Innovative tracer methods for infiltration monitoring

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ABSTRACT

Two innovative tracer methods for the quantification of parasitic discharge 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 article analyses the achieved state of development and demonstrates the application of the protocols with a comparative experimental study.

KEYWORDS

Infiltration; inflow; natural tracers; stable isotopes; pollutants; time series; APUSS

INTRODUCTION

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:

$$X_{Infiltration}(t) = \frac{C_{Foul\ Sewage}(t) - C_{Wastewater}(t)}{C_{Foul\ Sewage}(t) - C_{Infiltration}(t)} \quad (1) \quad Q_{Infiltration}(t) = X_{Infiltration}(t) \cdot Q_{Wastewater}(t) \quad (2)$$

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.

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: In 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 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 rare 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.

METHODOLOGIES

To overcome the above mentioned limitations, two methods for the quantification of parasitic discharges in sewers with natural tracers have been worked out:

Stable isotopes method

The approach uses the stable isotopes composition of mains water (proxy for the foul sewage) and local groundwater (proxy for the infiltrating water) as direct natural tracers. We focus on the investigation of the oxygen isotope ratios in this text. Values are cited in the δ-notation in reference to the international standard V-SMOW (IAEA 1995).

Compared to most other natural water characteristics, the stable isotopes composition 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. On a regional scale, the isotopic composition of water is predominantly controlled by the topographic elevation and continental setting of the region where ground water and surface water are recharged by precipitation (Craig 1961, Rozanski 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 (Kracht et al. 2003). 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 of importance 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 isotopes approach, the pollutants time series method does not require a direct investigation of drinking water or infiltration proveniences: The fraction of extraneous water is determined by analysing time series of pollutant concentrations and wastewater discharge measured at a single point in the sewer system. A characteristic feature of this method is the use of automatically operating in-line devices to obtain concentration time series with a high temporal resolution. 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 3). The model considers temporal fluctuations of the pollutant concentration in the foul sewage (equation 4) as well as time dependencies of the infiltration rate. For the latter, the amount of infiltrating water is conceptually divided into a constant baseflow and an exponentially receding interflow (equation 5):

$$C_{Wastewater,Model} = \frac{(Q_{Wastewater} - Q_{Infiltration}) \cdot C_{Foul\ Sewage} + Q_{Infiltration} \cdot C_{Infiltration}}{Q_{Wastewater}} \qquad (3)$$

$$C_{Foul\ Sewage} = f(t) + f(Q_{Foul\ Sewage}) \qquad (4) \qquad Q_{Infiltration} = Q_{Baseflow} + Q_{0,Interflow} \cdot e^{-k_{rec} \cdot (t - t_{0,Interflow})} \qquad (5)$$

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

Equation 3 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), for its concentration in the parasitic water is assumed to be negligible. $Q_{Infiltration}$ is then identifiable by fitting a modelled time series of pollutant concentrations to the measured data (Kracht and Gujer 2004).

FIELD APPLICATIONS

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 commune of about 5'400 inhabitants, located at the north-eastern boarder of the agglomeration of Zurich. The total length of its sewer system amounts to 23.1 km. The village has a mixed infrastructure 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 qualified for the stable isotopes 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.

Stable isotopes 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 declined north east to 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 in 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 fraction stems from precipitation in the Alps. Zurich lake water is about 1.8 ‰ ($\delta^{18}O$) respectively 11.5 ‰ (δ^2H) lighter compared to the local groundwater. 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 40 samples taken from monitoring wells (various screened sections form 2 to 10 meters below ground), two production wells (screened sections down to 17 m below ground), springs, surface water sites (local creeks) and artificial draining systems were considered. Furthermore, 21 drinking water samples taken from the mains network were analyzed for their isotopic composition.

