WASTEWATER TREATMENT PLANT MODELLING FOR CAPACITY ESTIMATION AND RISK ASSESSMENT

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WATER RESEARCH COMMISSION

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Report to the

WATER RESEARCH COMMISSION

by

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EXECUTIVE SUMMARY

Introduction

The Green Drop regulation programme seeks to identify and develop the core competencies required for the sector that, if strengthened, will gradually and sustainably improve the level of wastewater management in South Africa (Department of Water and Sanitation, 2013). It is a form of regulation that gives the focus, commitment, planning and resources needed to achieve excellence in wastewater management.

Simulation of wastewater treatment processes is a rapidly developing technology, increasingly used to achieve better designs of new plants and improved operation of existing ones.

The use of modelling offers the possibility of supplementing the Green Drop programme's current relativistic approach to assessing the performance of wastewater treatment works, by providing performance criteria which have a quantitative basis. Furthermore, a process model reflects a deeper understanding of the process than is needed for routine operation and compliance monitoring, and the discipline of constructing a model almost always reveals aspects of the process which were not previously understood.

The measurements required to support accurate modelling are more demanding than those for routine operation or compliance monitoring, so additional infrastructure must be developed to use modelling effectively. However, once a model has been developed, it provides a convenient and inexpensive platform for exploring different operating strategies or design improvements, which should greatly facilitate the search for continuous improvements.

Modelling tools

Both steady state and dynamic wastewater treatment models were evaluated. Two steady state models were investigated. The first was developed in WEST at the University of KwaZulu-Natal, the second in Excel at the University of Cape Town. The latter was William Wu's MSc project, and the uncertainty about when it would become available for use was part of the motivation for developing the former. When the UCT Excel model became available it proved to be better suited to the Green Drop objectives than the WEST version, and was used in a case study based on the Umhlanga Wastewater Treatment Plant (WWTP) in eThekwini.

The Excel based model, also known as the *Plant-wide Steady State Design* programme or PWSSD is a flexible and freely available programme which was created primarily to be a design tool. However, a capacity estimation feature was added specifically for the purposes of the Green Drop project. The capacity estimation tool can simultaneously estimate the capacities of all the major unit processes, including the capacity for biological nutrient removal, as a function of the raw wastewater characteristics and operating conditions. The capacity estimates can be compared to the measured plant performance, as required by Green Drop, and the results used to assess whether poor performance is due to design limitations or operating and maintenance issues.

Dynamic modelling was evaluated using the WEST dynamic simulator. The UCT/UKZN three phase (aqueous-gas-solid) plant-wide dynamic model (PWM_SA) was developed for simulation of nutrient removal activated sludge systems coupled with the anaerobic, aerobic and anoxic-aerobic digestion of sewage sludge including waste activated sludge (WAS) produced by Biological Excess Phosphorus Removal (BEPR) plants. PWM_SA has been incorporated into WEST versions 2014 and higher. The use of the PWM_SA model in WEST was evaluated in two case studies: Phoenix WWTP and Darvill WWTP.

A critical component of any modelling investigation, whether steady state or dynamic, is the characterisation of the wastewater being treated. Routine plant monitoring data are generally insufficient to characterise the influent wastewater for modelling purposes. However, a measurement campaign of the required scope would be far too expensive and time consuming to be considered for a Green Drop plant audit. To overcome this problem, a *probabilistic fractionator* was developed. This combines available plant data with assumptions based on literature or local experience to estimate the wastewater composition.

Case studies

Case studies were conducted on eThekwini's Umhlanga and Phoenix WWTPs, and Umgeni Water's Darvill WWTP.

The Umhlanga WWTP is a small extended aeration plant located in the Northern Coastal area of the eThekwini Municipality. The Umhlanga plant was inherited by eThekwini without its original design documents therefore there is some uncertainty about it is original design capacity. The case study focused on capacity estimation using the UCT steady-state model.

The Phoenix Wastewater Treatment Works (PWWTW) in eThekwini Municipality treats approximately 23 ML/day of raw sewage from the Phoenix and Ottowa areas in the north of the municipality. Its design is 25 ML/day and it is a conventional activated sludge plant capable of nitrification and COD removal. With development in the surrounding areas, this plant is currently being upgraded to a capacity of 50 ML/day. In addition new licence requirements from the Department of Water and Sanitation require tighter nutrient standards for nitrogen and phosphorus. Thus, the existing plant will have to be retrofitted for Biological Nutrient Removal (BNR). This case study focused on dynamic modelling.

The Darvill WWTP presented a much more complex problem because it is subject to more complex and variable loading from industrial contributions to the wastewater it received. It was also significantly overloaded, and is consequently undergoing a major upgrade. It was hoped that it would provide a good example for testing the Green Drop concept, i.e. how the dynamic model would identify overloading.

The Umhlanga WWTP

In this case study, PWSSD, was used to estimate the capacity of the Umhlanga WWTP, using available monitoring and operational data, in order to assess the potential use of the capacity estimation tool for the purposes of the Green Drop program.

The model is easy to set up, the data requirements are much less than that required for dynamic modelling and most of it should be available from the annual process audits WSI's are already required to undertake as part of the Green Drop programme. However, field trials are required to assess how well this will work in practice.

