and couplings. Non-intended inflows from brooks or land drains are a further source. Excessive amounts of infiltration are detrimental to the performance of wastewater treatment plants as the dilution of pollutant concentrations leads to lower pollutant removal efficiency. Moreover, the treatment plant’s available storm water capacity is reduced and emptying times of retention tanks are extended. As a result, non-treated wastewater is discharged into the environment more frequently. Infiltration causes an increase in energy consumption and operating costs of pumping stations and treatment plant infrastructure. Increased entry of sediments (eroded soil) may result in higher maintenance requirements and possible surface subsidence (Butler and Davies, 2004). Likewise, there is increasing evidence that in many European cities groundwater levels are particularly controlled by the serious drainage effect of permeable sewer systems.

Reported infiltration rates range widely between 0.01 to 1.0 m³/day/mm pipe diameter/km sewer length (Metcalf & Eddy INC., 1991). In the UK, infiltration ratios have been found to range from 15% to 50% of average dry weather flow (White et al., 1997). A nationwide census in Germany revealed an average infiltration ratio of 30% based on estimates provided by the treatment plant operators surveyed (Statistisches Bundesamt, 2003). About 10% of the German treatment plants may suffer infiltration ingresses exceeding 50% of average dry weather flow (ATV-DVWK Arbeitsgruppe Fremdwasser, 2003). Weiß et al. (2002) state that for a combined sewer system the typical relation between sewage, stormwater and infiltration inflow at the wastewater treatment plant may equal 0.30/0.35/0.35, thus more than 2/3 of the water passing through the plant was originally non- or little-polluted.

In view of the significant environmental and economic consequences, the accurate quantification of sewer infiltration is a crucial issue in the context of problem-oriented rehabilitation management. However, knowledge about the water-tightness of a sewer system is often poor and investments are frequently based on uncertain information and limited data sets. In conventional practice, the quantification of infiltrating waters is addressed by more or less complex statistical analyses that only take into account wastewater flow time series (Hager et al., 1985; Joint Task Force of the Water Environment Federation and American Society of Civil Engineers, 1994; Weiß et al., 2002). Thus the results obtained are subject to considerable uncertainties as the underlying assumptions are often too overly simplified or hardly verifiable. For example, practitioners often imply that the hydrographs diurnal night-time minimum equals the amount of extraneous discharge, which is oversimplified in view of today’s catchments sizes (long flow distances) and wastewater production patterns (24-hours usage of drinking water). The technical literature contains some brief information on commonly used correction factors to account for night residual domestic foul sewage flow (Abwassertechnische Vereinigung, 1982; Fischer, 1990; Triebel, 1973). However, these procedures generally involve a great degree of subjectivity and were supposedly not even intended for actual measurement purpose by their original authors (but rather aimed to provide
planning guidelines or schematic control of environmental regulations). To directly differentiate and quantify the origin of discharged waters, a tracer based approach would be an optimal alternative. However, to date little research has been devoted to this field (Hager et al., 1984; Kracht and Gujer, 2005; Verbanck, 1993).

2 Chemical Hydrograph Separation

The chemical hydrograph separation of wastewater is based on a tracer mass balance approach to differentiate discharge components from two or more sources. The simplest concept, applicable for dry weather situations, distinguishes between “real” foul sewage and infiltrating water. In such a binary mixing system, the fraction of wastewater discharge that is stemming from infiltration (infiltration ratio $X_{infiltration}$) and the corresponding amount of infiltration discharge ($Q_{infiltration}$) are given by:

$$X_{infiltration}(t) = \frac{C_{Foul\ Sewage}(t) - C_{Wastewater}(t)}{C_{Foul\ Sewage}(t) - C_{infiltration}(t)} \quad (Eq.1)$$

$$Q_{infiltration}(t) = X_{infiltration}(t) \cdot Q_{Wastewater}(t) \quad (Eq.2)$$

where $Q$ is the discharge and $C$ the tracer concentration. The accurate integration of $X_{infiltration}$ over time requires discharge measurements as a weighting function:

$$X_{infiltration,\ 24h-total} = \frac{\int_{24h} X_{infiltration}(t) \cdot Q_{Wastewater}(t) \cdot dt}{\int_{24h} Q_{Wastewater}(t) \cdot dt} \quad (Eq.3)$$

Whilst $C_{Wastewater}$ and $Q_{Wastewater}$ can easily be measured in the sewer, $C_{infiltration}$ and $C_{Foul\ Sewage}$ are hardly accessible at the catchment or even subcatchment scale: At the moment they enter the sewer system, foul sewage and infiltration are mixed and any separate analysis is hindered. This has a central consequence: The investigation of foul sewage and infiltrating water must be performed indirectly, for instance by examining the drinking water and the local groundwater, which constitute their intrinsic proveniences. Yet, the artificial labelling of drinking water is prohibited and the homogenous distribution of artificial tracers throughout an entire aquifer is neither feasible nor desirable for environmental reasons. In its place, specific inherent characteristics of the local drinking water, ground water and sewage have to be used as natural indicators of the mixing processes. Yet, it is generally uncommon to find such suitable natural tracers, since a vast number of dissolved species are added to the foul sewage. Most of these components exhibit large daily fluctuations and therewith obscure the natural tracer signals.
3 Methodologies
To overcome the limitations mentioned above, two methods for the quantification of parasitic discharges in sewers with natural tracers have been worked out:

Stable Isotope Method
This approach uses the stable isotope composition of mains water (proxy for the foul sewage) and local groundwater (proxy for the infiltrating water) as direct natural tracers. Compared to most other natural water characteristics, the natural ratio of stable oxygen and hydrogen isotopes in the water molecule is very robust with respect to changes in water chemistry or biological activities (Clark and Fritz, 1997; Fritz and Fontes, 1980; Hoefs, 2004). In particular, it is not supposed to be affected by utilization in the urban infrastructures. On a regional scale, the isotopic composition of water is predominantly controlled by the topographic elevation and continental setting of the region where groundwater and surface water are recharged by precipitation (Craig and Gordon, 1965; Gat et al., 2001; Rozanski et al., 1993). Suitable differences in the isotope ratios (“isotopic separation”), i.e. caused by the continental or the altitude effect, can exist where drinking water originates from a distant hydrological regime, whereas the parasitic water stems from groundwater that is recharged by local precipitation. However, beyond this it is critical to verify a low spatial variability of the isotopic composition within the ambient aquifer. The question of inhomogeneities of the local groundwater or other origins of parasitic waters can be crucial for the applicability of the method. It is therefore mandatory to investigate the hydrological and hydrogeological situation in the catchment thoroughly and on a broader basis. It is important to define a general concept about the principal hydraulic interactions and pathways of different waters in the catchment. Based on this, a sampling scheme is developed that will ideally cover all natural water occurrences that are potentially representative for the relevant sources of infiltration in the catchment. As a second constraint, variations in the water supply must be low or at least be precisely quantifiable: Drinking water networks are often highly intermeshed, as they are designed to provide security of uninterrupted supply and an equilibration of production and consumption in different parts of the system. This can result in an obscured tracer signal, when parts of a catchment receive drinking water from differing sources during the course of an experiment.

Pollutant Time Series Method
In contrast to the stable isotope approach, the Pollutant Time Series Method does not require a direct investigation of drinking water or infiltration proveniences: The method quantifies infiltrating non-polluted waters based on time series of wastewater flow and pollutant concentrations that are basically measured at a single point in the sewer system. In contrast to earlier pollutant-tracer approaches (Hager et al., 1984), it is based on high temporally resolved
measurements with in-line devices, which allows for considerable refinements of the data analysis. In particular, it does not require the assumption of stationary cyclic behaviour of the diurnal hydrograph and pollutograph.

Based on a transformation of Equation 1, the analysis uses a mixing model describing the pollutant concentration in dependency of wastewater discharge and time (Equation 4). The model considers temporal fluctuations of the pollutant concentration in the foul sewage (Equation 5) 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 (Equation 6):

\[
C_{\text{Wastewater, Model}} = \frac{(Q_{\text{Wastewater}} - Q_{\text{Infiltration}}) \cdot C_{\text{Foul Sewage}} + Q_{\text{Infiltration}} \cdot C_{\text{Infiltration}}}{Q_{\text{Wastewater}}} \quad \text{(Eq.4)}
\]

\[
C_{\text{Foul Sewage}} = f(t) + f(Q_{\text{Foul Sewage}}) \quad \text{(Eq.5)}
\]

\[
Q_{\text{Infiltration}} = Q_{\text{Baseflow}} + Q_{0,\text{interflow}} e^{-kt_{\text{rec}}}(1-e^{-t_{\text{rec}}}) \quad \text{(Eq.6)}
\]

where: \(Q_{0,\text{interflow}}\): initial magnitude of interflow at the time \(t_{0,\text{interflow}}\); \(k_{\text{rec}}\): recession constant

Equation 4 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) as its concentration in the parasitic water is assumed to be negligible. This avoids difficulties in characterizing the tracer mass contribution stemming from the infiltration itself. \(Q_{\text{Infiltration}}\) is then identifiable by fitting a modelled time series of pollutant concentrations to the measured data (Kracht and Gujer, 2005).

A characteristic feature of this method is the use of automatically operating in-line devices. Recently available submersible spectrometer probes allow for a direct measurement of COD-equivalents in the media by means of light absorbance in the UV-VIS range (Langergraber et al., 2003). This considerably simplifies the recording of longer time series in a high temporal resolution, which significantly improves the documentation of the wastewater composition dynamics.

4 Experimental Studies

The application of our two approaches is subsequently exemplified with data from measurement campaigns conducted in the village of Rümlang (CH) in 2003. Rümlang is a community of about 5'400 inhabitants, located at the north-eastern border of the agglomeration of Zurich. The total length of its sewer system amounts to 23.1 km. The village has a mixed in-
frrastructure with no predominant type of industry. All wastewater sampling and inline measurements were conducted in a trunk sewer that connects the village to the regional treatment plant. The catchment was chosen for the following reasons:

- The topographic setting makes it best suitable for the stable isotope approach.
- The number of households is large enough to provide a continuous wastewater discharge (averaging effect over a minimum number of single contributors).
- The spatial extent of the sewer network is comparably small and an approximate quantification by the classical “night time minimum” approach is justified.
- An almost complete congruence of the areas covered by the water supply and the sewer network allows for a plausibility check by conducting a water balance.

The measurements were carried out during a dry weather period. Some last rainfalls occurred three days before the campaign.

Isotope values are cited in the δ-notation in reference to the international standard V-SMOW (International Atomic Energy Agency, 1995). All data evaluation was performed with the R software environment for statistical computing (R Development Core Team, 2005) using in-house scripts written by the authors.

