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# Data evaluation of full-scale wastewater treatment plants by mass balance

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ABSTRACT

Measured data of wastewater treatment plants (WWTPs) often contains errors. These errors can prohibit the use of WWTP data for process evaluation, process design, benchmarking or modelling purposes. In this paper a practical stepwise methodology is presented to check WWTP data using mass balances. The presented results show that poor WWTP data quality leads to large errors when calculating key operational conditions such as the solids retention time (SRT), oxygen consumption (OC) and the different internal conversions rates. By improving WWTP data quality using mass balance calculations useful new information becomes available for process evaluation, WWTPs design and benchmarking.



## 1. Introduction

Stricter effluent demands for wastewater treatment plants (WWTPs) can result to the use of more energy and chemicals. To meet these demands WWTP designs become more complex and difficult to operate (Olsson, 2006). To manage the increasing costs for wastewater treatment, reliable process information is a first requirement. With this information, operation can be optimised and designs can be improved. In this sense, the WWTP performance benchmark has showed to be an effective management tool to compare WWTPs and draw conclusions of general validity (Benedetti et al., 2006; Nyserda, 2008; Unie van Waterschappen, 2003). Lindtner et al. (2004) proposed a methodology for the comparison of technical and economical process indicators for WWTPs with different process and operation modes.

Historically, the information collected on wastewater treatment is focussed on discharge legislation. To meet the legislation, effluent discharge and overall removal efficiency have to be reported. Therefore, in WWTP practice, mainly influent and effluent are measured. These data, combined with financial data from WWTP operation, are the main source of current benchmarks for comparison of operational effectiveness. These benchmarks have provided water boards

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the means to reduce cost, mainly by negotiating adventitious contracts for energy and chemicals supply. However, current benchmarks do not provide the appropriate operational information for (cost) optimization of individual WWTPs. This information often is available (e.g. the solids retention time (SRT), sludge loading and the oxygen demand). However, the quality of this information often is insufficient for WWTPs management purposes.

To improve the quality of WWTP information, Meijer et al. (2002) proposed a method based on mass balance calculations. This method was adapted from earlier work on fermentation processes (Van der Heijden et al., 1994a). This method was successfully tested on several full-scale WWTPs in the Netherlands (Meijer et al., 2001, 2002). The approach resembles a data administration with checks and (mass) balances. It detects errors, improves the data quality and provides accurate information concerning internal conversion rates (e.g. nitrification, denitrification, phosphorus removal, etc.). The improved information is well suited for WWTPs evaluation and could also be used for benchmarking and modelling purposes.

To obtain reliable information from raw WWTP data, a WWTP mass balance administration has to be put into practice. This requires a different measurement strategy then now often used. For mass balance calculations all in- and outgoing flows including the activated sludge composition and sludge production should be measured, thus not only the influent and effluent. With the proposed methodology, the quality of WWTP information can be improved dramatically, as also shown in this paper, and thereby becomes suited for both financial and technical benchmarking. This improved benchmark provides managers and engineers a tool for assessment, design and optimization of WWTPs.

Mass balancing is a well-known technique and widely used in engineering (Adgate et al., 1998; Baker and Hites, 2000). The quality of process data in e.g. chemical/petrochemical industries significantly affects the performance and the profit (Narasimhan and Jordache, 2000). However, the technology rarely is applied in WWTPs practice. Without a proper check of WWTP data, the measured information will probably contain large errors. In general, process engineers' deal with data uncertainty by applying (large) safety factors in design and operation (Bixio et al., 2002). However, in the field of wastewater treatment, the magnitude of these errors and its effect on the calculated operational conditions has not been previously investigated.

Nowak et al. (1999) already pointed out that mass balances over WWTPs are excellent tools to estimate the fluxes of substances, compare operational conditions and draw conclusions of general validity. Application of mass balances on WWTP data is difficult because the process is dynamic and the variability of the influent loading is unknown. Brdjanovic et al. (2000) proposed the estimation of the SRT from a total phosphorus (TP) balance as a more accurate method than calculating the SRT from the measured sludge waste. Meijer et al. (2002) showed how to apply integrated mass balances on WWTPs data to improve the general data quality. For this approach, a practical methodology needed to be developed.

This paper presents a practical stepwise methodology to check WWTP data using mass balances. It is shown how errors in WWTP data affect the accuracy of calculated key design parameters like the SRT and internal conversion rates. Also a practical stepwise methodology is presented that can be used to check WWTP data on errors and improve the data quality using reconciliation.

## 2. Materials and methods

### 2.1. Description of the full-scale wastewater plant

The WWTP studied in this case is Deventer WWTP located in the Netherlands and operated by the Duch Waterboard "Groot Salland". The WWTP was a 182.000 P.E. (influent 10140 $m^3\,day^{-1}$) modified UCT-process according to the BCFS®-concept (Van Loosdrecht et al., 1998). The process was designed for biological phosphorus removal with a special feature where phosphate could be recovered via an in-line anaerobic stripping process. The operational and design parameters of the full-scale WWTP are presented in Table 1.

Following gravitational primary treatment, the settled sewage is introduced in two parallel operated activated sludge units, each consisting of seven concentric compartments (Fig. 1): stripper reactor (PS), two anaerobic reactors (R1 and R2), a contact tank (R3), an anoxic reactor (R4), an aerated alternated reactor (R5) and an aerated rector (R6). The activated sludge is settled in six secondary settlers (C1). After settling, the effluent is discharged into the nearby recipient.