The Umhlanga case study was carried out almost exclusively with data that had already been captured electronically, which proved insufficient to resolve a number of questions, especially regarding the sludge age, dissolved oxygen profiles and aeration capacity. It is probable that many of the data gaps could have been addressed during onsite inspections and interviews with the operators during a process audit, however, this needs to be tested in full scale field trials.

The process of trying to gather data required by the steady state model is likely to be a useful exercise in itself, because it would draw attention to deficiencies in plant operating procedures and monitoring, and also potentially help operators and plant managers understand the operation of their plant better. However, there is no guarantee it would lead to accurate and reliable capacity estimates. The reliability of the capacity estimates generated by PWSSD is strongly dependent on the quality of the inputs as well as the extent to which the model assumptions deviate from the actual operating conditions.

There are many reasons why the data available to an audit team might be inadequate, including:

- Missing or incomplete operating and monitoring records;

- Uncalibrated instruments;
- Poor sampling procedures and inappropriate sample handling;
- Missing documentation;
- High staff turnover, with the result that operators cannot answer questions about the plant's history and typical operation;
- Spot measurements taken during onsite inspections may not be representative of typical operation;
- The plant configuration and operating procedures may not conform to the model assumptions.

Field trials are therefore required to assess:

1) Barriers to obtaining the necessary input data;
2) The amount of additional time, effort and resources required to obtain useful results;
3) Whether the enhanced capacity estimation capabilities of PWSSD actual do lead to improved plant operation and long term capacity planning.

The Phoenix WWTP

The dynamic model was implemented on the WEST modelling platform. The model development and calibration proved reasonably successful using the established IWA ASM2 model, which, however, was only able to represent the aerobic section of the plant. A comparison with a PWM_SA version also showed satisfactory agreement, which provides a measure of confidence in the new model. However, the anaerobic sludge digesters were not included in the model, as the investigation focused on the influent characterisation.

In spite of the reasonable success achieved by the modelling as such, the complexity of the procedures and the judgement required to get the model to a point where it gave a good representation of the plant performance suggested that it would be impractical for the purpose of a Green Drop audit. Where the model had been developed for other purposes, and confidence in its predictions had already been established, it could be used to good effect to investigate risk scenarios, such as the failure of an item of equipment.

To illustrate the use of the model in a risk assessment, the impact on plant operation of the sludge de-watering system being out of operation for two weeks was simulated. The model predicted that the plant would survive the failure without exceeding effluent discharge limits. Assessments of this kind could be used to establish which spare parts need to be kept on site, and which can be procured at the time that a failure takes place.

The Darvill WWTP

The work undertaken on the Darvill model highlighted a number of challenges relating to modelling plants operating above capacity, receiving industrial effluent as well as general problems relating to data availability:

Challenges related to data availability

In the Phoenix case study, the influent data was available in form of composite samples which are more representative of the average daily loads, while only daily grab samples were available for Darvill. It was hoped that the equalization tank at Darvill would provide sufficient damping of the normal daily variations in settled sewage data used in the calibration of the activated sludge system model, however, there was still a lot of scatter in the measurements and it was not clear if this was due to grab sample timing, measurement errors or actual variations in the daily loads. There were similar concerns about the dissolved oxygen measurements which were also single daily measurements. More work is required on resolving the issues related to diurnal variations if the proposed modelling tools are to be widely applied.

Challenges related to overloaded plants

- When a plant is overloaded with respect to its aeration capacity, it is expected that the dissolved oxygen levels in the activated sludge basins will vary with the influent loads. However, it turns out to be quite difficult to model this situations. The profile is very sensitive to the assumed oxygen transfer coefficients (Kla's), which probably also vary with changes in operating conditions and concentrations. Biological processes such as nitrification are strongly dependent on the dissolved oxygen levels therefore it is essential to get this aspect of the modelling right. Future work should focus on a more robust approach to modelling this type of situation.
- The Darvill WWTP was no longer operating as it was originally designed or as described in the operating manual. For example, most of the settled sewage bypassed the anaerobic selector, aerators have been installed in the anoxic zone and phosphorous removal is being achieved using chemical precipitation. This made it harder to make reasonable assumptions about the plant operation and performance when data was missing.
- Processes with highly variable performance are inherently more difficult to model than processes that are relatively stable. Extreme situations are more likely occur which may fall outside the range of validity of the various models and typical modelling assumptions.

Challenges relating to the industrial content of the influent

The fractionator tool defaults (set of components, stoichiometry and typical fractions) have been set up based on wastewater characterisation data for typical domestic wastewaters and may not handle wastewaters with a significant industrial component as well. It may therefore be necessary to adjust the PWM_SA components' stoichiometry or possibly even define new industry specific components with their own stoichiometries and degradation kinetics for some plants receiving industrial effluent. This would of course require a substantial modelling effort as well as additional measurements and would not be feasible as part of an audit.

Assessing the potential role of modelling in Green Drop

The overall goal of this project was to investigate and assess the potential for using wastewater treatment process models to support the goals of the Green Drop programme. The two aspects of a Green Drop plant audit that were considered were steady state modelling for estimating the treatment capacity of the WWTP, and dynamic modelling for the analysis of risk scenarios. As discussed above, technical objectives of the case studies were largely successfully met for the Umhlanga and Phoenix case studies, but not for the Darvill case study. However these technical objectives were not the main purpose of the investigations, which was to use the experience of undertaking them to assess whether it would be practical and advantageous to use such investigations as part of WWTP audits in the Green Drop programme. This assessment could not be made by the project team alone, because it involves issues relating to the ability of municipalities to provide the necessary scientific, technical and personnel resources. For this aspect we relied on the input of industry representatives in the two workshops and on the reference group.