**Field application of the Stable Isotope Method**

The shallow underground of Rümlang consists of quaternary gravel and sand deposits that form a local aquifer, which is partly covered and intersected by poorly permeable moraine sediments. The terrain is generally inclined north east towards the valley of the river Glatt, which is the natural receiving water for the area. Due to this topographic situation, it can be excluded that any hydraulic interaction (e.g. infiltration of river water to the local groundwater) influences the relevant areas of the sewer network. Thus, all relevant groundwater exclusively originates from local precipitation formed at a moderate altitude. In contrast, Rümlang depends for a major part of its water supply on lake water obtained from the Lake Zurich, which to a large extent stems from precipitation in the Alps (average altitude of the drainage basin ≈ 1220 m asl). Zurich lake water is about 1.8‰ (δ¹⁸O) and 11.7‰ (δ²H) lighter compared to the local groundwater (local terrain ≈ 450 m asl). This isotopic separation is basically sufficient to be used as an infiltration tracer. However, in previous measurement campaigns substantial uncertainties were caused by some varying amounts of local groundwater productions delivered to the water supply mains. For the experimental campaign demonstrated here, the entire water supply system of the investigated catchment was therefore changed such that only water from Lake Zurich was delivered. This required close cooperation with the local stakeholders as well as detailed investigations on the adjacent water
supply networks. Homogeneity of the drinking water was continuously controlled by automatic conductivity measurements that were installed in the water mains.

A sampling scheme was assembled to investigate the entire range of occurring shallow groundwater that, from hydrological a priori knowledge, represents possible sources for infiltration into the Rümlang sewer system. Finally 31 samples taken from monitoring wells (various screened sections from 2 to 10 meters below ground), two production wells (screened sections down to 17 m below ground), springs, surface water sites (local creeks) and land drain systems were considered (principal experiment 11/2003). Furthermore, the temporal stability of groundwater isotopic composition was positively confirmed by screening a series of monthly samples (07/2003 to 04/2004) taken from selected observation points. Drinking water was sampled from seven continuously purged observation points in the mains network (21 samples in total).

Figure 1 compares the $\delta^2$H and $\delta^{18}$O endmember ranges observed for groundwater and drinking water to a series of 28 hourly wastewater samples (charts 1.1 and 1.2). As expected, the wastewater is isotopically enriched during night time, indicating a higher fraction of isotopically heavier infiltrating water being present in the sewer. During day time the wastewater shows depleted isotope values due to the larger amount of isotopically lighter foul sewage being discharged. Based on these data, the infiltration ratio and the corresponding infiltration discharge are calculated (charts 1.3 and 1.4).

Estimates for the average isotopic composition of infiltrating water ($\delta^{18}$O\text{Infiltration} \approx \delta^{18}$O\text{Groundwater}$) and foul sewage ($\delta^{18}$O\text{Foul Sewage} \approx \delta^{18}$O\text{Drinking Water}$) were derived from the arithmetic means of the corresponding sample series ($\delta^2$H results were processed accordingly). Standard uncertainties can be predicted from Equation 7, where empirical distributions are corrected for imprecision of the analytical methods applied. Herewith we differentiate the magnitude of variance in the empirically observed distributions that stems from natural variability from the magnitude of variance being introduced by the laboratory measurements:

$$\sigma(\delta^{18}O_{\text{Mean}}) \approx \sqrt{\frac{1}{n-1} \sum_{\text{Sample}=1}^{n} (\delta^{18}O_{\text{Sample}} - \overline{\delta^{18}O_{\text{Sample}}})^2} - \sigma_{\text{Laboratory}}^2 \left(\frac{1}{\sqrt{n}}\right)^2$$  \hspace{1cm} (Eq.7)

where:

$$\overline{\delta^{18}O_{\text{Mean}}} \approx \overline{\delta^{18}O_{\text{Sample}}} = \frac{1}{n} \sum_{\text{Sample}=1}^{n} \delta^{18}O_{\text{Sample}}$$  \hspace{1cm} (Eq.8)
Considering a laboratory standard uncertainty of $\sigma(\delta^{18}O)_{\text{Laboratory}} = 0.08\%$ (0.8\% for $\delta^2$H) we obtain $\delta^{18}O_{\text{Infiltration}} = -9.54 \pm 0.15 \%$ and $\delta^{18}O_{\text{Foul Sewage}} = \delta^{18}O_{\text{Drinking Water}} = -11.31 \pm 0.02 \%$ ($\delta^2$H $\text{Infiltration} = -68.6 \pm 1.2 \%$ and $\delta^2$H $\text{Foul Sewage} = -80.3 \pm 0.2 \%$).

The accuracy of the final infiltration results is determined by uncertainties in the estimates for the isotopic composition of the two mixing endmembers (foul sewage and infiltrating water), the isotope values of the wastewater samples and possible systematic errors embedded in the discharge measurements. Their effect on the final results was studied with Monte Carlo simulation (MCS). Since it is hard to describe the true nature of randomness in these measured variables, we took an approximate approach by assuming the following probability distributions:

- $\delta^2$H and $\delta^{18}$O endmember values of infiltration and foul sewage: normal distributions with mean values and standard deviation as given above.
- $\delta^2$H$\text{Wastewater}$ and $\delta^{18}$O$\text{Wastewater}$: independent normal distributions with mean values as measured for each single sample and standard deviations $\sigma(\delta^{18}O)_{\text{Laboratory}} = 0.08\%$ and $\sigma(\delta^2$H)$_{\text{Laboratory}} = 0.8\%$, respectively.
- Discharge was calculated from water level and flow velocity, which were recorded using a combined air-bubbler / ultrasonic Doppler probe. This gauging is presumed to be affected by the following uncertainty ranges:
  - water level: ± 1\% maximum relative error, ± 5 mm maximum offset error
  - velocity: ± 5\% maximum relative error, no offset
  - pipe diameter: ± 20 mm maximum absolute error (900 mm diameter pipe)

For the MCS we used normal distributions with adjusted standard deviations being $1/\sqrt{3}$ \%, $5/\sqrt{3}$ mm, $5/\sqrt{3}$ \% and $20/\sqrt{3}$ mm respectively.

A stochastic realisation of 10000 hydrograph separations was generated and integrated over time. To obtain truthful results, the following simulation scheme was applied: In each run exactly one independent sample for (i) the $\delta^{18}$O and $\delta^2$H endmember composition of infiltration and foul sewage, (ii) the defined discharge error terms and (iii) each of the hourly isotopic values of the wastewater sample time series is drawn. Random number generation for the infiltration endmember required the use of a correlated sampling technique (joint $\delta^{18}$O - $\delta^2$H probability distribution), to account for the strong $\delta^{18}$O - $\delta^2$H correlation observed in our groundwater data (local meteoric water line effect, $r = 0.88$). Note that uncorrelated sampling of these terms would cause an underestimation of uncertainties. Infiltration ratios initially obtained separately from the respective $\delta^{18}$O and $\delta^2$H records are merged by weighted averaging in an intermediate calculation step before final integration.
The simulation yields a 95%-confidence interval (2.5th to 97.5th inter-percentile range of the MCS results) of \(Q_{\text{Infiltration, 24h-total}} = [584 - 882] \text{ m}^3/\text{day}\). However, this prediction is based on the most unfavourable hydrological assumption that the \(\delta^{18}O\) and \(\delta^2H\) values obtained from the individual groundwater samples would not be mixed (averaged) in the aquifer: Equation 7 presumes that with equal probability the value of each single water sample possibly represents the average isotopic composition of the whole parasitic infiltration and thus overestimates the actual uncertainty. In the real system, the mixing of infiltration quantities contributed from spots with different isotopic composition is basically controlled by the overall perforation state of the sewer system (i.e. the corresponding number and spatial distribution of defects). In order to account for this effect, an additional simulation step was incorporated into an extended bootstrap based MCS scheme (Kracht et al., 2007). Herein, we hypothesised 50 equally sized groundwater inresses to be randomly distributed over the entire sewer system, which is considered to reflect a realistic value for our catchment. In this way, the uncertainty for the isotopic composition of the infiltration endmember value effectively reduces to \(\sigma(\delta^{18}O)_{\text{Infiltration}} = 0.014\%\) and \(\sigma(\delta^2H)_{\text{Infiltration}} = 0.12\%\). Likewise, the 95%-confidence interval predicted for \(Q_{\text{Infiltration, 24h-total}}\) narrows down to \([632 - 794]\) \text{ m}^3/\text{day}\. Note that the graphical representations of 95%-confidence intervals for the calculated infiltration results in Figure 1 (charts 1.3 and 1.4) are based on this assumption of 50 infiltration points, whereas quantiles in charts 1.1 and 1.2 refer to the empirically observed values of the 31 field samples.

**Field application of the Pollutant Time Series Method**

COD-equivalents were measured in 2-minute intervals with a submersible UV-VIS spectrometer (chart 2.1 in Figure 1). The instrument (Spectrolyser, Scan-Messtechnik) was calibrated with a series of laboratory samples taken in parallel to the in-situ measurements. A mixing model according to Equations 4 to 6 was implemented, in which Equation 5 was specified by combining a harmonic and a polynomial term:

\[
\begin{align*}
    f(t) &= A \cdot \sin(f \cdot 2 \cdot \pi \cdot (t - \text{phase})) \quad \text{(Eq.9)} \\
    f(Q_{\text{Foul Sewage}}) &= a + b \cdot Q_{\text{Foul Sewage}} \quad \text{(Eq.10)}
\end{align*}
\]

Values for \(Q_{\text{Baseflow}}, Q_{\text{0,Interflow}}, k_{\text{rec}}, a, b, A\) and \(\text{phase}\) were estimated by minimizing the sum of squared residuals between the measured values and the modelled time series \(\text{COD}_{\text{Wastewater, Model}}\) (\(freq\) was fixed to \(1 \text{ day}^{-1}\)). The resulting hydrograph decomposition is displayed in charts 2.3 and 2.4 of Figure 1. The accuracy of the baseflow to interflow separation was to some extent impaired by identifiability problems. However, the separation of infiltration and foul sewage was not negatively affected.
The influence of input parameter uncertainties was propagated through the data analysis algorithm by MCS assuming the following probability distributions:

- Discharge measurements: as described for the stable isotope approach

- Uncertainty ranges for the COD measurements can be directly extracted from the regression analysis performed for the in-situ calibration of the UV-VIS spectrometer. Accordingly, standard deviations for the offset error term and relative error term were estimated to be 34 mg/l and 7% respectively. Both error terms are strongly correlated ($r = -0.9$), which required correlated sampling to be used for the MCS. Note that ignoring the correlation of these terms would cause a considerable overestimation of overall uncertainties.

- The uncertainty stemming from possible low background COD concentrations in the infiltrating water was accounted for by a rectangular distribution reaching from 0 to 5 mg/l. However, this proved to have no significant effect on the results.

In each of 10000 Monte Carlo runs exactly one sample for each of the five systematic error influences was drawn and the model parameters and hydrograph separation were estimated.

**Figure 1.** Measurement results for the Rümlang trunk sewer (November 2003)

*Stable Isotope Method:* 1.1 and 1.2: Measured isotopic compositions (error bars: $2\sigma$ standard uncertainty of wastewater samples; dotted lines: 2.5th/97.5th percentiles of the empirical distributions of sampled ground- and drinking water). 1.3: Infiltration ratio. 1.4: Hydrograph separation (dotted lines: 95% confidence interval, not shown for $Q_{\text{wastewater}}$).