### 2.2. Analytical methods

The WWTP was intensively sampled during 3 separate days (7th, 13th and 20th of December 2005). All measurements were performed during dry weather flow conditions. The flow scheme of the WWTP is presented in Fig. 1. 24-h composite samples were taken from the influent, waste sludge and the effluent. Grab samples were taken during high loading in the morning from the stripper effluent and the reactor tanks. The samples were analysed on total COD (TCOD), $NH_4^+$ (ammonium), $NO_3^-$ (nitrate), TKN (total Kjeldahl nitrogen), TP (total phosphorus), $PO_4^{3-}$ (ortho-phosphate) and MLSS (mixed liquor

**Table 1 – Design parameters of the WWTP Deventer**

| Process unit | Volume ($m^3$) | Depth (m) | HRT (h) |
|---|---|---|---|
| Stripper tank, $P_1$ | 181 | 5.35 | 3.6 |
| Anaerobic tank 1, $R_1$ | 370 | 5.35 | 0.9 |
| Anaerobic tank 2, $R_2$ | 680 | 5.25 | 0.9 |
| Anoxic contact tank, $R_3$ | 578 | 5.20 | 0.8 |
| Anoxic tank, $R_4$ | 2395 | 5.17 | 1.8 |
| Alternate aerated tank, $R_5$ | 3191 | 5.12 | 1.4 |
| Aerobic tank, $R_6$ | 3074 | 5.06 | 0.8 |
| Total clarifiers, $C_1$ | 12215 | 2.50 | 13.0 |
| Total WWTP (without settlers) | 10469 | – | 24.8 |
| Total WWTP (including settlers) | 22684 | – | 53.7 |

The HRT was calculated from the total actual flow rates through each compartment under dry weather conditions.

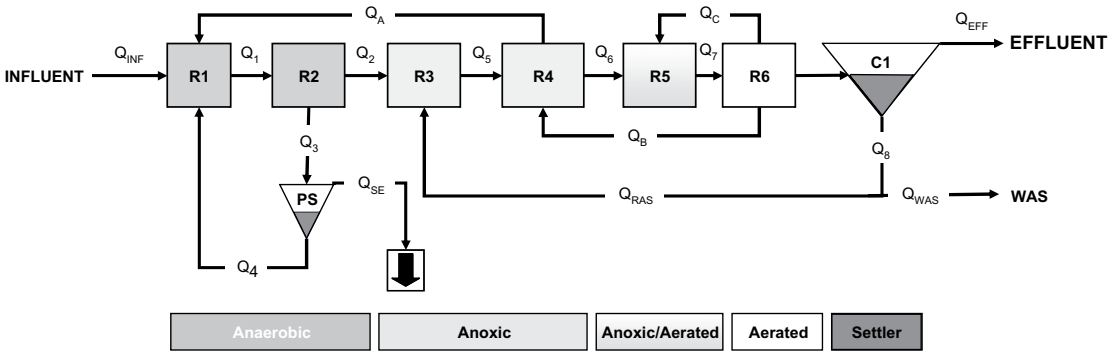


**Fig. 1 – Process flow diagram of the full-scale WWTP.**

suspended solids). Total nitrogen (TN) was calculated as the sum of TKN and $NO_3^-$ concentrations as mgN $L^{-1}$.

### 2.3. *Principles of mass balance evaluation*

Fig. 2 represents a simple overall WWTP mass balance in the solid, liquid and gas phase. As shown in Fig. 2, the main conversions in a WWTP are directly related to the $CO_2$ gas produced (COD conversion) and the $N_2$ gas produced (N conversions). These balances can be made for the total WWTP as well as, for the individual tank reactors. Because the main conversions are open balances – having a component in the gas phase that usually is not measured – these balances cannot be closed by direct measurement. Conversions only can be calculated from the measurable flows. An error in one of the measured flows thereby, automatically results in an erroneous calculated conversion. To be able to check the open balances, additionally the closed water and TP balances were calculated (Eqs. (1) and (2)). In chemical P-removal plants the salt added (e.g. Iron) could be a good element for mass balancing as well as the inorganic suspended solids concentration (Ekama et al., 2006). As shown in Fig. 2, phosphorus is a very suitable parameter for mass balance evaluation since TP only follows the wastewater flows and does not leave the system via the gas phase. In practice this means that, by measuring TP in all in- and outgoing flows, the water flows can be checked by solving the TP balance. After checking them, the internal conversions from the open COD and N balances can be calculated reliably by measuring the in- and outgoing COD and N components of the desired process unit.

The overall flow balance:

$$Q_{INF} - Q_{WAS} - Q_{EFF} - Q_{SE} = 0 \text{ (Closed)} \quad (1)$$

The overall TP balance (expressed as kg P $d^{-1}$):

$$TP_{INF} - TP_{WAS} - TP_{EFF} - TP_{SE} = 0 \text{ (Closed)} \quad (2)$$

The overall TN balance (expressed as kg N $d^{-1}$):

$$TN_{INF} - TN_{WAS} - TN_{EFF} - TN_{SE} = N_2 \text{ (denitrified\_load)} \neq 0 \text{ (non\_closed)} \quad (3)$$

Using the above methodology not only the consistence of the available data can be checked, but also it is possible to reconcile and improve the general data quality (Van der Heijden et al., 1994a, 1994b, 1994c). If the consistency is proven not to be sufficient, it is necessary to perform additional measurements in the plant. For example, in Eq. (2) all terms on the balance can be measured and should give zero as result. If the deviation is too large there is an indication for erroneous measurements or an incorrect process description (according to Fig. 1). Since mass balances are coupled (e.g. Eqs. (1) and (2)) it has advantage to solve the system of coupled mass balances simultaneously (Van der Heijden et al., 1994a, 1994b, 1994c). If the system of mass balances does not hold, errors will be calculated in the main operational conditions (such as the SRT, sludge production, oxygen consumption, nitrogen

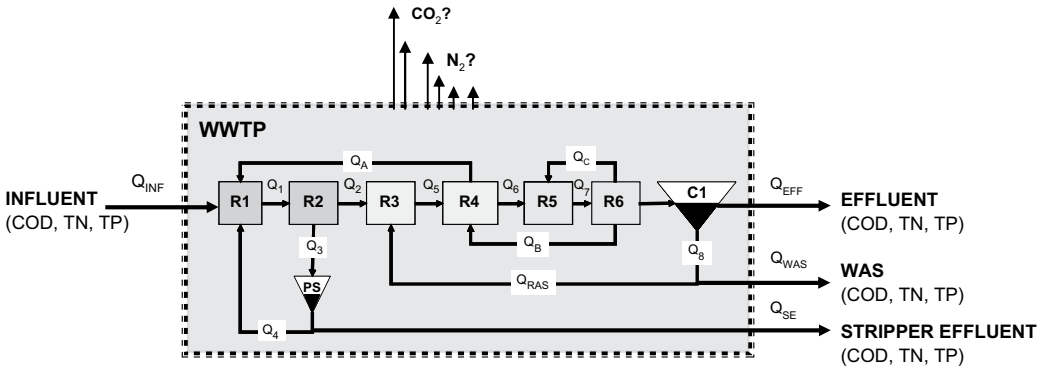


**Fig. 2 – Figure indicating mass balances over a complete WWTP.**

conversions and dinitrogen gas produced). Meijer et al. (2001) presented a detailed overview of this procedure within the activated sludge context, including the use of data reconciliation techniques based on the work of van der Heijden et al.