In the first workshop a group of engineers from eThekwini Water and Sanitation and Umgeni Water were taken through a modelling exercise over a period of several months. Several of these had been involved in Green Drop audits. The second workshop involved a wider audience, including a Green Drop inspector and engineers from both the large metros and small municipalities and was specifically aimed at eliciting their responses to the case studies. Feedback from the workshops and reference group largely shaped the overall project conclusions and recommendations.

Conclusions and Recommendations

1. Modelling requires a considerable investment in time and skill. In South Africa, only the large metros will be able to develop modelling capacity in the foreseeable future.
2. Steady state modelling is much less demanding than dynamic modelling, and should be introduced first, as part of a programme to develop modelling capacity in the industry.
3. Using dynamic simulators to assess various risk scenarios and provide quantitative plant specific data for Wastewater Risk Abatement Plans is seen as a very promising application of modelling. However, the effort and expertise required to set up and calibrate the dynamic models as well as the high cost of the dynamic simulators means that most municipalities are unlikely to be able to pursue this option in the near future. Other risk assessment tools are already available, and simulation would provide a supplement, not a replacement, for these.
4. The steady state capacity estimation model will be particular useful to avoid Green Drop penalties for older plants where proper design reports are not available. It can also be used to predict how plants will respond to increases in population or changes in wastewater characteristics.
5. Although a WTTP model can produce results which are useful for a Green Drop audit, setting up a model is too time consuming to be considered as part of the audit itself. Modelling would have to be completed beforehand as part of the preparatory work. The capacity estimation exercise could potentially be incorporated into the annual plant audits municipalities are supposed to carry our prior to the Green Drop audit. Both steady state and dynamic models can be useful for a range of other purposes besides providing data for the audit.
6. It will not be practical to introduce modelling as a Green Drop requirement in the near future. Rather, the Green Drop programme could adopt a strategy to progressively encourage the development of modelling capacity in order to enhance wastewater treatment practice. This could start with bonus points for the effective use of modelling to meet existing Green Drop goals better, such as capacity estimation or risk assessment. Bonus points can also be awarded for partnerships between larger and smaller municipalities which promote the use of modelling.
7. A specialist division of the Water Institute of Southern Africa should be established to promote and develop WWTP modelling practice.
8. While case-studies presented in this report provided valuable information on the capabilities and limitations of the modelling tools developed, they were essentially desk studies, and there is still a need for a pilot project or field trials to test the usefulness and practicality of the tools in the context of an actual audit.

Three new tools were developed, which are especially aimed at the issues that were expected to be encountered during a Green Drop audit. These are:

1. A steady state model which is suitable for estimating the capacity of a WWTP. This is particularly useful for older WWTPs where the original design documents have been lost, or where the characteristics of the wastewater are markedly different from those used for the design. It can also be used for long-term capacity planning.
2. A probabilistic wastewater fractionator for estimating the characteristics of the influent wastewater from routine plant measurements. This minimises the requirement for extensive additional measurements that would normally be required to characterise the wastewater for modelling purposes.

3. The PWM_SA (Plant Wide Model – South Africa) was introduced as a tool for dynamic modelling using the WEST software platform. This an outcome on of a many years of research carried out at the University of Cape Town and the University of KwaZulu-Natal.

Specific recommendations for the further development of the modelling tools include the following:

a) Developing guidelines on the minimum data requirements for modelling various types of plants with different treatment objectives.

b) Developing user friendly guidelines for data checking and reconciliation.

c) Generating more examples of typical raw wastewater profiles for different parts of the country and for plants serving different types of communities. This type of data could be collected as part of field trials conducted in different parts of the country.

d) Developing a more user friendly interface for the probabilistic fractionator.

e) Developing improved methods for characterising wastewaters with a significant industrial contribution.

CAPACITY BUILDING AND TECHNOLOGY TRANSFER

STUDENTS

Akash Singh, MScEng: Modelling of Wastewater Treatment Plants Receiving Industrial Effluents, registered 2014

WORKSHOPS

Training workshop

A dynamic wastewater modelling course consisting of six training sessions were conducted for personnel from eThekwini Water and Sanitation and Umgeni Water between May and August 2014. The theme of the workshops was to develop, calibrate use a model of a wastewater treatment plant, with the aim was to have a documented working model of an actual plant by the end of the course. A section of eThekwini’s Northern WWTP was chosen as the case study, and 4 of the 6 sessions were conducted at the Training Centre located at the plant. The sessions took place at approximately 2 week intervals, to allow the participants to obtain data or resolve issues identified during the sessions. A detailed report on the training programme is presented in Appendix B. The dynamic modelling course was followed up by a presentation on steady-state modelling to a number of the original participants in November 2015.