*Pollutant Time Series Method:* 2.1: Registered COD concentrations. 2.2: Modelled COD time series. 2.3: Hydrograph separation (full line: bulk infiltration to foul sewage separation, dashed line: baseflow to interflow separation). 2.4: Zoom into chart 2.3 (to be compared with 1.4; confidence intervals not shown for legibility reasons).
1.1

21/11/2003

δ²H [‰ vs VSMOW]

δ¹⁸O [‰ vs VSMOW]

1.2

21/11/2003

δ¹⁸O [‰ vs VSMOW]

1.3

21/11/2003

XInfiltration [%]

1.4

21/11/2003

Q [l/s]

Pollutant Time Series Method

2.1


COD [mg/l]

2.2


COD model [mg/l]

2.3


Q [l/s]

2.4


Q [l/s]

Wastewater

Groundwater

Drinking Water

Infiltration

Foul Sewage

Pollutant Time Series Method

Stable Isotopes Method

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5 Results and Discussion

Quantification of extraneous discharges

In practical applications, we are interested in assessing the structural integrity of a sewer network in terms of the infiltration discharge and infiltration ratio integrated over a meaningful time interval. Figure 2 compares integrations of $Q_{\text{Infiltration}}$ and $X_{\text{Infiltration}}$ over 24 hours that were obtained from MCS applied to both methods. Both methods do agree very well in their resulting infiltration estimates. This does support the validity of the methods as they are largely uncorrelated in their functional principle (thus the outcomes of the Pollutant Time Series Method are not considered to be conditional on those of the Stable Isotope Method and vice versa). The 2.5th to 97.5th inter-percentile ranges of the MCS outcomes provide approximate 95%-confidence intervals for the infiltration results (calculated means and confidence ranges are summarized in Table 1). The 95%-confidence interval for $Q_{\text{Infiltration, 24h-total}}$ equals about ± 4.5% of the total wastewater discharge for the Stable Isotope Method and ± 5.5% for the Pollutant Time Series Method. The infiltration ratio is estimated more precisely with both methods (95%-confidence intervals normalized to the total wastewater discharge = ± 2.5% and ± 4%, respectively). This was expected, since flow measurement errors largely cancel out in the computation of $X_{\text{Infiltration}}$. Especially with regard to application in comparative surveys, $X_{\text{Infiltration}}$ might thus offer a more solid benchmark value than $Q_{\text{Infiltration}}$. Compared to the Pollutant Time Series Method, the stable isotope approach seems somewhat more robust with respect to the error influences considered (slightly more narrow probability distributions). However, we regard these small differences to be insignificant and non-conclusive for a general statement of preference for one or the other. Taking the decision to use one of the two methods in future applications will rather depend on the specific boundary conditions of the respective investigation site.

On closer inspection, the structure of hydrograph separations obtained with the two methods clearly depends on the underlying assumptions (Figure 1). The Stable Isotope Method defines parasitic infiltration as the fraction of wastewater that is carrying a specific isotopic signature. In contrast, the Pollutant Time Series Method identifies infiltration as the discharge of clean water with predefined flow characteristics (constant baseflow, exponentially receding interflow). It is thus an intrinsic property of this approach that all dynamic patterns that are not included in the predefinitions will remain undetected. Note that the partition obtained with the stable isotope approach apparently reveals certain intraday fluctuations of $Q_{\text{Infiltration}}$ that are not covered by the model definitions of the time series approach. From our basic understanding of the process, the actual infiltration flux at the groundwater to sewer boundary remains constant during a diurnal cycle. Such fluctuations may therefore indicate irregular dispersion effects, probably caused by larger backwater zones. However, such phenomena do not seriously affect the 24h based quantifications (Figure 2). This is plausible as under dry
weather conditions potential backwater effects do basically influence the wastewater time series with a 24h-periodicity and thus do not affect the value of such quantities based on 24h-integrations.

Table 1 further lists infiltration estimates obtained with the classical “minimum night flow” assumption and by a water balance between drinking water delivery and wastewater discharge. All results agree in a plausible way. However, note that the stated confidence intervals for water balance and “minimum night flow” do not account for conceivable conceptual discrepancies or model structure uncertainties (only uncertainties in flow metering and uncertainties about potential drinking water losses were explicitly considered here). It has to be clarified that we do not generally consider water balances as a suitable approach for infiltration measurements. Yet, in the particular case it was supported by favourable conditions (i.e. congruence of water supply and sewer network, external water supply fully controlled by magnetic inductive measurements during the experiment). The water balance was

\begin{figure}[h]
\centering
\includegraphics[width=\textwidth]{image.png}
\caption{Monte Carlo simulation of infiltration estimates integrated over 24-hours (21.11.2003). Results for the \textit{Stable Isotope Method} consider a perforation state of 50 randomly distributed groundwater ingresses (full line). For comparison, a computation without groundwater mixing (i.e. hypothesizing one single sewer defect) is shown (dashed line). No hypothesis on the spatial distribution of sewer defects is required to perform the uncertainty analysis for the pollutant time series method.}
\end{figure}
calculated for the whole village, whereas the subcatchment investigated by the other methods covered only about three quarters of it for technical reasons. Therefore only a comparison of \( X_{\text{Infiltration}} \) is meaningful. As expected, the minimum night flow results suggest a higher infiltration ratio than the other methods. However, the differences are still rather marginal. This observation reflects the particularly narrow wastewater residence time distribution of the comparatively small Rümlang sewer catchment. It can be assumed that these differences get significantly more pronounced when the size of the investigated catchment is enlarged.

**Table 1.** Comparison of the infiltration results obtained with different approaches (24h-integrated values for 21/11/2003). Numbers in parenthesis state bootstrapped 95%-confidence intervals.

<table>
<thead>
<tr>
<th></th>
<th>stable isotopes</th>
<th>pollutant time series</th>
<th>minimum night flow</th>
<th>water balance</th>
</tr>
</thead>
<tbody>
<tr>
<td>( Q_{\text{Infiltration, 24h-total}} )</td>
<td>711 (632 - 794) m³</td>
<td>683 (578 - 793) m³</td>
<td>786 (696 - 879) m³</td>
<td>-</td>
</tr>
<tr>
<td>( X_{\text{Infiltration, 24h-average}} )</td>
<td>39 (36 - 41) %</td>
<td>37 (33 - 41) %</td>
<td>43 (41 - 44) %</td>
<td>41 (32 - 49) %</td>
</tr>
</tbody>
</table>

**Required boundary conditions for the Stable Isotope Method**

The two critical system properties for the use of the *Stable Isotope Method* are the existing isotopic separation between drinking water and infiltration and the natural variability of possible infiltration sources. The accuracy of infiltration estimates will further depend on the actual infiltration ratio and the uncertainty of the values obtained for the isotopic composition of foul sewage and wastewater. However, we consider these latter influences to be less critical since in most cases they will be basically determined by the analytical method applied and small scale sample inhomogeneities only (note that we take for granted that appropriate homogeneity of the drinking water supply is guaranteed by suitable technical arrangements). Such scatter introduced by the imprecision of the laboratory analysis can be reduced easily by taking an appropriate number of samples. Example given: in our case study both \( \sigma(\delta^{18}O_{\text{Wastewater}}) \) and \( \sigma(\delta^{18}O_{\text{Foul Sewage}}) \) were effectively decreased from 0.08‰ to values below 0.02‰. In contrast, the uncertainty linked to the estimate for the isotopic composition of the infiltration endmember is additionally controlled by the spatial distribution of groundwater ingresses (i.e. to what extend isotopic values obtained from individual groundwater samples will randomly mix in the network-wide infiltration process). This effect was shown to be inconsequential for most typical network situations, where a hypothesized number of more than 2 or 3 individual defects per km is considered to reflect a realistic value. However, the situation might be complicated when the method is applied to sewer networks with a particularly inhomogeneous spatial distribution of relevant sewer defects.

Figure 3 gives a simplified overview scheme of the required boundary conditions in terms of magnitude of the tracer signal (isotopic separation \( \Delta^{18}O = \delta^{18}O_{\text{Foul Sewage}} - \delta^{18}O_{\text{Infiltration}} \)) and accuracy of the isotopic infiltration endmember value \( \sigma(\delta^{18}O_{\text{Infiltration}}) \). Regions for different
95%-confidence interval ranges were mapped out by an iterative Monte Carlo procedure. All combinations that are plotting to the right of the respective division lines are suitable to estimate the infiltration ratio with the specified precision. For orientation, the approximate working basis for the Rümlang study is exemplified with open circles. Note how the predicted uncertainty depends on the hypothesized sewer perforation characteristics (number of groundwater ingresses).

This graphical representation is supposed to constitute a reasonable approximation for a wide range of typical situations encountered in practice. The purpose is to provide a first evaluation on the applicability of the Stable Isotope Method. A detailed prognosis of achievable measurement quality for a specific situation will certainly require an individual uncertainty calculation, i.e. using the Monte Carlo approach as described in Kracht et al. (2007).

![Graph showing 95% confidence intervals for infiltration ratio estimation.](image)

**Figure 3.** Required boundary conditions for the Stable Isotope Method. Isolines define regions for the estimation of $X_{\text{Infiltration}}$ with a desired 95%-confidence interval of either 2.5%, 5%, 7.5%, 10%, 15% or 20% of the total wastewater flow. Circles indicate the Rümlang study: 1) Without mixing of observed groundwater values (i.e. assuming one single sewer defect). 2) Bootstrap considering 50 independent groundwater ingresses.

**Calculation basis:**
- $X_{\text{Infiltration}} = 40\%$
- $\sigma(\delta^{18}O_{\text{Wastewater}}) = 0.02 \%$
- $\sigma(\delta^{18}O_{\text{Foul Sewage}}) = 0.02 \%$

**Circles marking the Rümlang example:**
- $\Delta^{18}O_{\text{Foul Sewage-Infiltration}} = 1.8 \%$
- $\sigma(\delta^{18}O_{\text{Infiltration}}) = 0.15 \%$ (1)
- $\sigma(\delta^{18}O_{\text{Infiltration}}) = 0.014 \%$ (2)

The diagram needs to be recalculated in case of differing $\sigma(\delta^{18}O_{\text{Wastewater}})$ resp. $\sigma(\delta^{18}O_{\text{Foul Sewage}})$. 
Note that the *Stable Isotope Method* presented here is basically limited to sewer catchments where only two components (one drinking water source, one groundwater source) are interacting. The combination of $\delta^{18}$O and $\delta^{2}$H may theoretically enable application where two isotopically different sources contribute to sewer infiltration, too. However, in relevant situations the discrimination between two different infiltration origins is likely to be impossible because of the high correlation between $\delta^{18}$O and $\delta^{2}$H observed in most natural waters (meteoric waterline effects). Second, the methodology would attain reduced precision and suffer the loss of redundant control with respect to potentially interfering evaporation effects (Kracht et al., 2007).

**Required boundary conditions for the Pollutant Time Series Method**

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 in particular simplifies the realisation of measurements in the rainy seasons, when infiltration rates typically increase due to elevated groundwater tables. The method may be used at the outlet of any sub-catchment where a continuous discharge of wastewater can be assured. However, a certain minimum amount of wastewater flow is required for the disturbance free operation of the measuring devices. Furthermore, predominant types of industrial effluents should be excluded as these may hinder a regular data analysis.