The estimates for the expected average isotopic composition of the infiltrating water ($\delta^{18}O_{Infiltration} \approx \delta^{18}O_{Groundwater}$) and the foul sewage ($\delta^{18}O_{Foul\ Sewage} \approx \delta^{18}O_{Drinking\ Water}$) were calculated as the arithmetic means of these sample series (equation 6). Standard uncertainties were predicted from equation 7, where empirical distributions are corrected for imprecision of the applied analytical method. Herewith we differentiated the magnitude of variance in the empirically observed distributions that was stemming from natural variability from the magnitude of variance being introduced by the laboratory measurements.

$$\delta^{18}O_{Mean} \approx \overline{\delta^{18}O_{Sample}} = \frac{1}{n} \sum_{Sample=1}^{n} \delta^{18}O_{Sample} \tag{6}$$

$$\sigma(\delta^{18}O_{Mean}) \approx \sqrt{\left(\frac{1}{n-1} \sum_{Sample=1}^{n} (\delta^{18}O_{Sample} - \overline{\delta^{18}O_{Sample}})^2\right) - \sigma^2_{Laboratory} + \left(\frac{\sigma_{Laboratory}}{\sqrt{n}}\right)^2} \tag{7}$$

Considering a laboratory standard uncertainty of $\sigma(\delta^{18}O)_{Laboratory} = \pm\ 0.08$ ‰ we obtained $\delta^{18}O_{Infiltration} = -9.54 \pm 0.17$ ‰ and $\delta^{18}O_{Drinking\ Water} = \delta^{18}O_{Foul\ Sewage} = -11.31 \pm 0.02$ ‰. Figure 1.1 compares these endmember ranges with the $\delta^{18}O$ values of a series of 28 hourly waste-water samples. As expected, the wastewater is isotopically enriched during night time, indicating a higher fraction of "heavy" infiltrating water being present in the sewer. During day time the wastewater shows isotopically depleted values, due to the larger amount of "lighter" foul sewage being discharged. Based on these data, the infiltration ratio and the corresponding infiltration discharge are calculated (Figures 1.3 and 1.5).

The accuracy of these 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:

- $\delta^{18}O_{Infiltration}$ and $\delta^{18}O_{Foul\ Sewage}$: normal distributions with mean values and standard deviation as given above
- $\delta^{18}O_{Wastewater}$: independent normal distributions with mean values as measured for each single sample and standard deviation $\sigma(\delta^{18}O)_{Laboratory} = 0.08$ ‰
- Flow measurements are presumed to be affected by a maximum absolute offset error of $\pm\ 1.2$ l/s (15 % of the minimum in the time series) and a maximum relative error of $\pm\ 15\%$. For the MCS we used normal distributions with expected values being zero and adjusted standard deviations being $(1.2/\sqrt{3})$ l/s and $(0.15/\sqrt{3})$ respectively.

A stochastic sample of 10000 hydrograph separations was generated and integrated over time. To obtain truthful results, the following scheme was applied: In each run exactly one independent sample for $\delta^{18}O_{Infiltration}$ and $\delta^{18}O_{Foul\ Sewage}$, for both the absolute and the relative discharge errors and for each of the hourly $\delta^{18}O_{Wastewater}$ value of the sample series was drawn.

Pollutant time serie method
COD-equivalents were measured in 2 minutes intervals with a submersible UV-VIS spectrometer (Figure 1.2). The instrument (spectro::lyser, s::can Messtechnik) was calibrated with a series of laboratory samples taken in parallel. A mixing model according to equations 3 to 5 was implemented, in which equation 4 was specified by combination of a harmonic respectively polynomial term:

$$f(t) = A \cdot \sin(freq \cdot 2 \cdot \pi \cdot (t - phase)) \quad (8) \qquad f(Q_{Foul\ Sewage}) = a + b \cdot Q_{Foul\ Sewage} \quad (9)$$

Values for $Q_{Baseflow}$, $Q_{0,Interflow}$, k_{rec}, a, b, A and *phase* were estimated by minimizing the sum of squared residuals between the measured values and the modelled time series $COD_{Wastewater,\ Model}$ (*freq* was fixed to 1 day^{-1}). The resulting hydrograph decomposition is displayed in figures 1.4 and 1.6. The accuracy of the baseflow to interflow separation was strongly affected 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 isotopes approach
- COD measurements are presumed to be maximally affected by an offset error of ± 45 mg/l (10 % of the average COD concentration in the time series) and a relative error of ± 15%. For the MCS we used normal distributions with expected values being zero and adjusted standard deviations being (45/√3) l/s and (0.15/√3) l/s respectively.
- 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 revealed 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.