Dissemination workshop

A workshop was held at the University of KwaZulu-Natal on the 24th July 2015 to engage with practitioners with experience in conducting Green Drop audits to assess whether the idea of involving process modelling in Green Drop audits was worth pursuing to the next stage, which might involve a follow up WRC project coinciding with the next round of Green Drop evaluations (expected to take place in 2016).

The workshop consisted of presentations on the modelling tools, discussions on their potential application in Green Drop and a questionnaire to elicit quantitative feedback. The general feeling was the modelling tools would be useful to those larger municipalities who have the capacity to use them but lack of capacity would be a major barrier to their implementation especially for the smaller rural municipalities. Consequently modelling cannot be a requirement of Green Drop but it may possibly be used to earn bonus points, however, further trials would be required to demonstrate its usefulness.

Participants tended to be most excited about the potential for using dynamic simulation to improve risk assessment. A detailed report on the workshop and its outcomes is presented in Appendix C.

PUBLICATIONS AND PRESENTATIONS

Conference presentations

DS Ikumi, PA Vanrolleghem, CJ Brouckaert, MB Neumann and GA Ekama (2014). Towards calibration of phosphorus (P) removal Plant-Wide models. 4^{th} IWA/WEF Wastewater Treatment Modelling Seminar 2014 (WWTmod 2014) 30 March - 2 April 2-14, Spa, Belgium. (Poster)

I Lizarralde, CJ Brouckaert, PA Vanrolleghem, DS Ikumi, GA Ekama, E Ayesa and P Grau (2014). Incorporating aquatic chemistry into wastewater treatment process models: a critical review of different approaches. 4^{th} IWA/WEF Wastewater Treatment Modelling Seminar 2014 (WWTmod 2014) 30 March - 2 April 2-14, Spa, Belgium. (Poster)

A Singh (2015), Making do with what you have: a practical guide to reconciling existing plant data for wastewater for model calibration. 4^{th} YWP-ZA and 1^{st} African YWP Conference: Pretoria,16-18 November 2015 (Poster)

CJ Brouckaert, BM Brouckaert and A Singh (2016) A Probabilistic Influent Fractionator, 5^{th} IWA/WEF Wastewater Treatment Seminar: Annecy, France 2-6 April 2016 (Submitted)

CJ Brouckaert, BM Brouckaert and E Musvoto (2016). Validation of a Probabilistic Influent Fractionator, 5^{th} IWA/WEF Wastewater Treatment Seminar: Annecy, France 2-6 April 2016 (Submitted)

BM Brouckaert, CJ Brouckaert, A Singh, W Wu (2016) Steady State Modelling for Capacity Estimation, WEFTEC 2016, New Orleans, LA, USA,24 - 28 September, 2016 (Submitted)

BM Brouckaert, CJ Brouckaert, E Musvoto, W Wu, GA Ekama (2016) The Impact of Wastewater Characterization Data Quality on Wastewater Treatment Plant Capacity Estimates, WEFTEC 2016, New Orleans, LA, USA,24 - 28 September, 2016 (Submitted)

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Dr DS Ikumi	University of Cape Town
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Dr L Maharaj	Umgeni Water
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1.1 Motivation

The Green Drop regulation programme seeks to identify and develop the core competencies required for the sector that, if strengthened, will gradually and sustainably improve the level of wastewater management in South Africa (Department of Water and Sanitation, 2013). It is a form of regulation that gives the focus, commitment, planning and resources needed to achieve excellence in wastewater management.

The Green Water Services Audit is the tool whereby incentive- and risk-based regulation is conducted in South Africa. The Green Drop process measures and compares the results of the performance of Water Service Institutions, and subsequently rewards (or penalises) the institution upon evidence of their excellence (or failures) according to the minimum standards or requirements that has been defined.

While the Green Drop assessment considers the entire municipal wastewater service, its *Cumulative Risk Assessment* component focuses on the wastewater treatment function. Treatment is one of the high risk components within the wastewater value chain. Risk-based regulation allows the municipality to identify and prioritise the critical risk areas within its wastewater treatment process and to take corrective measures to abate these.

The conventional way of assessing wastewater treatment process performance uses a combination of general discharge standards and historical records for the particular plant, according to a 'continuous improvement' principle.

Simulation of wastewater treatment processes is a rapidly developing technology, increasingly used to achieve better designs of new plants and improved operation of existing ones.

The use of modelling offers the possibility of supplementing the Green Drop programme's current relativistic approach to assessing the performance of wastewater treatment works, by providing performance criteria which have a quantitative basis. Furthermore, a process model reflects a deeper understanding of the process than is needed for routine operation and compliance monitoring, and the discipline of constructing a model almost always reveals aspects of the process which were not previously understood. Both eThekwini Water and Sanitation (EWS) and Umgeni Water experience considerable difficulties at some of their WWTPs arising from the presence of significant loads of industrial effluents, and there are frequently questions as to whether poor treated water quality is due to industrial components in the wastewater or to deficiencies in the treatment processes.

The measurements required to support accurate modelling are more demanding than those for routine operation or compliance monitoring, so additional infrastructure must be developed to use modelling effectively. However, once a model has been developed, it provides a convenient and inexpensive platform for exploring different operating strategies or design improvements, which should greatly facilitate the search for continuous improvements.

A series of previous WRC projects had led to the development of steady-state and dynamic WWTP models at UCT and more recently with the collaboration of UKZN. However, previous to this project, these models have been based on data from laboratory studies, and had not yet been applied to full scale plants.