Some dynamic of the quotient $Q_{\text{Infiltration}} / Q_{\text{Wastewater}}$ is required for successful parameter estimation from the measured data. This condition is usually met by the common diurnal variations of wastewater discharge. However, problems may arise in the case of very large sewer networks and / or extremely high infiltration ratios. The identifiability of the set of model parameters applied further depends on the structure of the individual hydrograph and pollutograph. This has to be handled with care, especially when refinements of the basic mass balance model are introduced (i.e. Equations 9 and 10). Note that extending the basic mixing model (Equation 4) by incorporating time dependent intraday fluctuations of the foul sewage concentration (Equation 5 and 9) in particular requires non-stationary behaviour of the infiltration discharge (example given a pronounced receding interflow component caused by a preceding rain event). Parameters related to Equation 9 are generally not justifiable from purely stationary cyclic pollutant concentration and discharge time series, but will in this case interfere with the correct identification of the infiltration quantities.

From the broader practical experience gathered in the APUSS project, we can conclude that the successful implementation of the pollutants method 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 connected with the required bias and drift free operation of the submersible...
UV-VIS spectrometer (which has been chosen for all field campaigns within APUSS). As one finding from our respective experiences it must be accepted that the continuous monitoring of wastewater quality parameters with optical measurements in raw sewage is obviously not a trivial task. Measurement quality will strongly depend on the skills and experience of the technical field team and the required investments for a successful measurement campaign should not be underestimated.

**Tracer based approaches to sewer exfiltration problems**

Undesirable infiltration of groundwater into sewers can contribute substantial proportions of the total discharge and is particularly detrimental to the pollutant removal efficiency of wastewater treatment plants. Inversely, exfiltration of sewage may potentially occur when deteriorated pipes are situated above the local groundwater table. Exfiltration of raw sewage from leaking sewers is considered to be a serious threat to humans and the environment since it can directly impact drinking water resources. It is agreed that such losses are unlikely to exceed the order of few percent of the total wastewater discharge and would thus not significantly alter the bulk isotopic composition of local groundwater. The stable isotope approach introduced in this paper can consequently not serve for their detection. Meaningful information about sewage losses is as well not justifiable from the pollutant concentration time series. However, promising methods for the direct measurement of sewer leakage based on the continuous or discontinuous dosing of artificial tracers into suspect reaches have been proposed recently (Rieckermann et al., 2007; Rieckermann et al., 2005; Rutsch et al., 2006).

6 **Conclusions and Outlook for Future Applications**

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 explanatory quality 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 trough 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 is considered to be an optimal alternative.

In this context, two novel tracer methods for the quantification of parasitic discharges in sewer systems have been developed. Both approaches employ intrinsic physico-chemical characteristics of the wastewater as a natural tracer for the differentiation of its constituent components (i.e. “real” foul sewage or infiltrating water): The Stable Isotope Method uses the different isotopic signatures of mains water and local groundwater as a direct natural tracer. 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 (i.e. when a city uses drinking water from a watershed that is situated in a different hydrological regime than the urbanized area). The proper use of this method requires a detailed hydrologic and hydrogeological investigation to define the principal hydraulic interactions and pathways of different waters in the catchment. 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. In contrast to earlier tracer based approaches it is based on high temporally resolved measurements with in-line devices, which allows for certain refinements of the underlying model for parameter estimation. The approach requires a certain dynamic of the wastewater hydrograph and pollutograph, which is usually met by the diurnal variations of the wastewater discharge. Both methods were evaluated and tested under field conditions and have been validated in the course of a comparative experimental study. They are proposed for routine applications on the catchment or subcatchment scale.

Narrow confidence ranges obtained with both approaches in our experimental study demonstrate the comparably low combined measurement uncertainty that is achievable. Basically, the new methods do fully satisfy the requirements for practical applications in the assessment of sewer networks. However, successful application of the tracer methods depends on the local boundary conditions at the investigation site. Therefore, a generalized statement on the overall precision is not possible. In practice it will be required to assess the confidence in the obtained results on an individual case basis.

Acknowledgements

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References


Conclusions
Conclusions

Sustainable strategies for the planning, operation and maintenance of urban drainage networks require adequate knowledge about their functional behaviour and interaction with the surrounding aquifers. Against this background, two generic detection methods have been developed, which make it possible to substantially improve the knowledge about extraneous discharges and the hydraulic processes connected therewith. The methods were evaluated and tested under field conditions and are proposed for routine application on the catchment or subcatchment scale.

1 New Infiltration Measurement Methods

Both infiltration methods developed are based on natural tracer signals, but principally differ in the utilized tracer signal and the associated experimental setup and investigations.

The Stable Isotope Method uses the stable isotopes compositions of mains water and local groundwater as suitable proxies for the isotopic compositions of the foul sewage and the infiltrating water respectively. It therewith allows for a direct calculation of infiltration ratios. The method is suited to quantify infiltration, when useable differences in the oxygen or hydrogen isotope ratios of the drinking water and the infiltrating water exist. This is possible (but not necessarily granted) where drinking water originates from a distant hydrological regime, whilst infiltrating water stems from local precipitation.

In contrast to the stable isotopes approach, the Pollutant Time Series Method does not require investigations of the drinking water or the infiltration origins. The fraction of extraneous water is estimated by means of a combined analysis of measured time series of pollutant concentrations and wastewater discharge. Successful application requires a certain dynamic of the wastewater hydrograph and pollutograph, which is usually suited by the diurnal variations of the wastewater discharge. 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. Measurements (concentration and discharge) are carried out at a single point in the sewer system without recourse to any additional external information.

A significant effort has been devoted to the evaluation of uncertainties and errors both in field measurements and in the data processing phase. For both methods, experimental protocols, uncertainty analysis procedures and data processing codes have been established and are available. The methods have been evaluated and tested under field conditions and have been validated in the course of a comparative experimental study.
2 Future Prospects and Limitations

The two tracer methods exclude several conventional sources of error (i.e. uncertain assumptions on residual foul sewage discharge at night time, amount and pattern of drinking water consumption, and temporal behaviour of extraneous flows). They do in this way overcome critical limitations of current practice in infiltration measurements. Our studies demonstrated the suitability of both approaches in practical application. Furthermore, the cooperation in an international research project enabled us to test our methods in several European cities and under the variable local conditions of our research partners. We also received valuable feedback about the applicability as well as about the restrictions that have to be considered in individual cases of operation.

Successful implementation of the tracer methods depends on the local conditions. In practice it will thus be required to assess the confidence in the obtained results on an individual case basis. We presume that most European sewer systems suffer infiltration ingresses that approximately fall within in a range of 20% to 70% of the wastewater discharge. Based on our experience, we are confident that under suitable conditions a careful experimenter can estimate such infiltration ratios with an accuracy of better than ± 4% (95% confidence level related to the total amount of wastewater). Basically, the new methods do therewith highly satisfy the requirements for practical applications in the assessment of sewer networks.

With regard to future implementations, the following critical points are of importance:

Stable Isotope Method

The two critical system properties for the applicability of the method 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 this reason it is mandatory to conduct a broader hydrological and hydrogeological investigation to obtain sound information on the principal hydrologic interactions and pathways of different waters in a catchment. The principle sources of infiltration (i.e. groundwater) need to be definable, accessible for sampling and statistically describable with sufficient precision.

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 basically limited to catchments or sub-catchments 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 combination of δ^{18}O and δ^{2}H may theoretically enable application also in constellations where two isotopically different sources contribute to sewer infiltration. However, in relevant situations the discrimination between two different infiltration origins is likely to be impossible because of the high correlation
between $\delta^{18}$O and $\delta^2$H observed in most natural waters (meteoric waterline). Second, the methodology would attain reduced precision and suffer the loss of redundant control with respect to potentially interfering evaporation effects.

Beyond this, practical difficulties may result from regional cross-linking of individual drinking water systems. Disturbed tracer signals can be caused, when parts of a catchment area receive water from different drinking water supplies in the course of a day.

**Pollutant Time Series Method**

The method may be used at the outlet of any sub-catchment 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. For investigations in smaller catchments this can be critical during minimum night flow. Furthermore, predominant types of industrial effluents should be excluded, as these may hinder a regular data analysis.

From the broader experience gathered in the APUSS project we can conclude that the successful implementation of the method 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 practical limitations of the method seems to be bound to the required bias and drift free operation of the submersible UV-VIS spectrometer (which has been chosen by all collaborators). 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. The required investments for a successful measurement campaign should not be underestimated.

3 Open Questions and Recommendations for Further Research

In this thesis different studies have been presented which aimed to support a more solid quantification of sewer infiltration. However, the tracer methods have the potential to be further developed and expanded. Future research may be directed toward the following topics:

**Stable Isotope Method**

Further experimental work in different types of urban catchments can significantly broaden the knowledge about typical variations and spatial structures of groundwater isotopic composition to be expected. This would allow a more precise statement to be made about (i) the required comprehensiveness of groundwater investigations and (ii) robustness of infiltration estimates obtained in related studies. For this reason, the systematic collection of data obtained from practical implementations of the method would be desirable.
The stable isotopes composition of water is in principal very robust with respect to changes in water chemistry or biological activities. Nevertheless, a higher degree of evaporation can affect the isotopic values of the remaining water. Our wastewater data have so far not shown any alteration by evaporative enrichment (no discernible deviation from the groundwater / foul sewage mixing line in the $\delta^{18}$O / $\delta^{2}$H-plot). Evaporative enrichment was additionally investigated by a smaller set of experimental in-household tests, with no relevant effects being observed. However, the critical question of possible evaporative enrichments should be further investigated, especially when transferring the methodology to cities in differing climate zones.

In our study, pure nugget variance in the groundwater isotope data indicated the absence of spatial autocorrelation. However, in cases where groundwater isotopic composition reveals larger systematic variations it should be considered to adapt the statistical analysis by explicitly considering the spatial interrelation of groundwater and sewer perforation. This particularly applies for sewer systems with significant heterogeneous distribution of relevant groundwater ingresses, which can be a consequence of the specific structural conditions of a sewer network or the positioning of groundwater tables in relation to the sewer pipes.

**Pollutant Time Series Method**

The time series based approach has the advantage that it does not require recourse to any additional external information: all measurements are carried out at a single point in the sewer system. The tracer system that was most applied in our investigations was the UV-VIS absorption of the wastewater that was calibrated to the chemical oxygen demand (COD). To broaden the range of applications, it would be useful to further investigate other suitable tracers or measurement techniques:

In our investigations, simple bulk parameters like electrical conductivity or wastewater temperature were found to be not applicable mainly because of the large uncertainties bound to the assumptions about their values in the infiltrating water (electrical conductivity was found inapplicable also because of its high scatter in the wastewater time series). It is thus advisable to primarily focus future investigations to typical wastewater pollutants parameters (COD, Ammonia, Boron, etc.) as their concentration in the infiltrating water is negligible in most cases, avoiding laborious surveys to ascertain average concentrations in the groundwater.