### 2.4. Principles of SRT calculation

The SRT is the most crucial parameter for activated sludge design, as it affects the treatment process performance, aeration tank volume, sludge production and oxygen requirements (Metcalf and Eddy, 2003). SRT should be known with an accuracy of approximately 95% to be able to perform a reliable simulation study (Brdjanovic et al., 2000; Meijer et al., 2001). A check on the SRT (or sludge production) is therefore an essential requirement for a reliable modelling study.

The classical SRT calculation is based on the sludge production (Eq. (4)). Measuring the solids in the waste activated sludge (WAS) or return activated sludge (RAS) flows however, are inaccurate due to large fluctuations in the settled solids concentration; $X_{WAS}$ (Meijer et al., 2001). Mass balancing brings the possibility of an alternative method of determining the WAS load and thus, calculating the SRT. There are several different ways to calculate the SRT. These are based on TP leaving the process (Eq. (5)), on TP entering the process, eliminating the need to know exactly the excess sludge wastage rate (Eq. (6)), or via COD particulate ($COD_{TSS}$) leaving the process via the WAS (Eq. (7)). It is possible to calculate the particulate phosphate in the sludge ($TP_{TSS}$) as the difference between the total phosphorus minus the soluble phosphate. The $TP_{TSS}$ is best measured at the end of the aerobic zone where the phosphorus uptake is at its maximum and the excess sludge is retrieved. The SRT calculation (Eqs. (4)–(7)) can be simplified if the stripper tank is not used or not available.

$$SRT_{classical} = \frac{V_{REACTOR} \times TSS_{REACTOR}}{(Q_{WAS} \times TSS_{WAS}) + (Q_{EFF} \times TSS_{EFF}) + (Q_{SE} \times TSS_{SE})} \quad (4)$$

SRT calculated from the conserved TP balance based on TP leaving the process:

$$SRT_{TP_{TSS\_OUTPUTS}} = \frac{V_{REACTOR} \times TP_{TSS\_REACTOR}}{(Q_{WAS} \times TP_{TSS\_WAS}) + (Q_{EFF} \times TP_{TSS\_EFF}) + (Q_{SE} \times TP_{TSS\_SE})} \quad (5)$$

SRT calculated from the conserved TP balance based on TP entering the process:

$$SRT_{TP_{TSS\_INPUTS}} = \frac{V_{REACTOR} \times TP_{TSS\_REACTOR}}{(Q_{INF} \times TP_{INF}) - (Q_{WAS} \times PO_{4\_WAS}) - (Q_{EFF} \times PO_{4\_EFF}) - (Q_{SE} \times PO_{4\_SE})} \quad (6)$$

SRT calculated from the COD particulate leaving the process:

$$SRT_{COD_{TSS\_OUTPUTS}} = \frac{V_{REACTOR} \times COD_{TSS\_REACTOR}}{(Q_{WAS} \times COD_{TSS\_WAS}) + (Q_{EFF} \times COD_{TSS\_EFF}) + (Q_{SE} \times COD_{TSS\_SE})} \quad (7)$$

### 2.5. Principles of error calculation

In this paper, the average measurement is expressed as the arithmetic mean ± the standard deviation ($s_D$) over the 3 measurement points. The coefficient of variation (CV) (ratio of the standard deviation to the mean) was calculated as a measure of dispersion of a probability distribution. In this paper, CV is expressed as a percentage and referred to as the relative standard deviation (RSD).

When data containing errors (e.g. $a \pm \Delta a$ and $b \pm \Delta b$) are used to calculate operational conditions, the errors ($\Delta a$ and $\Delta b$) propagate in the equation according to the following propagation rules: the absolute error of a sum equals the sum of the absolute errors according to $\Delta a + \Delta b = \Delta c$, and the relative error of a product or a quotient equals approximately the sum of the relative errors according to $\Delta a/a + \Delta b/b$. For a series of random samples the error of measurement can be expressed as the variance being the square root of the standard deviation.

### 2.6. Method for error detection and data quality improvement

Gross error detection and data reconciliation techniques were implemented in the free domain software ''Macrobal'' by Hellinga (1992). The technique is based on the fact that all systems can be described by a set of linear relations based on mass balances (i.e. COD, N, P and volumetric (water) flows). If more measurements are available than the minimum needed to solve the system of linear equations, the system is over determined. For over determined systems, gross errors can be detected by evaluating the balance residuals ($\neq 0$). An elaborate description of this procedure is found in Van der Heijden et al. (1994a, 1994b, 1994c). Meijer et al. (2002) used successfully Macrobal for the evaluation of the data of Katwoude WWTP.