WWTP modelling had not been put into practice to any significant extent by South African municipalities, although consultants had been using it to some extent. EWS and Umgeni Water had previously decided that it was necessary to develop modelling expertise to enhance their competence in WWTP design, control and optimisation. Both organisations had purchased the WEST modelling software. However, they found it difficult to allocate the necessary skilled manpower resources and analytic facilities, particularly during the learning curve before the investment of time began to enhance the effectiveness of the engineers concerned. Thus, although several models had already been developed, they had not found practical application.

1.2 Overall course of the project

- The UCT/UKZN plant-wide model was incorporated into the 2014 version of the WEST wastewater treatment modelling software as PWM_SA (Plant Wide Model – South Africa). This dynamic model was the basis for the dynamic modelling undertaken during the project
- A steady-state model based on ASM1 and implemented in WEST was developed for monitoring small, basic WWTPs, since eThekwini has a number of these However, it was subsequently superseded by an Excel based steady state model received from the Water Research Group at UCT. This was applied to the Umhlanga WWTP.
- A workshop comprising a series of six training sessions was conducted between May and August 2014. Twelve members of staff from eThekwini Water and Sanitation participated, and 2 from Umgeni Water. The outcome of the workshop was a working and partially calibrated model of a section of the Northern WWTP.
- A number of issues in the WEST plant wide mode PWM_SA were fixed, and will be distributed in the 2016 version of WEST.
- The plant-wide model was applied successfully to the Phoenix WWTP.
- Some progress was made with modelling of Umgeni Water's Darvill plant, which has the most complex configuration of those considered during the project. However the situation at Darvill proved to be very complex, and it was concluded that it was not suitable as a case study for the project, as the time and effort required to develop a model would not be viable for a Green Drop audit.
- A workshop was held on the 24th July 2015 to present the project outcomes to a range of people involved in wastewater treatment and the Green Drop programme, and to gauge their opinions on whether modelling should become a part of the programme.

2 Modelling Tools

Two steady state models were investigated. The first was developed in WEST at the University of KwaZulu-Natal, the second in Excel at the University of Cape Town. The latter was William Wu's MSc project, and the uncertainty about when it would become available for use was part of the motivation for developing the former. When the UCT model became available it proved to be better suited to the Green Drop objectives than the WEST version, and was used in the Umhlanga WWTP case study.

The UCT three phase (aqueous-gas-solid) plant-wide dynamic model (PWM_SA) was developed for simulation of nutrient removal activated sludge systems coupled with the anaerobic, aerobic and anoxic-aerobic digestion of sewage sludge including waste activated sludge (WAS) produced by Biological Excess Phosphorus Removal (BEPR) plants.

2.1 A WEST steady state model for small WWTPs

A steady-state simulation model was developed based on the Mariannridge WWTP, located near Pinetown in the eThekwini district municipality. This WWTP was the subject of a study that was part of WRC project K5/1734 Investigation into methods for the development of a protocol for quantitative assessment of industrial effluents for permitting of discharge to sewer, during which a dynamic model was developed and calibrated using operating data over 2 years (2006 and 2007) together with laboratory characterisation of the raw wastewater (Mhlanga et al., 2009). The data gathered during that project was used as the basis for the new steady state model. Mariannridge is typical of a number of the smaller WWTPs in eThekwini (and probably many other South African municipalities). It performs only COD removal and nitrification, i.e. without nutrient removal. It is operated in extended aeration mode, with a sludge retention time (SRT) of about 27 days and a hydraulic retention time (HRT) of 1.5 days.

The new steady state model presented here was an attempt to combine the advantages of the UCT steady state approach with those of the IWA dynamic approach. It is an input-output model, similar to the ASM1 model. It is implemented in the WEST simulation platform, and uses the ASM1 component set, and so can be seamlessly integrated into a WEST ASM1 model configuration. It also uses most of the ASM1 kinetic and stoichiometric parameters. On the other hand, it uses most of the steady state approximations described in Henze et al. (2008), with a few additional ones to address minor processes that appear in ASM1. These were calibrated to match the outputs of a full (dynamic) ASM1 model of the Mariannridge WWTP.

The Water Research Group (WRG) at the University of Cape Town (UCT) was also working on a steady state model as part a separate project. This became available after this model, and was subsequently adopted for the case study that was undertaken (chapter 3). Since the UCT model was implemented in Excel, it would be more readily adopted by smaller municipalities than this model implemented in WEST. Nevertheless, the WEST model is more suited than the Excel to evaluating the plant's operation of a period of time by comparing predicted outputs to time series of measurements, as shown in figures 2.2 and 2.3.