With respect to the question of alternative measurement techniques, we found ion-selective electrodes to perform unsatisfactorily in sewer applications due to cross-sensitivity and fouling of the membranes. However, future developments in sensor technology should be attended to.
A principal drawback of the UV-VIS based COD measurements are the comparably time consuming calibration procedure and the ensuring of drift free operation. However, the effective work investments are considered to be reduced when a suitable integration of the automatic measurements provides synergy effects, i.e. where collected time series data as well contribute valuable information for process control or continuous inspection of industrial discharges (probes operated in the influent of larger treatment plants).

It should be investigated if operating expense can be reduced when the UV-VIS-probe is not calibrated to a real concentration parameter, but alternatively generic absorption properties of foul sewage and infiltrating water are used as an infiltration tracer. The direct evaluation of absorbance may avoid the laborious calibration to COD. However, drift free operation of the probe will still have to be controlled.

Further experimental work in different types of sewer catchments would broaden the knowledge about the temporal structures of wastewater pollutographs. This would allow a more generalized statement to be made about parameter identifiability and measurement uncertainty in diverse hydrological settings. As indicated for the stable isotopes method, the systematic collection of data obtained from practical implementations would thus be advantageous.
APPENDIX A

The APUSS Project

*Synthetic review of objectives and results*
Appendix A.

The APUSS Project

Assessing Infiltration and Exfiltration on the Performance of Urban Sewer Systems (APUSS)

European Commission 5th R&D Framework Programme

APUSS contributed to the implementation of the Key Action “Sustainable Management and Quality of Water” within the Energy, Environment and Sustainable Development Contract n° EVK1-CT-2000-00072.

This text is a synthetic excerpt from the executive summary and the final report of the project. The full text of all reports, software and other documents are publicly available on the APUSS website at “http://www.insa-lyon.fr/Laboratoires/URGC-HU/apuss/”.

1 Objectives

Urban sewer systems constitute a 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 or exfiltration. Infiltration of groundwater is detrimental to wastewater treatment plant efficiency while exfiltration of wastewater can lead to groundwater pollution. The European standard EN 752-2 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.

During the period 2001-2004, the APUSS project (Assessing infiltration and exfiltration on the Performance of Urban Sewer Systems), associating universities, small and medium enterprises (SME) 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. The APUSS project had the following main objectives:

- To develop new methods and techniques based on tracers (chemicals and natural isotopes) in order to assess and quantify infiltration and exfiltration in sewer systems, at
different spatial scales (from the sewer reach or elementary sub-catchment to the
whole catchment) and under different conditions (steady and dynamic groundwater
levels, seasonal effects).

- To develop volumetric methods for the measurement of infiltration and exfiltration in
  house connections.

- To establish models and accompanying tools for large scale applications and end-user
decision support.

- To develop and validate modelling approaches at various time scales, including
  statistical methods to (i) extrapolate experimental results to similar unmonitored sewer
  reaches and upscale the acquired properties to larger catchment areas, and (ii)
  optimally choose the measurement locations.

- To propose approaches to help end users to assess the performance of their sewer sys-
tems and to choose investment strategies according to a multi-criteria methodology.

2 Scientific Achievements

For infiltration measurements, two methods have been developed. The first method is based
on oxygen and/or hydrogen isotopic ratios and has been tested most extensively in Rümlang
(CH), Lyon (F) and Rome (I). An in depth investigation of heterogeneities and variability in
the local groundwater as well as in the drinking water network has been carried out in
Rümlang. Such variations can obscure the natural tracer signal and remain crucial for the
applicability of the method. The second method is based on continuous flow and pollutant
concentration time series. In particular COD time series measured by means of UV-visible
spectrophotometers have been tested. Data analysis procedures have been established to
evaluate infiltration rates from the field data series. The experimental protocols and data
processing programs have been applied and tested by the APUSS partners.

Concerning exfiltration measurements, two novel tracer methods have been developed, tested
and validated under field conditions: the QUEST pulse injection method and the QUEST-C
continuous dosing method. Analytical procedures and data processing programs have been
developed and tested by the APUSS partners. Laboratory tests have been carried out to
evaluate (i) the adsorption and interaction of tracers with wastewater solids, and (ii) the effect
of sewer solids and sediments on exfiltration.

For the tracer methods, computational tools were implemented in the platform R, which is
freely available under the terms of GNU General Public License. A significant effort has been
devoted to the evaluation and the reduction of uncertainties and errors in both the field
measurements and the data processing phase. The suggested evaluation procedures are documented in the corresponding protocols.

As house connections are a further key component in infiltration / exfiltration phenomena, volumetric methods have been tested and applied at different experimental sites to measure infiltration and exfiltration. As there are many thousands of house connections in a city, a scaling-up method for a catchment-wide extrapolation has been proposed.

Conceptual models to simulate infiltration and exfiltration at various time scales have been selected and implemented in the *AquaBase* software.

Since it is hardly ever possible to cover all pipes within a city with infiltration / exfiltration experiments, a statistical method based on the similarity approach has been developed in order to facilitate the identification of representative catchments and the extrapolation of experimental values to wider catchments or the whole city scale. *AquaBase* serves as a software platform to describe sewer systems, to store experimental data series on groundwater levels and infiltration / exfiltration rates, to calibrate the models and to display all results. A user manual and case study applications have been provided. Models and tools have been developed and tested in the framework of a large scale example application, with an end-user oriented perspective.

The cost structure of sewer systems has been analysed, and a cost benefit analysis has been carried out for rehabilitation and replacement of sewers. The range of costs of the new measurement methods has been estimated. Specific performance indicators for both infiltration and exfiltration have been established and applied to various case studies. A software tool has been adapted for the calculation of these performance indicators and their associated information.

A generic methodology including modelling of the sewer system and of the wastewater treatment plant has been established to compare different investment strategies (rehabilitation of the sewer system, adaptation of the treatment plant, stormwater retention tanks, or any combination of these solutions). The multi-criteria method *Electre III* is used to compare and rank the various investment strategies regarding their effect on infiltration / exfiltration and the corresponding impact with respect to different criteria (environmental, operational, financial, etc.).

3 Conclusions

Appropriate methods to measure infiltration and exfiltration in sewer systems have been developed, evaluated and applied in various experimental sites with different contexts. For all methods, experimental protocols, uncertainty analysis and data processing codes have been established and are publicly available on the project website.
Infiltration and exfiltration models have been selected and implemented in an integrated software package based on the AquaBase platform. Performance indicators, investment strategies and economic valuations have been defined, with examples of application. All results are also publicly available on the project website.

Infiltration and exfiltration are critical phenomena on a long-term basis for sustainable urban water management and exert important socio-economic relevance and policy implications. The accurate and robust measurement methods, models and accompanying software and tools developed in the project will help end-users to develop better investment and rehabilitation strategies by taking the performance of actual systems into account. They can also contribute to future new regulations and policies including the EU Water Framework Directive.
Estimating Sewer Leakage from Continuous Tracer Experiments

Jörg Rieckermann, Vojtěch Bareš, Oliver Kracht, Daniel Braun, Willi Gujer

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Estimating Sewer Leakage from Continuous Tracer Experiments

Jörg Rieckermann, Vojtěch Bareš, Oliver Kracht, Daniel Braun, Willi Gujer

Abstract

Direct measurements of sewer leakage with continuous dosing of tracers are often considered too imprecise for practical applications. However, no mathematical framework for data analysis is reported in literature. In this paper, we present an improved experimental design and data analysis procedure together with a comprehensive framework for uncertainty assessment. Test runs in a 700m-long watertight sewer showed no significant bias and a very high precision of the methodology. The error in the results was assessed to 2.6% of the labeled flow with a simplified model which could be reduced to 1.2% when a dynamic data analysis procedure was applied. The major error contribution was caused by transient transport phenomena, which suggests that careful choosing of the experimental time is more important than the choice of a very specific tracer substance. Although the method is not intended to replace traditional CCTV inspections, it can provide complementary information for rational rehabilitation planning.

KEYWORDS: Sewer exfiltration, Environmental monitoring, Urban water management, Uncertainty analysis, Monte Carlo simulation, Experimental design

1 Introduction

In the last decades, several attempts have been undertaken to assess the magnitude of sewer leakage. Indirect methods try to deduct information on exfiltration from groundwater monitoring or a catchment-wide water balance. Direct measurements perform pressure testing on cracks or dose tracer substances into the sewer to identify leakage. In a review on sewer exfiltration, (Vollertsen et al., 2002) criticize the indirect measurements to rely on a number of critical assumptions. Regarding the tracer methods, which have also been applied to measure streamflow losses (Zellweger, 1994), the researchers recommend to interpret the results, which often show considerable exfiltration, “with care”. This is mostly, because on the one hand sewer leakage is expected to be in the order of only a few percent of the wastewater and on the other hand it is considered difficult to perform tracer experiments with a higher precision. However, no mathematical framework has been yet reported for the analysis of tracer experiment data that use a continuous dosing strategy (Jensen and Madsen, 1996; Knudsen, 1996) and the criticism is not based on a formal assessment of uncertainty.

This is a deeply unsatisfactory situation, because in contrast to traditional methods for the assessment of sewer leakage, tracer measurements have a large potential for practical application: the experiments are controllable to a large degree, readily applied and comparably cheap. Their routine application would be highly favorable because knowledge
on the magnitude of sewer leakage could not only aid the prevention of environmental and health hazards (Clara et al., 2004; Wakida and Lerner, 2005), but also contribute to improved rehabilitation strategies of the costly sewer infrastructure.

In this paper, we will present a robust method to perform exfiltration measurements with continuous tracer experiments together with a mathematical framework for data analysis. This contains the following specific innovations:

1. the QUEST-C method (QUantification of Exfiltration from Sewers using artificial Tracers with Continuous dosing) reduces systematic errors by a defined experimental setup
2. a comprehensive procedure to rigorously assess the accuracy of this method using reliable methods of uncertainty analysis
3. a new approach for data analysis that accounts for dynamic flow, which substantially reduces the remaining uncertainty

A brief overview about the methodology is presented together with two different models for data analysis and the error analysis framework. Finally, we will demonstrate the usefulness of this approach on a case study and discuss important benefits and limitations of our approach.