RESULTS AND DISCUSSION

The hydrograph separation obtained with the two methods clearly depends on the underlying assumptions (Figure 1). The stable isotopes 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 discharge of clean water with predefined flow characteristics (constant baseflow, exponentially receding interflow). However, the partition obtained with the stable isotopes approach apparently reveals certain intraday fluctuations of $Q_{Infiltration}$ that are not covered by the model definitions of the time series approach. Such fluctuations can be caused by larger dead zones and hydrodynamic or backwater effects.

In practical applications, we are interested to assess 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 $Q_{Infiltration}$ and $X_{Infiltration}$ over 24 hours that where obtained from MCS applied to both methods. Table 1 summarizes the calculated means and 2.5 to 97.5 percentiles. The 2.5 to 97.5 inter percentile range for $Q_{Infiltration,\ 24h-total}$ equals about ± 11 % of the total wastewater discharge for the stable isotopes and ± 9 % for the pollutant time series method. The infiltration ratio is estimated more precisely with both methods. This has been expected, since flow measurement errors largely cross out in the computation of $X_{Infiltration}$. Compared to the stable isotopes approach, the pollutant time series method seems more robust with respect to the considered errors (more narrow probability distributions). However, it has to be clarified that the analysis did not account for possible model structure uncertainties. Furthermore, we predicted the uncertainties of the isotopic endmember values based on the most unfavourable hydrological assumption that the δ^{18}O-values obtained from the individual groundwater samples would not be mixed (averaged) in the aquifer: Equation 9 presumes that with an 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.
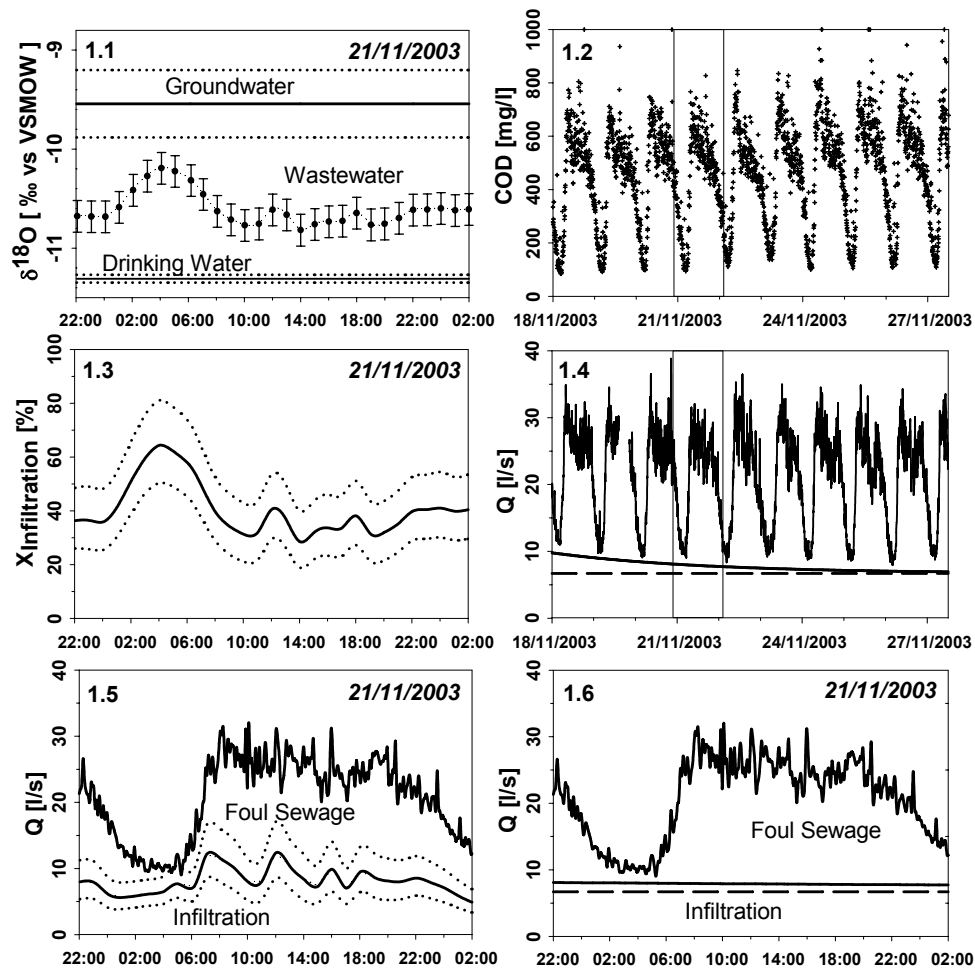
Figure 1. Measurements in the Rümlang trunk sewer (November 2003)
Stable isotopes method: 1.1: estimated isotopic compositions (dotted lines and bars: 2σ standard uncertainty); 1.3: infiltration ratio; 1.5: hydrograph separation (dotted lines: 95 % conf. interval, not shown for $Q_{Wasterwater}$)
Pollutant time series method: 1.2: registered COD time series; 1.4: hydrograph separation (full line: bulk infiltration to foul sewage, dashed line: baseflow to interflow); 1.6: zoom out of 1.4 to be compared with 1.5
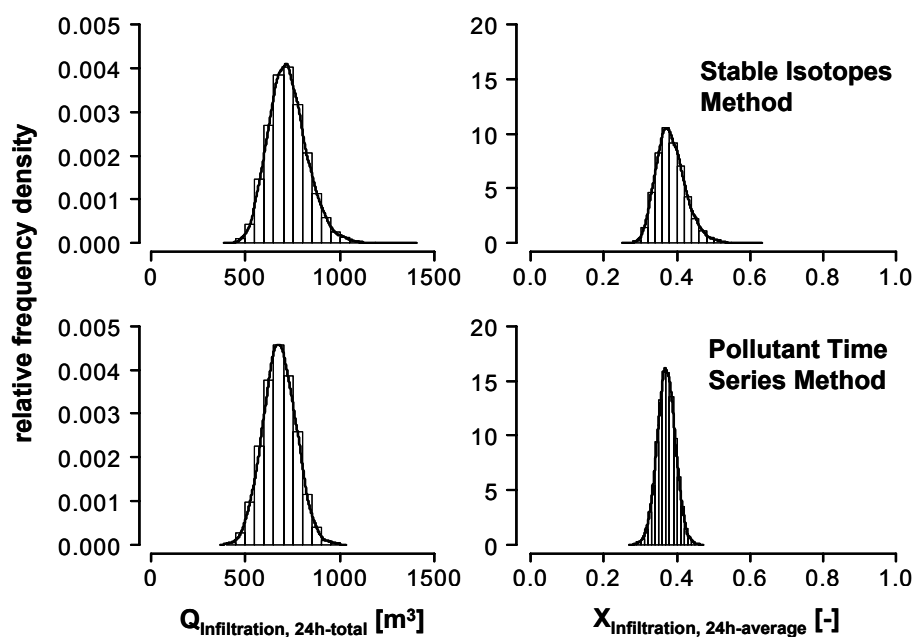