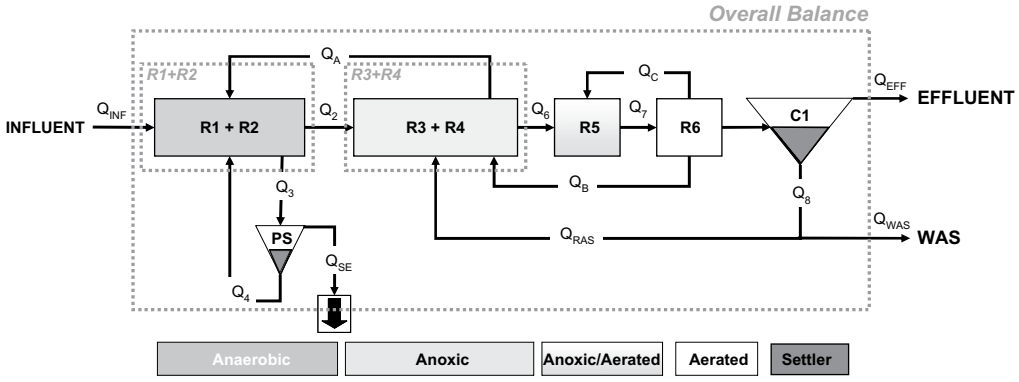


**Fig. 3 – Mass flow diagram of the full-scale WWTP.**

### 2.7. *Measurement strategy*

In this study, the aim of the measurement strategy was to provide all information needed for mass balance evaluation and data improvement (optimize information) and, at the same time, minimize the amount of measurements taken (cost reduction). Generally, the detailed layout of the plant is well known. However, in time process changes could be introduced that are not properly documented. Therefore it is important to draw up the actual process flow diagram (PFD) including all reactor compartments, overflows, pumps, settling units, valves splitters, etc. A complete PFD of the full-scale WWTP of study is presented in Fig. 1. The first step in the mass balancing procedure is to check the actual flow diagram.

To reduce the amount of measurements the total PFD is reduced to only the mass balances that are needed to obtain the required information. In our study, the two anaerobic and the two anoxic tanks were considered as combined (Fig. 3). Each system of mass balances requires a minimum amount of measurements to be solved. If fewer measurements are available, the system cannot be solved and no information is gained from the balancing procedure. If more measurements are available, the system is over determined and error detection and data improvement by reconciliation becomes possible. A choice between more or less balances or measurements has to be made based on the required accuracy and the measurements costs.

## 3. Results

### 3.1. *Case of study measurements*

The results of the measurement campaign are presented in Table 2. The overall in- and outflow measurements were done to close the overall flow balance (Fig. 3). The measurements in the anaerobic ($R_2$) and the anoxic ($R_4$) tanks were performed to check the recirculation flows ($Q_A$ and $Q_B$).

The measurements presented in Table 2 are 3-days average measurements (24-h composite samples were taken from the influent, waste sludge and the effluent and daily-grab samples were taken from the stripper effluent and the reactor tanks). High standard deviation values as well as relative errors were observed, especially in the effluent flow, indicating the high variability of the measurements. This is caused by the influent dynamics.

### 3.2. *Mass balance calculations*

All the selected mass balances (over the overall WWTP, clarifier, anaerobic and anoxic tanks) were introduced in a matrix format in the balancing software. From the matrix in Table 3, it is demonstrated how each compartment is affected by different flows and how they interact (i.e. the waste activated sludge flow ($Q_{WAS}$) affects the WWTP overall and the balances over the clarifier). From the complete matrix system of flows and measurements, the reconciliation software calculates one unique solution and at the same time, reduces the inaccuracy in the measured data.

Table 4 presents the results of the overall flow and TP balances before and after balancing. From the balances it was shown that the measured data contained errors (998 $m^3\,d^{-1}$ and 4.1 kg P $d^{-1}$). When the operational conditions are calculated from non-balanced data, errors range from 6 to 13% (Table 5), making it practically impossible to use unbalanced data for comparison and benchmarking purposes. After simultaneously solving the system of balances (Table 3) using the reconciliation software, both the water and TP balances could be fitted (Table 4). The standard deviation in the balanced data of each measurement decreased, especially for the influent and effluent (Flows: from 55–60 to 5% and TP: from 21 and 91 to 2 and 7%, respectively). The initial standard deviation was calculated on the influent dynamics. After balancing the data, the standard deviation of the influent and effluent flow measurements was brought back to the actual error of measurement (5%).

The open mass balances (COD, nitrogen and oxygen) are calculated in Table 5. Overall, the calculated conversions were approximately 10% larger after balancing the data. This increase is the direct result of the adjustment of the flow balance with approximately 5%, showing the high sensitivity of the internal conversions for the influent loading. From the calculation of the internal conversions also immediately the typical process characteristic are determined. Based on this information (e.g. the aerobic and anoxic COD, the net stoichiometric oxygen consumption) it is possible to effectively

**Table 2 – Measured data used for mass balance evaluation**

| Measurements | | Units | Influent | | Effluent | | Stripper effluent | | Overflow WAS | | Anaerobic tank 2; R2 | | Anoxic tank; R4 | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | Average ± sD | RSD (%) | Average ± sD | RSD (%) | Average ± sD | RSD (%) | Average ± sD | RSD (%) | Average ± sD | RSD (%) | Average ± sD | RSD (%) |
| Flows | Q | $m^3\ d^{-1}$ | 10140 ± 5624 | 55 | 9853 ± 5936 | 60 | 1149 ± 141 | 12 | 135 ± 20 | 15 | – | – | – | – |
| Total COD | TCOD | g COD $m^{-3}$ | 338 ± 46 | 14 | 28 ± 6 | 22 | 87 ± 12 | 13 | 10800 ± 265 | 2 | 2703 ± 146 | 5 | – | – |
| Total Kjeldahl N | TKN | g $Nm^{-3}$ | 53.03 ± 6.14 | 12 | 1.93 ± 0.80 | 41 | 26.00 ± 6.62 | 25 | 689.50 ± 370.70 | 54 | 189.67 ± 12.66 | 7 | – | – |
| Nitrate | $NO_3$ | g $Nm^{-3}$ | – | – | 5.13 ± 2.02 | 39 | – | – | – | – | 0.13 ± 0.09 | 68 | 0.22 ± 0.30 | 134 |
| Total phosphorus | TP | g $Pm^{-3}$ | 6.67 ± 2.25 | 34 | 0.44 ± 0.36 | 82 | 15.17 ± 5.30 | 35 | 369.33 ± 4.73 | 1 | 93.33 ± 8.62 | 9 | – | – |
| Ortho-phosphate | $PO_4$ | g $Pm^{-3}$ | 4.90 ± 1.21 | 25 | 0.29 ± 0.36 | 122 | 13.33 ± 5.03 | 38 | 0.29 ± 0.36 | 122 | 15.67 ± 5.13 | 33 | 3.17 ± 1.97 | 62 |

characterise the operation of a WWTP for benchmarking purposes. Using the balancing and reconciliation technique this also can be done with high accuracy, as shown from the highly reduced standard deviations of the calculated conversions.