2 Methods

Conceptual outline

The basic principle of exfiltration measurements with tracers is to dose a well-known amount of tracer to the sewer under investigation and apply a mass balance on the investigation reach (Rieckermann et al., 2005). Given non-reactive behavior of the substance and full mixing, the tracer loss is directly related to the leakage in the reach. Losses of the so-called indicator tracer (Figure 1) are generally identified relative to a reference tracer which is not affected by leakage. Exfiltration \( E \) is expressed as a ratio relative to the labeled flow:

\[
E = 1 - \frac{\text{mass}_{\text{REF,in}} \cdot \text{mass}_{\text{IND,out}}}{\text{mass}_{\text{IND,in}} \cdot \text{mass}_{\text{REF,out}}} = 1 - \frac{\int c_{\text{REF}} \cdot q(t)_{\text{REF}} dt}{\int c_{\text{IND}} \cdot q(t)_{\text{IND}} dt} \cdot \frac{\int Q(t) \cdot C_{\text{IND}}(t) dt}{\int Q(t) \cdot C_{\text{REF}}(t) dt}
\]  

(Eq.1)

where \( c_{\text{REF}} \) and \( c_{\text{IND}} \) are the tracer concentrations of the dosing solution, \( q_{\text{IND}} \) and \( q_{\text{REF}} \) are the dosing rates, and \( C_{\text{IND}} \) and \( C_{\text{REF}} \) are the tracer concentrations in the sample. \( Q \) is the discharge at the sampling cross section (Figure 1).
(Rieckermann et al., 2005) used slug injections of tracers (i.e., Dirac pulses), which has the advantage that only one tracer substance is needed and only one single measurement error has to be considered. However, a continuous dosing of two tracer substances has the advantages that (i) practical difficulties with inline measurements are avoided (i.e., clogging, fouling, etc.), (ii) a variety of specific tracer substances and analytical techniques can be used which can be specifically tailored to avoid problems caused by the wastewater composition (e.g., significant tracer background concentrations) and (iii) exfiltration could be determined from a single grab sample if the discharge in the channel was steady (Figure 1, right). Nevertheless, we believe that the traditional setup of continuous tracer experiments (Jensen and Madsen, 1996; Knudsen, 1996) suffers some weaknesses that can be improved substantially by the modifications suggested below.

**Experimental setup of the QUEST-C method**

Based on a systematic analysis of the experimental setup of a tracer experiment and the according analytical procedures, we defined a novel experimental setup for the QUEST-C method that has the following important improvements regarding previous studies:

1. using gravimetric measurements rather than volumetric measurements (Figure 2)
2. measuring the ratio of the two tracer concentrations in the samples relative to the concentration ratio of the tracers in a working standard produced from the two dosing solutions
3. systematic checks for gross errors by collecting redundant information (e.g., additional discharge measurements)
Tracer selection - Obviously, the computed exfiltration ratio $E$ (equation 1) is systematically wrong if the tracer concentrations are reduced or magnified in the sewer reach (e.g., adsorption or natural tracer background in the wastewater). It might be for those reasons that early studies (Jensen and Madsen, 1996; Knudsen, 1996) chose very specific tracer substances which are unlikely to be present in wastewater (e.g., radioactive isotopes, fluorescent dyes). We obtained satisfactory results using Lithium as indicator and Bromide as reference tracer, which will be discussed in detail further below. For references to the use of Lithium and Bromide as wastewater tracers see (Flury and Papritz, 1993; Headley et al., 2005; Netter and Behrens, 1992; Seguret and Racault, 1998; Seguret et al., 2000).

Dosing technique - Both containers with tracer dosing solution were placed on balances with data acquisition facilities. This was more precise than volumetric measurements and allowed for a continuous recording of the tracer input (Figure 2).

Working standard - To compute exfiltration, the ratio of tracers at the sampling point must be compared against the ratio of dosed tracer masses. To reliably estimate the dosed tracer masses from the gravimetric measurements, we prepared a stock solution (here: working standard) by mixing definite amounts of both tracer dosing solutions on an analytical balance. Then, the mixture was diluted to meet the concentration range of the tracers in the samples. This avoids systematic errors from pipettes and graduated flasks, which might well be in the order of a few percent. To provide redundant information, a second working standard was
prepared in the field by pumping from both tracer solutions into a single container for a defined time.

*Sampling strategy* - We performed a time-proportional continuous composite sampling instead of time-proportional discrete sampling (Smith, 2001). This minimized random errors, because eventual fluctuations in tracer concentrations were integrated out. Although monitoring by ion-selective electrodes would have been desirable, we found that available devices for Lithium and Bromide performed unsatisfactory in the sewer environment.

*Additional discharge measurements* - We installed a discharge measurement device at the sampling point. This provided additional information to check the tracer data for systematic errors. The flow data can also be used for a more complex data analysis that accounts for dynamic flow effects, given that i) a second measuring device is installed at the beginning of the investigation reach and ii) the reach characteristics are appropriate, as presented below.

*Site suitability* - The QUEST-C method is conceptually applicable to any section of the sewer system where exfiltration estimates are desired. However, (Rieckermann et al., 2005) point out that the interpretation of estimated tracer mass loss as “loss of a proportion of discharge” is more difficult in a reach with considerable inflows, infiltration/exfiltration or non-uniform discharge.

### Data Analysis

In the following we present two approaches to analyze data from a QUEST-C experiment: the *simplified* approach assuming steady discharge and the *dynamic* approach. For each method we developed an uncertainty analysis framework.

#### A) Simplified approach

**Model:** In the simplified approach, the discharge is assumed to be steady during the experiment. This modifies equation 1 to:

\[
E = 1 - \left( \frac{c_{Br}}{c_{Li}} \frac{w_{Li}}{w_{Br}} \right) \cdot \frac{m_{Br}}{m_{Li}} \cdot \frac{C_{Li}}{C_{Br}} 
\]

(Eq.2)

where \(c_{Br}\) and \(c_{Li}\) are the tracer concentrations of the working standard, \(w_{Br}\) and \(w_{Li}\) are the masses of tracer solutions mixed to obtain the working standard, \(m_{Li}\) and \(m_{Br}\) are the dosing rates of the tracer solutions and \(C_{Li}\) and \(C_{Br}\) are the tracer concentrations in the sample.
Uncertainty analysis: For the simplified approach, we identified seven major sources of uncertainty which will be presented below. Uncertainty due to variations in the wastewater matrix, adsorption on sewer slimes, transmission errors in the measurement chain and numerical computation errors are considered negligible (Figure 3).

① Inaccuracy in the dosing rates - As dosing rates are recorded continuously, they can be checked for gross errors easily. In case that no irregularities occur, a linear model describes the decrease in weight adequately. Parameter standard errors are estimated by linear regression analysis.

② Incomplete mixing of tracer over the sewer cross-section - Generally, mixing can be considered complete when the coefficient of variation of concentration in the cross-section is less than 2% (Rutherford, 1994). In case that mixing by natural turbulence is found to be insufficient, it is recommended to improve it by external means (e.g., sewerage pumps).
Natural background concentration of tracer - The expected natural background of each tracer in the investigation reach must be tested beforehand and in case of significant background concentrations of the indicator tracer, a different substance should be chosen. Background concentrations of the reference tracer can be corrected through additional background sampling at the dosing point.

Adsorption of tracer to wastewater solids - Any significant adsorption of tracer to wastewater solids introduces a bias in the computed exfiltration ratio. Tracer behavior should be assessed by laboratory batch tests.

Concentration measurement errors - The measurement error is estimated from repetitive measurements on the analytic device. Systematic errors are avoided by measuring a working standard instead of performing direct concentration estimations.

Error of analytical balance - The analytical balance is used to produce the working standard with a high accuracy. Information on its expected precision can be obtained from the manufacturer.

Error due to unsteady flow transport phenomena - The separation of wave and fluid causes variations of the tracer ratio in the samples (here: transport error). It has to be considered that discharge fluctuations in time result in waves which travel at a higher velocity than the main water body (Rieckermann et al., 2005). (Henderson, 1966) estimated that, under the assumptions of a wide rectangular channel and a constant friction coefficient, kinematic waves travel with a speed of 5/3 of the mean velocity. For the QUEST-C method, this means that the two tracer substances that have been dosed to the same flow element are diluted differently at their dosing points. 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 (Figure 3).

Full error propagation - As equation 2 is linear in the parameters, the overall random error is assessed by Gaussian error propagation.

B) Dynamic approach

Model: When additional discharge measurements \( Q(t) \) are available, the exfiltration is computed from the ratio of tracer loads instead of the ratio of tracer concentrations:

\[
E = 1 - \left( \frac{c_{Br}}{c_{Li}} \right) \left( \frac{w_{Li}}{w_{Br}} \right) \left( \frac{m_{Br}}{m_{Li}} \right) \frac{\int Q(t) \cdot C_{Li}(t) dt}{\int Q(t) \cdot C_{Br}(t) dt}
\]  
(Eq.3)
Uncertainty analysis: From a conceptual point of view further information should lead to more accurate exfiltration estimates. However, additional errors to those mentioned above are also introduced (Figure 3):

Errors in discharge measurement - Standard flow measurement devices compute the flow from area-velocity measurements. Systematic errors, affecting the calculation of the cross-sectional area, stem from the installation procedure of the device \( h_{\text{sys}} \) and the sewer diameter estimate \( d_{\text{sewer}} \). Random errors affect the water level \( h_{\text{rand}} \) and velocity \( v \).

Integration error caused by limited time resolution of flow monitoring and sampling - Discharge measurements generally have a higher time resolution than composite samples. For the computation of the tracer loads, the sample concentrations are assumed to be representative for their respective sampling interval. Although this seems reasonable because of the constant tracer dosing, it causes some error in the computed exfiltration. In the data analysis procedure, this uncertainty is considered by a bootstrap resampling procedure, which will be discussed in the next paragraph.

The available discharge information makes it possible to estimate the probability distribution of the transport error. For this purpose, we suggest hydrodynamic transport modeling in combination with bootstrap resampling (Efron and Tibshirani, 1998): First, the continuous tracer concentrations at the measuring point should be simulated with a numeric model that uses the discharge recorded during the experiment. Synthetic samples are then created according to the experimental sampling scheme by drawing time-proportional composite samples from the simulated concentration time series, maintaining the time order of measurements but randomly determining the starting point of the integration \( t_0 \) (Figure 4).

From the synthetic samples, the ratio of tracer loads is computed for the duration of the entire experiment. The ratio of tracer loads, together with the tracer input information, is used to calculate exfiltration with equation 3. This is done repeatedly, resulting in a distribution of exfiltration estimates that reflects the uncertainty due to transport phenomena in the investigation reach and the experimental sampling scheme. The procedure is valid so long as it can be assumed that the model is sufficiently calibrated to represent realistic transport characteristics of the reach. To calibrate the hydrodynamic model, additional tracer experiments with inline measurements are essential.

Full error propagation - For the dynamic approach to data analysis, we propose a Monte Carlo approach, which makes it possible to assess error contributions from field and laboratory measurements together with errors from transient transport phenomena and the applied sampling scheme in one framework.

Values for the tracer concentrations in the samples \( (C_{Li}, C_{Br}) \) and the working standard \( (c_{Li}, c_{Br}) \) are drawn from the specified distributions. Similarly, values are drawn from distributions of the dosing rates \( (m_{Li}, m_{Br}) \), which are obtained from regression analysis on the balance data.
Masses of tracer solutions in the working standard (wBr and wLi) are drawn, considering the error of the analytical balance. The error of the velocity readings (v) can also be assessed with information from the manufacturer, whereas the systematic and random error contributions in the water level and sewer diameter (h_syst, h_rand, d_sewer) should be assessed from practical experience. Values of the error due to the transport phenomena and the composite sampling scheme are drawn from the error distribution resulting from the bootstrap resampling.