2.1.1 The Mariannridge WWWTP

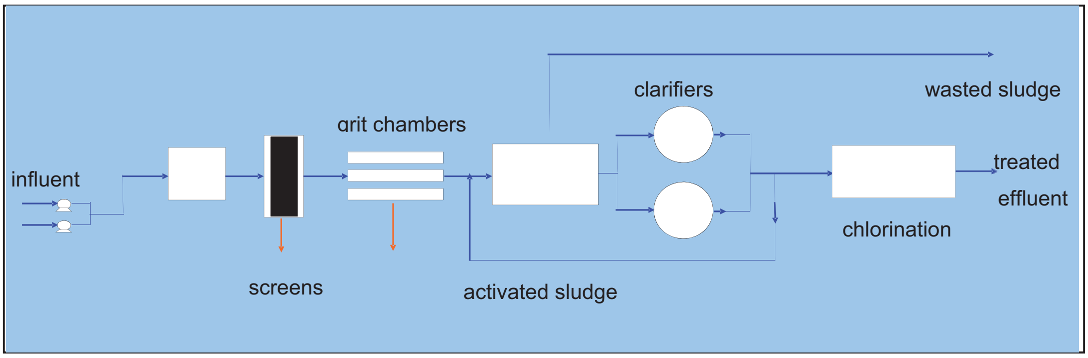


Figure 2.1: Flow diagram of the Mariannridge WWTP

The Mariannridge WWTP has a sister plant on the same site called Shallcross WWTP. These two parallel plants make up what is known as the Umhlatuzana Works. Mariannridge receives an average of 8 000 m^3/d wastewater, while Shallcross receives an average of 2 000 m^3/d. Results from the original dynamic modelling study are shown in the follow graph.

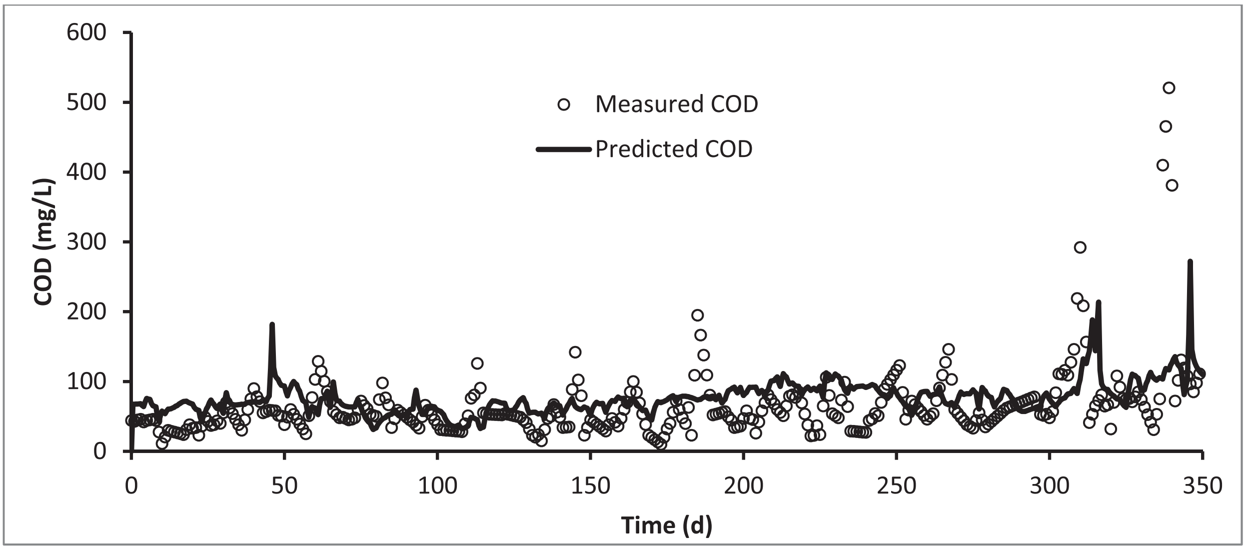


Figure 2.2: Simulation of Effluent COD for the year 2006

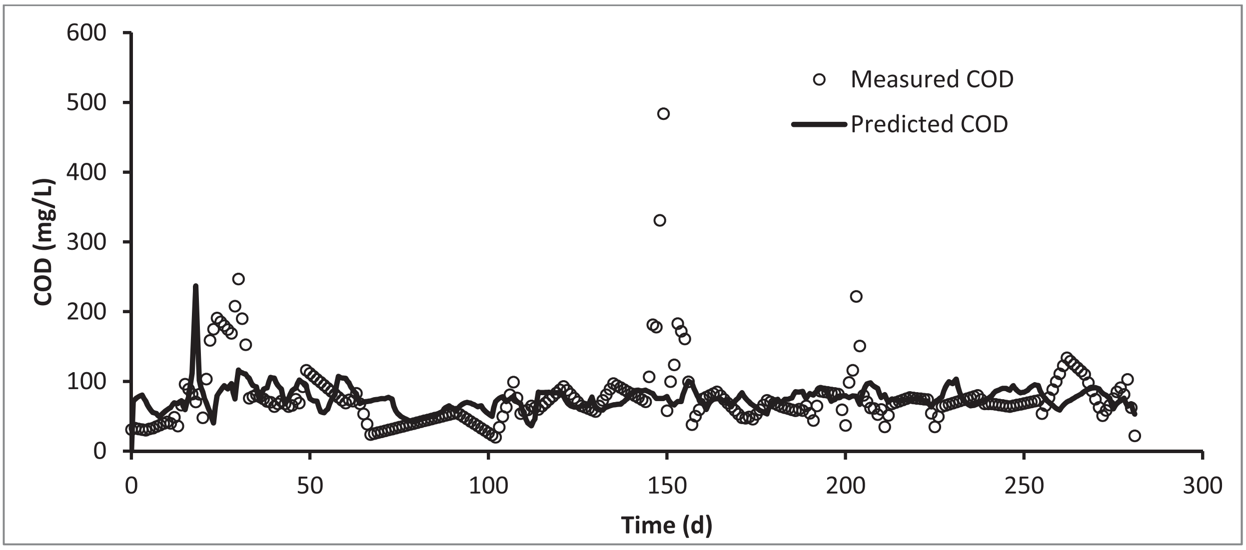


Figure 2.3: Effluent COD simulation after calibration for the year 2007

The dynamic model was calibrated on the 2006 data, and the 2007 data was used for validation. (WRC project K5/1734)