### 3.3. SRT calculations

Based on the measured and balanced data, the SRT was calculated in four different ways: from the solids balance (Eq. (4)), from the TP balance based on TP leaving the process (Eq. (5)) or on TP entering the process (Eq. (6)) and from the COD particulate ($COD_{TSS}$) leaving the process (Eq. (7)). The results of the SRT calculations are presented in Table 6.

Meijer et al. (2002) concluded that measuring solids in RAS and WAS flows based on grab sampling can result in measurement errors up to 40%. In this case study, extra attention was given to accurately measure the flow and (solids) concentration in the WAS (and RAS) flow. Therefore, in this study, on line flow measurements and 24-h composite sampling were performed on the WAS flow. Table 6 clearly shows that with these measurement devices in place, it was possible to get a good initial estimation of the SRT based on non-balanced data (Table 5). In this particular case, no (gross) measurement errors were detected and the result of the mass balancing evaluation underlines the accuracy of the WAS measurements.

After balancing flow and TP over the WWTP, the main error in the process showed to be in the measurement of the influent flow (error of approximately 5%). The error in the influent flow is expressed in the SRT by the difference in the calculated SRT based on TP influent, before and after balancing (Table 6). Underestimation of the influent flow by approximately 5% (Table 4), results in underestimation of the calculated internal conversions of approximately 10% (Table 5). The relative sensitive relation between the calculated internal conversions and the influent flow (loading) shows that the calculated internal conversions are specific for the way the WWTP is operated and therefore, are suited for benchmarking purposes.

High standard deviations (12–22 days) were observed when the SRT was calculated based on TSS, COD particulate and on TP entering the process presented (Table 6). This case shows that the difference in the standard deviations of the TSS, COD and TP measured in the outflows is mainly the result of the accuracy of the analytical measurement. Moreover, the SRT varied from 35 to 39 days depending on the method for SRT calculation. This demonstrates how a key design parameter such as the SRT is influenced by errors. It should be noted that the chosen SRT is very sensitive in any mathematical model (Brdjanovic et al., 2000; Van Veldhuizen et al., 1999) and strongly affects the treatment process performance, sludge production and oxygen requirements (Kreuzinger et al., 2004; Metcalf and Eddy, 2003).

When the SRT was calculated using balanced data (the TP balance), the estimated error decreased dramatically. This decrease is the result of the balancing technique and the fact that a unique solution is calculated using the reconciliation software in which the error of measurements is minimised. After the mass balance evaluation, the SRT error based on the

**Table 3 – Error diagnosis and data reconciliation of the measurements.**

| Component | | Conserved moiety | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | WWTP | R1 + R2 | R3 + R4 | R5 | R6 | C1 | WWTP-TP | R1 + R2-TP | R3 + R4-TP | C1-TP | WWTP-TKN | WWTP-NO | WWTP-COD | WWTP-OC |
| | | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 | 13 | 14 |
| $Q_{INF}$ | 1 | 10140 | 10140 | | | | | 6.67 | 6.67 | | | 53.03 | | 338 | |
| $Q_{SE}$ | 2 | −1149 | −1149 | | | | | −15.17 | −15.17 | | | −26.00 | −0.13 | −87 | |
| $Q_2$ | 3 | | −17784 | 17784 | | | | | −93.33 | 93.33 | | | | | |
| $Q_{RAS}$ | 4 | | | 13604 | | | −13604 | | | 369.33 | −369.33 | | | | |
| $Q_A$ | 5 | | 914 | −914 | | | | | 189.00 | −189.00 | | | | | |
| $Q_6$ | 6 | | | −54721 | 54721 | | | | | −189.00 | | | | | |
| $Q_7$ | 7 | | | | −96524 | 96524 | | | | | | | | | |
| $Q_C$ | 8 | | | | 41803 | −41803 | | | | | | | | | |
| $Q_B$ | 9 | | | 32126 | | −32126 | | | | 189.00 | | | | | |
| $Q_8$ | 10 | | | | | −13739 | 13739 | | | | 189.00 | | | | |
| $Q_{EFF}$ | 11 | −9853 | | | | | −9853 | −0.44 | | | −0.44 | −1.93 | −5.13 | −28 | −0.20 |
| $Q_{WAS}$ | 12 | −135 | | | | | −135 | −369.33 | | | −369.33 | −689.50 | −5.13 | −10800 | −0.20 |
| $L_{NIT}$ | 13 | | | | | | | | | | | −1 | 1 | | −4.57 |
| $L_{DEN}$ | 14 | | | | | | | | | | | | −1 | −2.87 | |
| $OC_{COD}$ | 15 | | | | | | | | | | | | | −1 | −1 |
| $OC_{CNET}$ | 16 | | | | | | | | | | | | | | 1 |

Each column represents a balance. Columns 1–6 are flow balances, Columns 7–10 are P-balances, Columns 11–12 are N-balances, Column 13 is a C-balance and Column 14 is oxygen balance. C1: balance over the clarifier, WWTP: balance over the total plant. The last 4 rows are ''open'' COD and N balances ($L_{NIT}$: nitrified load, $L_{DEN}$: denitrified load, $OC_{COD}$: oxygen consumption by COD, $OC_{NET}$: net oxygen consumption). All empty spaces in the matrix represent zeros, elements with negative signs are outflows, and positive elements are inflows.