For each parameter set, an exfiltration estimate considering only the measurement errors is first computed with equation 3. Then, it is corrected for the transport error to obtain a final estimate (Figure 3). This process can be repeated several hundred of times to yield a distribution of final exfiltration estimates. In the following section we present a case study to evaluate the practical applicability of the method and to determine the accuracy of the computed results with regard to precision and eventual bias.

Figure 4. Schematic representation of the computation of synthetic tracer load ratios from simulated concentration time series and discharge data. Li.sim, Br.sim = simulated tracer concentrations, $C_{Li,r}$ and $C_{Br,r}$ = mean tracer concentrations in the k-th interval, $dt_{sample}$ = duration of sampling for one sample, $t_0$ = starting point of the experiment, $t_k$ = starting point of the k-th sample. In the proposed bootstrap approach, $t_0$ is varied repeatedly, resulting in a distribution of exfiltration estimates that reflects the error due to transport phenomena in the investigation reach and the experimental sampling scheme.
3  Case Study

Site description

A QUEST-C experiment was performed in a trunk sewer connecting the village of Rümlang to Oberglatt (CH). The investigation reach is 643 m long and the total length of the section amounted to 760 m. The sewer has a circular profile with a diameter of 0.9 m. The average discharge during dry weather is $24.4 \text{l s}^{-1}$ with an average water depth of 0.11 m and a mean velocity of 0.48 m $\text{s}^{-1}$. The investigation reach, which has a constant slope of 0.9‰, has no lateral inflows and is in very good structural condition and expected to be watertight, because it is located in the groundwater. This makes it possible to check the obtained results for bias. Although we acknowledge that this provides no full formal validation, we expect that it provides a more advanced plausibility check than (i) measuring reaches with pipe defects or artificial leaks (e.g., from pumping) or (ii) performing experiments under laboratory conditions. This is, because such alternative exfiltration estimates from (i) would be prone to relative large uncertainties in the order of a few percent (e.g., from discharge measurements) or the results from ii) would not be transferable to real-world conditions (e.g., sewer sediments and slimes).

Experimental design

Lithium Chloride was used as indicator tracer, because the natural background of Lithium in this sewer was negligible. A $15 \text{ g l}^{-1}$ Li$^+$ dosing solution was prepared to obtain the desired concentration in the sample. The reference tracer was a $25 \text{ g l}^{-1}$ Br$^-$ solution of Sodium Bromide. Peristaltic dosing pumps (ISMATEC BVK, ISMATEC MV-CA4) were used for the dosing of the tracer solutions and both containers with tracer dosing solution were placed on balances (OHAUS DP150) to record the dosing rates. The discharge during the experiment was measured with two flow meters based on the Doppler Ultrasonic Area-Velocity principle (SIGMA 950, American Sigma) which were installed upstream and downstream of the sewer reach. Complete cross-sectional mixing was ensured with sewerage pumps.

The experiment was conducted from 11:00 – 13:30 hrs which was identified as a period of almost steady flow from previous flow measurements. Time-proportional continuous composite sampling over 10 minutes was performed. At the measuring point two series of 10 samples were taken in parallel. Upstream the Bromide dosing point 10 samples were taken for Bromide background correction (Figure 2). The sampling at the different locations was synchronized with wooden floats. The samples were filtered immediately in the field through a cellulose acetate filter (0.45 µm) and later analyzed by ion chromatography (Metrohm, Compact IC).
After the experiment, the working standard was prepared by mixing 1.0108 g of the LiCl dosing solution and 1.7980 g of the NaBr dosing solution on an analytical balance (Mettler Toledo AB-S). As indicated above, this standard was diluted (1:10'000) to meet the concentration range of the wastewater samples. As the IC-suppressor solution (4 mmol/L tartaric acid, 1 mmol/L dipicolinic acid) causes peak areas to be not exactly proportional to the tracer concentration, this avoids systematic errors.

The analysis showed a very good reproducibility of ±1.2% of the measured value for Li\(^+\) and ±0.5% for Br\(^-\) (single standard deviation). Batch test results suggested a non-reactive behavior of the tracers in wastewater and in the sample bottles. During the development of the method, we found that analysis of Lithium in wastewater on ICP-OES (Spectro CIROS VISION) performed unsatisfactory. Lithium concentrations were overestimated by 45-47%, which was due to significant cross-sensitivities with other wastewater components. For data analysis we used the raw peak areas of the IC measurements from both samples and working standard. In the following, we will therefore use the subscript \(e_{q}\) (e.g., \(C_{eq}\)) to indicate the use of concentration equivalents instead of absolute tracer concentrations.

**Data Analysis**

**A) Simplified approach**

From two series of samples, the average exfiltration ratio was calculated to 1.2% (series 1) of the labeled flow, respectively to 1.0% for series 2 (Figure 5). The average of 1.1% was in good accordance with the expected result of zero exfiltration.

![Figure 5](image.png)

**Figure 5.** Concentration equivalents (Li\(^+\), Br\(^-\)) and computed exfiltration ratio from the QUEST-C experiment. Exfiltration was computed from the average of two series. The averaged discharge is plotted as a plausibility check, it is not used for data analysis with the simplified model.
Assessment of uncertainty:

Considering all 20 samples from series 1 and 2 we obtained a total standard error of 2.6% for the exfiltration estimate. All uncertainty contributions for the individual model parameters are summarized in Table 1. The largest uncertainty contributions stem from the Bromide and Lithium concentration equivalents in the samples ($C_{eq,Li}$ and $C_{eq,Br}$) (Table 1, column 5). This is due to errors of the analytical procedure, the sampling and wave separation during the experiment. However, these effects cannot be assessed separately, because no discharge information is used in the simplified model.

Table 1. Parameters of the simplified model, parameter uncertainties and corresponding error contributions from Gaussian error propagation.

<table>
<thead>
<tr>
<th>Parameter ($\theta_i$)</th>
<th>Unit</th>
<th>Description</th>
<th>$\mu_{\theta_i}$</th>
<th>$\sigma_{\theta_i,\text{random}}$</th>
<th>$\left( \frac{\partial E}{\partial \theta_i} \cdot \sigma_{\theta_i} \right)^2$</th>
</tr>
</thead>
<tbody>
<tr>
<td>$w_{Li}$</td>
<td>[g]</td>
<td>analytical balance</td>
<td>1.011</td>
<td>2.00E-04</td>
<td>4.74E-11</td>
</tr>
<tr>
<td>$w_{Br}$</td>
<td>[g]</td>
<td>analytical balance</td>
<td>1.798</td>
<td>2.00E-04</td>
<td>1.27E-08</td>
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<td>$m_{Li}$</td>
<td>[g s$^{-1}$]</td>
<td>reference mass input to sewer</td>
<td>1.77</td>
<td>3.67E-05</td>
<td>$\frac{\partial E}{\partial \theta_i} \cdot \sigma_{\theta_i}$ = 379.213</td>
</tr>
<tr>
<td>$m_{Br}$</td>
<td>[g s$^{-1}$]</td>
<td>indicator mass input to sewer</td>
<td>3.37</td>
<td>8.94E-05</td>
<td>7.20E-10</td>
</tr>
<tr>
<td>$c_{eq,Li}$</td>
<td>[IC units]</td>
<td>tracer concentration, standard</td>
<td>16.790</td>
<td>0.033</td>
<td>3.89E-06</td>
</tr>
<tr>
<td>$c_{eq,Br}$</td>
<td>[IC units]</td>
<td>tracer concentration, standard</td>
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<td>0.069</td>
<td>5.79E-06</td>
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<tr>
<td>$C_{eq,Li}$</td>
<td>[IC units]</td>
<td>tracer concentration, sample</td>
<td>11.125</td>
<td>0.150$^1$</td>
<td>1.85E-04</td>
</tr>
<tr>
<td>$C_{eq,Br}$</td>
<td>[IC units]</td>
<td>tracer concentration, sample</td>
<td>20.868</td>
<td>0.333$^1$</td>
<td>2.61E-04</td>
</tr>
</tbody>
</table>

1) The sample concentration uncertainties consider the uncertainty of the IC and the variability of the samples.
Figure 6. Variation of discharge during the experiment and the corresponding exfiltration ratio computed with Equation 3. The tracer loads were computed progressively from 10 min time-proportional continuous composite samples (average of two series).

B) Dynamic approach

We obtained an exfiltration ratio of 0.3% of the labeled flow for the average of the two sample series (Figure 6), which again was in accordance with our expectations.

Assessment of uncertainty:

*Exfiltration estimate considering measurement errors* - All error contributions for the individual model parameters are summarized in Table 2.
Table 2. Parameter uncertainties for the dynamic exfiltration model (Equation 3)\(^2\)

<table>
<thead>
<tr>
<th>Parameter ((\theta_i))</th>
<th>Unit</th>
<th>Description</th>
<th>(\mu_{\theta_i})</th>
<th>(\sigma_{\theta_i},\text{ random})</th>
<th>(\sigma_{\theta_i},\text{ systematic})</th>
</tr>
</thead>
<tbody>
<tr>
<td>(h)</td>
<td>[m]</td>
<td>water level measurement</td>
<td>(h_{\text{meas}})</td>
<td>0.006</td>
<td>0.02</td>
</tr>
<tr>
<td>(d_{\text{sewer}})</td>
<td>[m]</td>
<td>sewer diameter</td>
<td>0.9</td>
<td>--</td>
<td>0.015</td>
</tr>
<tr>
<td>(v)</td>
<td>[m s(^{-1})]</td>
<td>velocity measurement</td>
<td>(v_{\text{meas}})</td>
<td>0.05 (v_{\text{meas}})</td>
<td>n.c.</td>
</tr>
<tr>
<td>(w_{Li})</td>
<td>[g]</td>
<td>analytical balance</td>
<td>1.011</td>
<td>2.00E-04</td>
<td>--</td>
</tr>
<tr>
<td>(w_{Br})</td>
<td>[g]</td>
<td>analytical balance</td>
<td>1.798</td>
<td>2.00E-04</td>
<td>--</td>
</tr>
<tr>
<td>(m_{Li})</td>
<td>[g s(^{-1})]</td>
<td>peristaltic dosing pump</td>
<td>1.77</td>
<td>3.67E-05</td>
<td>n.c.</td>
</tr>
<tr>
<td>(m_{Br})</td>
<td>[g s(^{-1})]</td>
<td>peristaltic dosing pump</td>
<td>3.37</td>
<td>8.94E-05</td>
<td>n.c.</td>
</tr>
<tr>
<td>(c_{eq, Li})</td>
<td>[IC units]</td>
<td>tracer concentration, standard</td>
<td>16.790</td>
<td>0.033</td>
<td>--</td>
</tr>
<tr>
<td>(c_{eq, Br})</td>
<td>[IC units]</td>
<td>tracer concentration, standard</td>
<td>29.049</td>
<td>0.069</td>
<td>--</td>
</tr>
<tr>
<td>(C_{eq, Li})</td>
<td>[IC units]</td>
<td>tracer concentration, sample</td>
<td>(C_{eq, Li,\text{meas}})</td>
<td>0.012 (C_{eq, Li,\text{meas}})</td>
<td>n.c.</td>
</tr>
<tr>
<td>(C_{eq, Br})</td>
<td>[IC units]</td>
<td>tracer concentration, sample</td>
<td>(C_{eq, Br,\text{meas}})</td>
<td>0.005 (C_{eq, Br,\text{meas}})</td>
<td>n.c.</td>
</tr>
<tr>
<td>(\sigma_{\text{exf transport}})</td>
<td>[- ]</td>
<td>transport error</td>
<td>assessed separately</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

\(^2\) All error contributions were considered normally distributed with the specified mean values and standard deviations (“n.c.” = not considered, “--” = not required).