Figure 2. Monte Carlo simulation of infiltration estimates integrated over 24-hours (21.11.2003)

Table 1. Comparison of the results obtained with different approaches (data refer to 21/11/2004)

|	stable isotopes	pollutant time series	minimum night flow	water balance
$Q_{Infiltration, 24h-total}$	725 (546 - 943) m^3	682 (516 - 854) m^3	780 (605 - 959) m^3	-
$X_{Infiltration, 24h-average}$	38 (32 - 47) %	37 (32 - 42) %	42 (38 - 46) %	41 (30 - 52) %

Table 1 further lists infiltration estimates obtained by the classical "night time minimum" assumption and by a water balance between drinking water delivery and wastewater discharge. All results agree in a plausible way. 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 calculated for the whole village, whereas the subcatchment investigated by the other methods approximately covered only three quarters of it for technical reasons. Therefore only a comparison of $X_{Infiltration}$ is meaningful.

Required boundary conditions for the stable isotopes method
The two critical system properties for the use of the stable isotopes method are the existing isotopic separation between drinking water and infiltration and the natural variability of possible infiltration sources. The accuracy will further depend on the actual infiltration ratio and the uncertainty of the estimates for $\delta^{18}O_{Foul\ Sewage}$ and $\delta^{18}O_{Wastewater}$. Presuming a suitable low variability of the two mixing endmembers, the scatter being introduced by the imprecision of the laboratory analysis can be reduced by taking an appropriate number of samples. In the case study both $\sigma(\delta^{18}O_{Wastewater})$ and $\sigma(\delta^{18}O_{Foul\ Sewage})$ were effectively reduced from 0.08 to values below 0.02 ‰. $\sigma(\delta^{18}O_{Infiltration})$ was slightly reduced from 0.19 to 0.17 ‰. Figure 3 gives a simplified overview of the required conditions to determine the infiltration ratio $X_{Infiltration}$ with a desired 95%-confidence interval of 7.5 %, 10 % or 15 % of the total wastewater respectively.

Required boundary conditions for the pollutant time series method
Some dynamic of the quotient $Q_{Infiltration}$ / $Q_{Wastewater}$ is required for successful parameter estimation from the measured data. This condition is usually fulfilled by the diurnal variations of the wastewater discharge. However, problems may arise in the case of very large sewer networks and / or extremely high infiltration ratios. The identifiability of the applied set of model
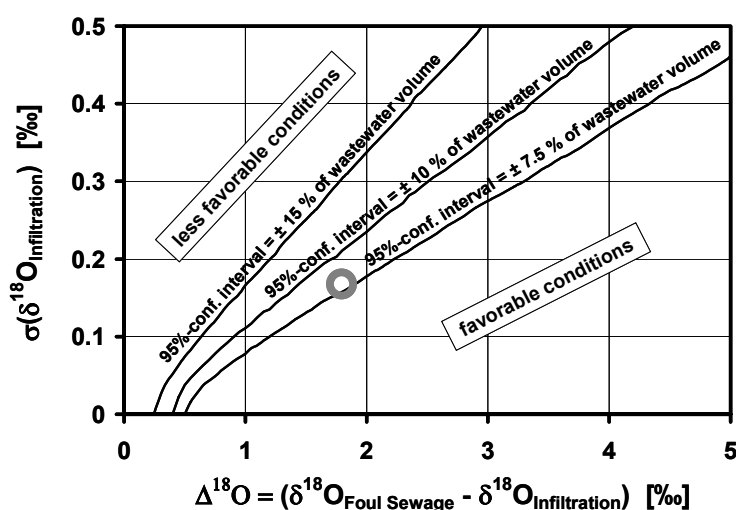
Calculation basis:
$X_{Infiltration}$ = 40 %
$\sigma(\delta^{18}O_{Wastewater})$ = ± 0.02 ‰ (40 samples)
$\sigma(\delta^{18}O_{Foul\ Sewage})$ = ± 0.02 ‰ (21 samples)

Grey circle marking the Rümlang example:
$\Delta^{18}O_{Foul\ Sewage-Infiltration}$ = 1.8 ‰
$\sigma(\delta^{18}O_{Infiltration})$ = ± 0.17 ‰ (25 samples)

The diagram must be recalculated in the case of a differing $\sigma(\delta^{18}O_{Wastewater})$ or $\sigma(\delta^{18}O_{Foul\ Sewage})$.

Figure 3. Required boundary conditions for the stable isotopes method.

parameters further depends on the structure of the individual hydrograph and pollutograph. This has to be handled with care, especially when refinements of the basic model are introduced (i.e. equations 10, 11).

Successful application of the tracer methods depends on the local boundary conditions at the investigation site. A final statement on the overall precision is not possible. In practice it will be required to asses the confidence in the obtained results for the individual case.

CONCLUSIONS

Two novel methods for the chemical separation of wastewater hydrographs have been demonstrated and evaluated by a comparative experimental study: The stable isotopes method uses the different isotopic signatures of mains water and infiltrating water as a direct natural tracer. A sufficient isotopic separation can exist, when drinking water originates from a distant hydrological regime, but parasitic water stems from subsurface waters recharged by local precipitation. The pollutant time series method estimates the fraction of infiltrating water from a combined analysis of measured time series of pollutant concentrations and wastewater discharge. 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 methodologies are of high future interest, as they avoid the conventional hypothesis that the hydrographs diurnal night-time through equals the amount of extraneous discharge.

ACKNOWLEDGEMENTS

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