**Table 4 – Closed mass balance calculations; flow and total phosphorus**

| | Units | Measured data | | Balanced data | | |
|---|---|---|---|---|---|---|
| | | Average ± sD | RSD (%) | Average ± sD | RSD (%) | Estimated error (%) |
| Flow balance | | | | | | |
| Influent flow | $m^3 d^{-1}$ | 10140 ± 5624 | 55.5 | 10591 ± 508 | 4.8 | 4.3 |
| Effluent flow | $m^3 d^{-1}$ | −9853 ± 5936 | 60.2 | −9370 ± 449 | 4.8 | −5.2 |
| Stripper effluent | $m^3 d^{-1}$ | −1149 ± 141 | 12.3 | −1085 ± 52 | 4.8 | −5.9 |
| Waste activated sludge | $m^3 d^{-1}$ | −135 ± 20 | 14.8 | −136 ± 6 | 4.8 | 0.2 |
| Error in measurements | $m^3 d^{-1}$ | −998 | | 0 | | |
| TP balance | | | | | | |
| Influent flow | $kg P d^{-1}$ | 67.6 ± 14.5 | 21.5 | 70.6 ± 1.3 | 1.9 | 4.3 |
| Effluent flow | $kg P d^{-1}$ | −4.3 ± 3.9 | 91.2 | −4.1 ± 0.3 | 7.3 | −5.2 |
| Stripper effluent | $kg P d^{-1}$ | −17.4 ± 0.6 | 3.3 | −16.5 ± 0.2 | 1.2 | −5.9 |
| Waste activated sludge | $kg P d^{-1}$ | −49.9 ± 0.4 | 0.8 | −50.1 ± 0.1 | 0.2 | 0.2 |
| Error in measurements | $kg P d^{-1}$ | −4.1 | | 0.0 | | |

Elements with negative signs are outflows, and positive elements are inflows.

MLSS and $COD_{TSS}$ measurements remained the same. However, the SRT errors from both TP balances were brought in accordance and could be reduced (Table 6). This is due to the fact that using the P balance, more information can be used to improve the data quality in the reconciliation procedure. With the COD balance, the non-measured oxygen consumption makes reconciliation of the balance impossible. It was therefore impossible to detect the measurement error in the measured $COD_{TSS}$ concentration. After balancing the difference in the measurement accuracy between TSS and TP is expressed in the standard deviation of the calculated SRT values.

## 4. Discussion

### 4.1. *Mass balance evaluation and error detection of full-scale WWTPs*

For the evaluation of a full-scale WWTP a correct design of the measurement strategy is crucial. In a static evaluation it is important to capture the average sludge composition over one mixing period. This requires an evaluation over approximately 3 times the SRT. Also the aim of the study should be defined and accordingly, the type of measurements (e.g. TP, TSS or Fe for balancing purposes). The choices should be related to the measurement costs and to the required accuracy of the study. In this case, influent WWTP flow data from 3-month average measurements including rain events were taken. This causes relative high standard deviations in the flow data (55%; Table 4). The 3-month period was chosen based on the average SRT which was 36 days (Table 6). For each measurement point where 24-h composite samples were not available, 3 grab samples were taken.

Once the measurement campaign was done, a manual and rigorous check was performed to identify gross errors (such as wrong labels in the database, wrong samples (method or place), and wrong electronic connections in automation system). The erroneous data could be detected after manually checking a conserved balance (i.e. TP balance). After the gross error detection, in this study the TP balance based on measured data presented an estimated error of 4.1 $kg d^{-1}$ (6% of the influent TP; Table 4) as well as the flow balance of 998 $m^3 d^{-1}$ (10% of the influent flow). These (statistical) errors on measured data often happen in practice, especially in

**Table 5 – Open mass balance calculations; COD, nitrogen and oxygen balance**

| | Units | Measured data | | Balanced data | | Estimated error (%) |
|---|---|---|---|---|---|---|
| | | Average ± sD | RSD (%) | Average ± sD | RSD (%) | |
| COD balance | | | | | | |
| COD denitrification | $kg COD d^{-1}$ | −990 ± 75 | 7.5 | −1073 ± 7 | 0.6 | 7.7 |
| COD oxidation | $kg COD d^{-1}$ | −601 ± 132 | 21.9 | −687 ± 12 | 1.7 | 12.5 |
| Nitrogen balance | | | | | | |
| Nitrification | $kg N d^{-1}$ | −396 ± 42 | 10.6 | −422 ± 4 | 0.9 | 6.2 |
| Denitrification | $kg N d^{-1}$ | −345 ± 44 | 12.8 | −374 ± 4 | 1.0 | 7.7 |
| Oxygen balance | | | | | | |
| Total oxygen consumption | $kg O_2 d^{-1}$ | 2409 ± 159 | 6.6 | 2615 ± 14 | 0.5 | 7.9 |
| Nitrogen oxidation | $kg O_2 d^{-1}$ | 1808 ± 90 | 5.0 | 1928 ± 8 | 0.4 | 6.2 |
| COD oxidation | $kg O_2 d^{-1}$ | 601 ± 132 | 21.9 | 687 ± 12 | 1.7 | 12.5 |

Elements with negative signs are outflows, and positive elements are inflows.

**Table 6 – Calculation of the solids retention time**

| SRT calculation | Units | Measured data | | Balanced data | | |
|---|---|---|---|---|---|---|
| | | Average ± sD | RSD (%) | Average ± sD | RSD (%) | Estimated error (%) |
| SRT based on MLSS | days | 36.3 ± 22.4 | 1380 | 36.2 ± 22.2 | 1365 | 0.1 |
| SRT based on CODTSS | days | 34.6 ± 12.1 | 421 | 34.6 ± 11.9 | 406 | −0.1 |
| SRT based on TPTSS outputs | days | 36.4 ± 2.4 | 16 | 36.4 ± 1.5 | 6 | −0.1 |
| SRT based on TPTSS influent | days | 39.4 ± 13.4 | 455 | 36.4 ± 1.9 | 10 | 8.1 |

full-scale WWTPs. Neglecting both gross, as well as, statistical errors leads to erroneous calculations of operational conditions, incorrect process designs and errors in model parameter calibrations (Meijer et al., 2002; Van Veldhuizen et al., 1999).

After the first manual check, all balances (close and open balances) were included into the matrix (Fig. 2). A unique solution could be calculated without further indication of large errors. As shown in Table 4, the flow and TP balances (close balances) fitted perfectly using the balanced data. This result was achieved by effectively fitting the balances by tuning only the flow values (large standard deviations) and considering the measurement concentration correct (small standard deviations).