\(^3\) The sample concentration uncertainties only consider the uncertainty of the IC.

Uncertainty estimates that reflect practical knowledge were considered to be normally distributed in order to not over-predict the influence of marginal values in uniform error distributions. After (Evans et al., 2000) the standard deviation of a uniform distribution with \([-a, +a]\) was transformed into the standard deviation of a normal distribution \((\sigma)\) by

\[
\sigma = \frac{a}{\sqrt{3}} \quad \text{(Eq.}4\text{)}
\]

2000 simulations of equation 3 were performed for the Monte Carlo error propagation. The samples were drawn from the joint distribution of the model parameters without considering correlation of parameters or autocorrelation in time. For the average of the two series, the mean exfiltration ratio is computed to -0.3% with a standard deviation of 0.5% (Figure 8, left).
Transport error - To assess the error from transient transport phenomena, a numeric model of the Rümlang sewer reach was implemented in AQUASIM (Reichert, 1994) using the upstream discharge measurements, data of pumping rates and the concentrations of the dosing solutions. It was calibrated to concentration and discharge data of two previous experiments, for which inline data of conductivity were recorded. The grid space was chosen very narrow ($\Delta x = 0.33$m) to reduce numerical dispersion, which allowed for the estimation of a dispersion coefficient ($K = 0.02 \text{ m}^2\text{s}^{-1}$) together with a Manning-Strickler roughness coefficient ($k_{st} = 54.5 \text{ m}^{1/3}\text{s}^{-1}$). The very good agreement of modeled and observed discharge during the experiment (Figure 7, upper graphs) provides validation.

It can be seen, that Lithium and Bromide were affected differently by the dynamic flow pattern. In this specific constellation of flow pattern, transport characteristics of the reach, dosing procedure and sampling strategy, the magnitude of error from discrete sampling could be in the order of several percent (Figure 6, dots in the lower graph). Note that the short-time fluctuations in the Lithium concentration (black line in top diagram) are due to scatter in the upstream discharge measurements (not shown), which was propagated through the hydrodynamic model. As the water level readings were rather stable, this was mostly due to turbulence scattering the velocity readings. As we had no additional information on the velocity in the reach we chose to perform the simulation on the recorded data and discuss the consequences, instead of arbitrarily smoothing out the data. From the bootstrap resampling analysis, the transport error was computed to -0.8% with a standard deviation of 1.1% (Figure 8, centre).

Final corrected exfiltration estimate - The final estimate is assessed by combining the measurement and transport errors (Figure 8). A mean exfiltration of 0.5% is obtained with a standard deviation of 1.2%. The 95% confidence interval is estimated as [-0.027; +0.024]. To determine the relative importance of the various sources of uncertainty, sixteen error propagations were performed, each of which excluded the error from one source (not shown). This procedure has the advantage that correlated effects of the other sources are taken into account. The largest single error contribution was found to be the transport error, despite the fact that the study was performed at a time of rather steady discharge.
Figure 7. The impact of transient transport phenomena on the exfiltration ratio, investigated by hydrodynamic modelling; Top: measured ($Q_{meas}$) and simulated discharge ($Q_{sim}$), simulated tracer concentrations ($Li_{sim}$, $Br_{sim}$); Bottom: exfiltration computed progressively from the ratio of tracer concentrations (grey dots) and loads (black line).

Figure 8. Left: Exfiltration estimate considering the various measurement errors; Centre: error contribution from transient transport phenomena; Right: final exfiltration estimate, corrected for the transport error.
4 Discussion

The dynamic analysis of this QUEST-C experiment yielded a substantially lower uncertainty (1.2%) than the simplified analysis (2.6%), because some of the variability (e.g., transport error) could be captured by the more complex model structure. In general, this is a very promising result. However, two aspects must be discussed in more detail:

First, we must recall that the results of the hydrodynamic model are also uncertain because of model structure and parameter uncertainties. Therefore, the sensitivity of the transport error to parameter uncertainty was investigated. As parameter standard errors of $K$ and $k_{st}$ from the estimation procedure were considered inconclusive because of systematic patterns in the residuals, a scenario analysis was performed rather than an error propagation. We chose two extreme settings for the parameters $K$ and $k_{st}$, which lead to particularly smooth ($K_1 = 0.04 \text{ m}^2\text{s}^{-1}, k_{st1} = 50 \text{ m}^{1/3}\text{s}^{-1}$) or pronounced ($K_2 = 0 \text{ m}^2\text{s}^{-1}, k_{st2} = 60 \text{ m}^{1/3}\text{s}^{-1}$) tracer concentrations in comparison to the calibrated model. Computing the transport error for all three scenarios, it was found that they differed only by 0.2% in their standard deviations. Therefore, we considered the hydrodynamic model sufficiently calibrated in this particular case.

Second, it must be considered that the uncertainty analysis for the dynamic model is only feasible if all significant inflows into a reach can be monitored, which might be impractical in a network situation. However, the information of an independent flow meter at the measuring point is always recommendable as it allows for a plausibility check and makes it possible to compute exfiltration from tracer loads instead of concentration ratios, even when no uncertainty analysis is performed.

The uncertainty of the computed exfiltration ratio is to a large extent dependent on the data quality and a very careful performance of the experiment. However, it also depends on the characteristics of the reach (i.e., baseline of tracer, flow properties) and we agree with (Vollertsen et al., 2002) that the results of tracer studies must be interpreted “with care”, especially if (i) the time of the experiment has not been selected carefully, (ii) the suggested methodological improvements are not applied (e.g., gravimetric measurements) and (iii) no assessment of uncertainty has been provided. For these reasons, a general statement about the uncertainty of the methodology as suggested by (Knudsen, 1996) does not seem realistic and we absolutely recommend to perform an individual error assessment for each experiment according to the procedures discussed herein.

In our study, discharge variations were the most important source of error, and from the experience gathered in the European research project APUSS (Bertrand-Krajewski et al., 2005), we expect this to generally be the case when a continuous dosing procedure is applied. To a certain degree this can be counteracted by a prolonged sampling scheme. However, the duration of an experiment depends on many objectives: the location of the investigation reach
in the network, the daytime of the experiment, the financial budget and, which is often overlooked, the desired accuracy of the method.

The application of any of the tracer methods requires the investigator to make a number of decisions regarding experimental design. These include the number of tracer additions to use, the mass of tracer used in each addition, the sampling protocol, the starting time of the experiment, and the chosen model for data analysis. These choices influence the amount of uncertainty in the final estimate of exfiltration. However, there is no universal answer to the question what is the optimal choice of options. 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. These issues are addressed in greater detail in a further investigation, where we apply the formal framework of decision analysis to identify the most suited experimental design (Rieckermann et al., 2005).

In natural systems, discharge variations are considered less problematic for the applicability of the tracer method, because hydrological processes are often much slower. Yet, this is clearly a question of scale and might depend on the individual case. In contrast, when our method is applied to a river or stream, insufficient mixing is to be seen much more critical, because it can hardly be influenced technically. This might make the application of the method difficult in channels with a number of tributaries, highly irregular cross-sections or low velocities. Nevertheless, we believe that our tracer methods can also be a useful tool to investigate the spatial and temporal variations of surface- and groundwater interactions in-situ. Complementary information might be gained compared to other investigation methods (e.g. groundwater monitoring, remote sensing coupled with GIS). Furthermore, the estimation of uncertainty in the obtained result allows for quantitative risk assessment.

Although, regarding sewer maintenance and rehabilitation, the QUEST-C method is not intended to replace traditional CCTV inspections, urban water managers could use the information on sewer losses for efficient detection of extreme leaks, a cost-effective long-term monitoring of leakage or to consider sewer leakage in problem-oriented rehabilitation strategies.

5 Conclusions

In this study we demonstrated a robust experimental procedure to accurately measure losses from open channels with artificial tracers. Our main conclusions are:

- The QUEST-C method is useful to identify sewer leakage relative to the labeled flow, because it avoids many errors of previous studies.
- We expect the uncertainty in the result to depend more often on the reach characteristics and the applied model for data analysis than on the applied tracer substance.

- A general statement about the remaining uncertainty of the methodology is not realistic. Instead, we suggest to individually assess the error for each experiment.

- Future improvement of the tracer method should concentrate on the interpretation of experimental results in a network situation and to indicate possible limitations, where tracer methods do not perform satisfactory.

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We thank Mark E. Borsuk for the stimulating discussions and the Swiss Federal Office for Education and Science (BBW) for financial support. This study has been carried out within the framework of the European research project APUSS (Assessing Infiltration and Exfiltration on the Performance of Urban Sewer Systems) with partners at INSA de LYON (FR), EAWAG (CH), TUDRES (DE), LERMO (CZ), DHI (CZ), Hydroproject (CZ), MDX University (UK), LNEC (PT), Emschergenossenschaft (DE) and IRSA (IT). APUSS was supported by the European Comission under the 5th Framework Programme and contributed to the implementation of the Key Action "Sustainable Management and Quality of Water" within the Energy, Environment and Sustainable Development Contract n° EVK1-CT-2000-00072. APUSS is part of CityNet, a network of European research projects on integrated urban water management.
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APPENDIX C

Documentation and Availability
Appendix C.

Documentation and Availability

Both the *Pollutant Time Series Method* and the *Stable Isotope 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 have been described in scientific publications which have been submitted to peer-reviewed journals.

The algorithms for data analysis and statistical evaluation have been programmed in the R language (R Development Core Team, 2005) and are assembled in libraries. All libraries and code examples are freely available for public use. The complete software, documentation and exemplary datasets are made available for download from the APUSS homepage (see Appendix A) and will be added to the ETH e-collection (“http://e-collection.ethbib.ethz.ch/”).

References


R is available under the GNU public license: http://www.R-project.org.
Curriculum Vitae
Curriculum Vitae

2001 – 2006  Dissertation at the Swiss Federal Institute of Aquatic Science and Technology (Eawag), Duebendorf

2000 – 2001  Research assistant, University of Lausanne (Institute of Mineralogy and Geochemistry)

1992 – 2000  Diploma in Geology at the Ruhr University Bochum

1996 – 2000  Internships and consulting works (Stadtwerke Düsseldorf GmbH; Halbach und Lange GmbH; Bilfinger und Berger AG)

1997  IASTE traineeship, Tampere University of Technology (Institute of Engineering Geology)

1995 – 1999  Undergraduate research and teaching assistant, Ruhr University Bochum (Division of Applied Geology)

1991 – 1992  Foundation course in Experimental Physics, University of Essen

1980 – 1989  Gymnasium Essen-Überruhr, Germany

October 06, 1969  Born in Essen, Germany