2.1.2 Steady State Model

The model is based on the ASM1 set of components:

Table 2.1: Components of the IWA ASM1 model.

Symbol	Description	Symbol	Description
H_2O	Water	S_ALK	Alkalinity (molal units)
S_S	Soluble biodegradable COD	X_I	Inert particulate organics
S_I	Soluble inert COD	X_S	Biodegradable particulate organics
S_O	Dissolved oxygen	X_BH	Heterotrophic biomass
S_NO	Nitrate	X_BA	Autotrophic biomass
S_ND	Soluble organic nitrogen	X_P	Particulate endogenous residue
S_NH	Free and saline ammonia	X_ND	Particulate organic nitrogen

Figure 2.4 shows the model in a simple WEST configuration:

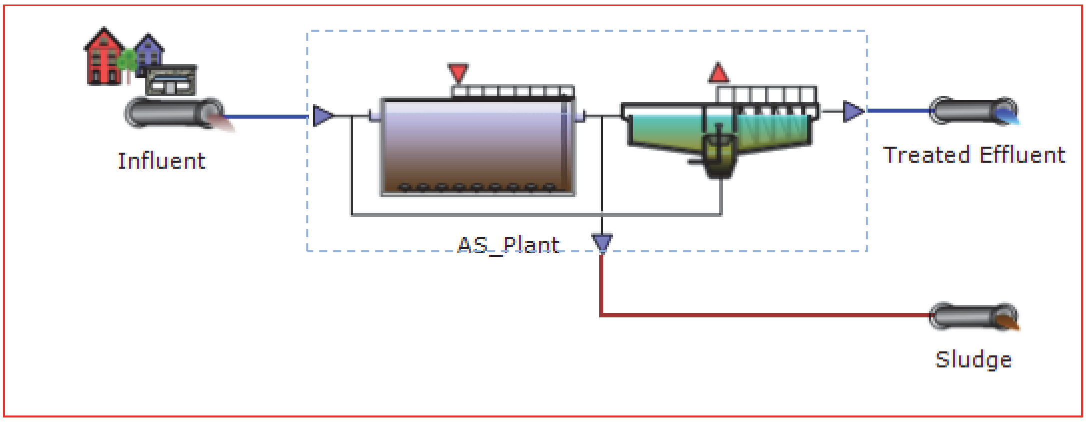


Figure 2.4: Steady state model configuration

The central steady state model (AS_plant) is represented by the coupled aeration basin and clarifier with recycle, the Influent, Treated Effluent and Sludge are standard WEST blocks. The influent stream is the input to the model, and the treated effluent and sludge streams are the computed outputs.

2.1.3 The reference ASM1 (dynamic) model

In developing the steady state model, the intention was not to match all aspects of the conditions prevailing during the previous study, as the model should be capable of representing the plant operation under a range of conditions. What were taken from the previous study were just the plant configuration and the feed characterisation. Furthermore, the original feed characterisation was expressed as a detailed breakdown of ASM3 (dynamic) model components, something which will seldom be available to operators for plant monitoring. WEST provides a fractionation model for estimating component concentrations from COD (Chemical Oxygen Demand), TKN (Total Kjeldahl Nitrogen) and TSS (Total Suspended Solids) which are often measured on a routine basis. So, for the steady state model, the detailed component fractions of the dynamic model for 2007 were aggregated into COD, TKN and TSS values. These were passed to the WEST influent model, which re-fractionated them to ASM1 components.

WEST has a steady state mode, which operates by integrating the dynamic model for a suitably long time, with the average influent composition as a steady input. To provide reference output values, the dynamic ASM1 model was run in steady state mode on the aggregated 2007 input data for ranges of HRTs (1.21-2.43 d), SRTs (8.3-40d) and aeration rates (0.1-5.6 mg/L DO).

2.1.4 Steady state model development

The model calculations are based largely on chapters 4 and 5 (Organic matter removal and Nitrogen removal) of Henze at al. (2008), adapted to the ASM1 set of components. These calculations are essentially identical for the components S_I, X_I, X_BH, X_BA , and X_P.

S_O is not addressed in the Henze et al. book, its concentration is assumed to be 'high enough'. In the WEST implementation of ASM1 it is controlled via the aeration mass transfer coefficient (kLa). For the steady state model it was implemented as a user-set parameter.

2.1.4.1 Biodegradable organics

In Henze et al. the soluble and particulate biodegradable components are assumed to be completely degraded. However the dynamic ASM1 model reflects low residual concentrations. Although these have a negligible influence on the COD balance, they do affect the balance of ammonia and nitrate in the effluent, according to the ASM1 model. For this reason, the results of the reference model were used to establish a simple correlation to estimate their concentrations. These could be adequately represented as proportional to the active biomass concentration (e.g. figure 2.5).