The results from the open mass balances calculations (COD and nitrogen balances) are shown in Table 5. In the nitrogen balance, the nitrified load was 422 kg N $d^{-1}$ and the denitrified load was 374 kg N $d^{-1}$ (75% and 67% of the incoming nitrogen of 561.7 kg N $d^{-1}$, respectively). For all internal process rates (kg converted per day) the relative standard deviations of the balanced data (between 0.4 and 1.9%) were smaller than the error of the measured data (between 5.0 and 21.9%). This indicates there is a systematic error in the internal conversion rates which causes the actual conversion rates to be higher than originally measured. The systematic error could be related to underestimation of the influent flow with approximately 5% (Table 4).

The total oxygen consumption (OC) was also determined using mass balances (Table 5). A proper evaluation of the OC is advised because the OC is directly related to the aeration input and the operational costs of the WWTP. The OC conversion increased with 9% after balancing to 2615 kg $O_2$ $d^{-1}$. 74% of this oxygen amount was used for nitrogen removal. The ratio of oxygen consumed over nitrogen removed was 3.4 kg $O_2$ kg $N^{-1}$ versus 0.2 kg $O_2$ kg $COD^{-1}$. These amounts of oxygen were required to concert each kg of nitrogen and organic matter in the influent.

### 4.2. *Influence of measurements errors on process parameters*

Errors in the measurements affected important process parameters such as the SRT and internal conversion rates. We compared the SRT calculations using measured and balanced data in the full-scale WWTP (Fig. 4). The traditional SRT calculations based on TSS resulted in high standard deviations (Table 6). In spite of reducing the error in the flow balance, no improvement could be achieved in the accuracy of the SRT based on the TSS measurements. This result follows Ekama et al. (2007) who observed that solids are not mass conservative and therefore, provide no assessment of mass balances.

The SRT calculated from COD measured in the WAS only slightly improved by balancing (Table 6). This is related to the fact that COD is an open balance in which OC is not measured and therefore, has to be calculated. Thus, the best SRT results were obtained by using the phosphate balances (Table 6) showing that data reconciliation is an effective technique.

Finally, the fact that all four calculation methods lead to a comparable SRT indicates that this is a correct value.

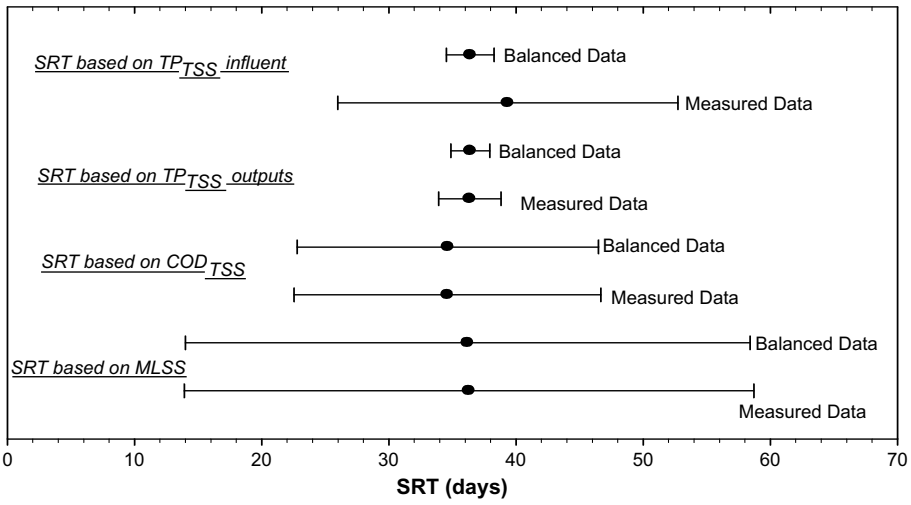


**Fig. 4 – SRT calculation based on MLSS, $COD_{TSS}$, $TP_{TSS}$ influent and outputs using measured or balanced data in the full-scale WWTP.**

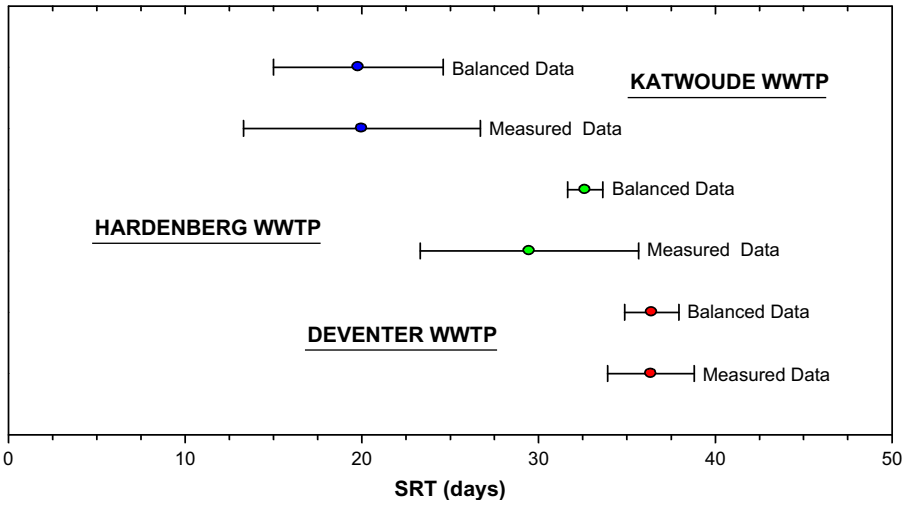


**Fig. 5 – Decision making in full-scale WWTPs based on SRT calculations based on P balances.**

### 4.3. *Benchmarking between full-scale WWTPs*

When different WWTPs are compared on their efficiency, the calculated SRT is an important operational criteria. Fig. 5 presents the different SRTs of three WWTPs (before and after balancing): Katwoude WWTP (Meijer et al., 2002), Hardenberg WWTP (Meijer et al., 2001) and Deventer WWTP (this study).

If we compare the obtained values in this study with that of other studies using measured data, we see that the difference between the WWTPs is not really pronounced. After balancing, the standard deviation in the SRT decreases (Table 6). By reducing the error in the SRT calculations, the WWTPs become significantly different from each other. In this way, also other operational conditions like nitrification, denitrification or OC can be compared more effectively. This example shows that by using mass balancing techniques and data reconciliation, better and more reliable WWTPs benchmarks can be developed.

## 5. Conclusions

In this paper it has been demonstrated that poor quality of WWTP information leads to large errors in key process parameters such as SRT and conversion rates. A practical stepwise methodology was presented in which mass balances and data reconciliation techniques were applied on full-scale WWTP data. It has been shown that by improving the data quality using mass balance calculations, useful new information becomes available for process evaluation, WWTP design and benchmarking.

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## REFERENCES

Adgate, J., Willis, R., Buckley, T.J., Chow, J.C., Watson, J.G., Rhoads, G.G., Lioy, P.J., 1998. Chemical mass balance source apportionment of lead in house dust. Env. Sci. Tech. 32 (1), 108–114.

Baker, J.I., Hites, R.A., 2000. Is combustion the major source of polychlorinated dibenzo-p-dioxins and dibenzofurans to the environment? A mass balance investigation. Env. Sci. Tech. 34 (14), 2879–2886.

Benedetti, L., Bixio, D., Vanrolleghem, P.A., 2006. Benchmarking of WWTP design by assessing costs, effluent quality and process variability. Water Sci. Technol. 54 (10), 95–102.

Bixio, D., Parmentier, G., Rousseau, D., Verdonck, F., Meirlaen, J., Vanrolleghem, P.A., Thoeye, C., 2002. A quantitative risk analysis tool for design/simulation of wastewater treatment plants. Water Sci. Technol. 46 (4–5), 301–307.

Brdjanovic, D., van Loosdrecht, M.C.M., Versteeg, P., Hooijmans, C.M., Alaerts, G.J., Heijnen, J.J., 2000. Modelling COD, N and P removal in a full-scale wwtp Haarlem Waarderpolder. Water Res. 34 (3), 846–858.

Ekama, G.A., Wentzel, M.C., Sötemann, S.W., 2006. Mass balance-based plant-wide wastewater treatment plant models – part 2: tracking the influent inorganic suspended solids. Water SA 32 (3), 277–285.

Ekama, G.A., Sötemann, S.W., Wentzel, M.C., 2007. Biodegradability of activated sludge organics under anaerobic conditions. Water Res. 41 (1), 244–255.

Hellinga, C., 1992. Macrobal 2.02. Delft University of Technology, The Netherlands. Download: <http://www.tnw.tudelft.nl/live/pagina.jsp?id = 8e805079-dcd5-40f8-9ef0-f00ad1fb09f8&lang = en&binary = /doc/Mbal202.exe>.

Kreuzinger, N., Clara, M., Strenn, B., Kroiss, H., 2004. Relevance of the sludge retention time (SRT) as design criteria for wastewater treatment plants for the removal of endocrine disruptors and pharmaceuticals from wastewater. Water Sci. Technol. 50 (5), 149–156.

Lindtner, S., Kroiss, H., Nowak, O., 2004. Benchmarking of municipal waste water treatment plants (an Austrian project). Water Sci. Technol. 50 (7), 265–271.

Meijer, S.C.F., van Loosdrecht, M.C.M., Heijnen, J.J., 2001. Metabolic modelling of full-scale biological nitrogen and phosphorus removing WWTP's. Water Res. 35 (11), 2711–2723.

Meijer, S.C.F., van der Spoel, H., Susanti, S., Heijnen, J.J., van Loosdrecht, M.C.M., 2002. Error diagnostics and data reconciliation for activated sludge modelling using mass balances. Water Sci. Technol. 45 (6), 145–156.

Metcalf, Eddy, 2003. Wastewater engineering: treatment and reuse, fourth ed. McGraw-Hill Higher Education, New York, ISBN 0-07-112250-8.

Narasimhan, S., Jordache, C., 2000. Data Reconciliation and Gross Error Detection: an Intelligent Use of Process Data. Gulft Publishing Company, Houston, ISBN 0-88415-255-3.

Nyserda, 2008. Focus on water and wastewater. http://www.nyserda.org/Programs/Environment/muniwaterwwt.asp.

Nowak, O., Franz, A., Svardal, K., Müller, V., Kühn, V., 1999. Parameter estimation for activated sludge models with the help of mass balances. Water Sci. Technol. 39 (4), 113–120.

Olsson, G., 2006. Instrumentation, control and automation in the water industry – state-of-the-art and new challenges. Water Sci. Technol. 53 (4–5), 1–16.

Unie van Waterschappen, 2003. Bedrijfsvergelijking Zuiveringsbeheer 2002. http://www.samenwerkenaanwater.nl/content/benchmarking/bedrijfsvergelijking_zuiveringsbeheer_eindrapport.pdf.

Van Loosdrecht, M.C.M., Brandse, F.A., de Vries, A.C., 1998. Upgrading of wastewater treatment processes for integrated nutrient removal – the BCFSs process. Water Sci. Technol. 37 (9), 209–217.

Van Veldhuizen, H.M., van Loosdrecht, M.C.M., Heijnen, J.J., 1999. Modelling biological phosphorus and nitrogen removal in a full-scale activated sludge process. Water Res. 33 (16), 3459–3468.

Van der Heijden, R.T.J.M., Heijnen, J.J., Hellinga, C., Romein, B., Luyben, K.Ch.A.M., 1994a. Linear constraint relations in biochemical reaction systems: I. Classification of the calculability and the balance ability of conversion rates. Biotechnol. Bioeng. 43 (1), 3–10.

Van der Heijden, R.T.J.M., Romein, B., Heijnen, J.J., Hellinga, C., Luyben, K.Ch.A.M., 1994b. Linear constraint relations in biochemical reaction systems: II. Diagnosis and estimation of gross errors. Biotechnol. Bioeng. 43 (1), 11–20.

Van der Heijden, R.T.J.M., Heijnen, J.J., Hellinga, C., Romein, B., Luyben, K.Ch.A.M., 1994c. Linear constraint relations in biochemical reaction systems: III. Sequential application of data reconciliation for sensitive detection of systematic errors. Biotechnol. Bioeng. 44 (7), 781–791.