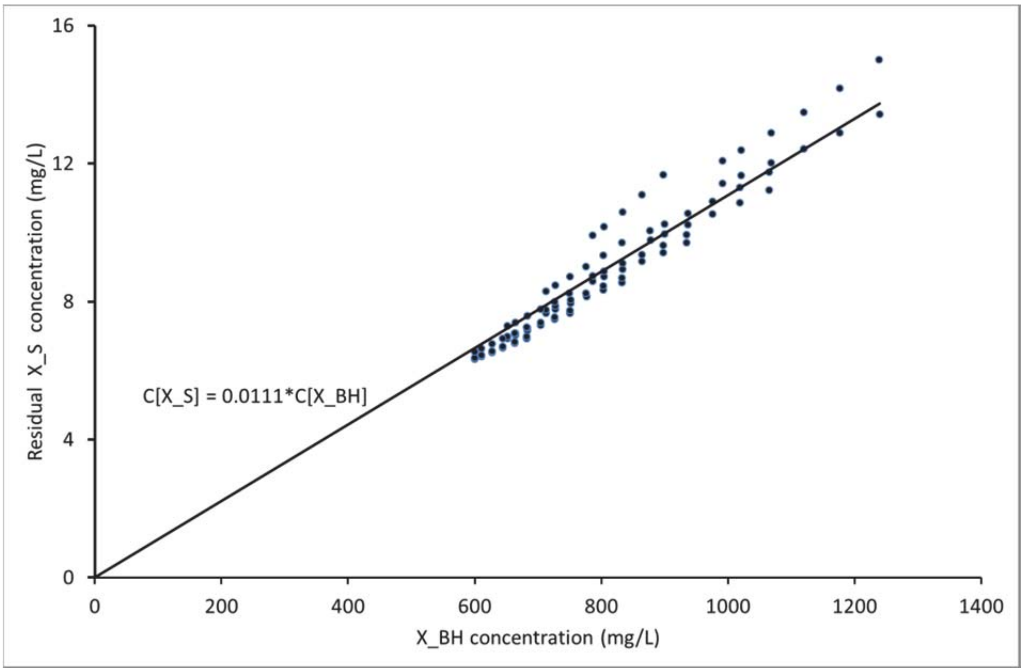


Figure 2.5: Correlation between residual biodegradable particulate concentration and active biomass concentration from the reference ASM1 model.

2.1.4.2 Nitrate

In Henze et al. (2008) denitrification (conversion to of nitrate to nitrogen gas) is not considered as part of the aerobic reactor steady state model, but is modelled as taking place in a separate anoxic reactor with a separate model. The ASM1 model, on the other hand uses the same model formulation to represent both aerobic and anoxic operation, with a continuous transition between the two, based on the DO concentration. This can be seen in Figure 2.6, which was prepared using the reference ASM1 model of the Mariannridge WWTP.

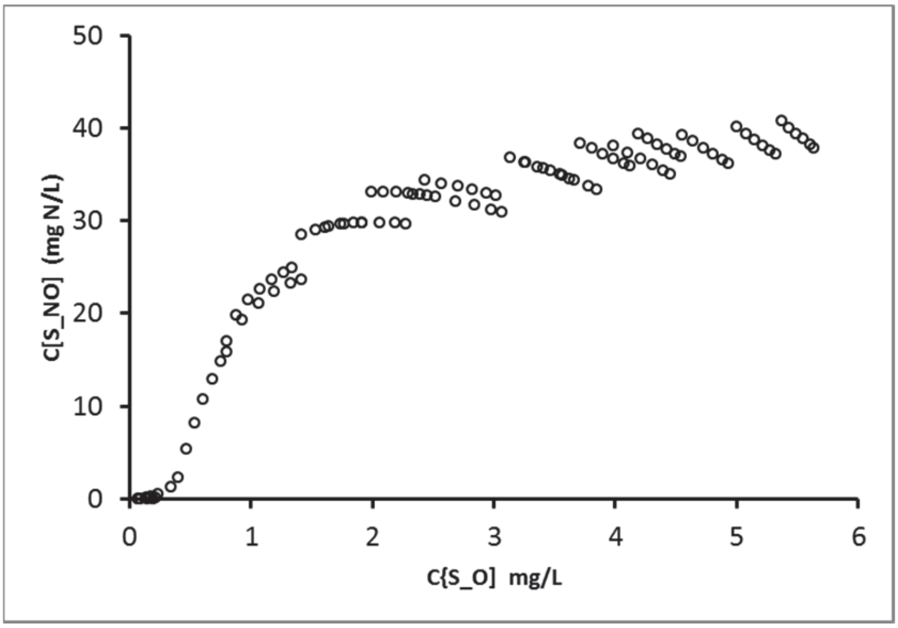


Figure 2.6: Relationship between dissolved oxygen and nitrate concentrations according to the reference ASM1 model.

As can be seen, the relationship is not exact, with HRT and SRT also having lesser influences. In the ASM1 reference model, the rate expression is fairly complex, depending on concentrations of X_BH, S_NH, S_NO and S_O.

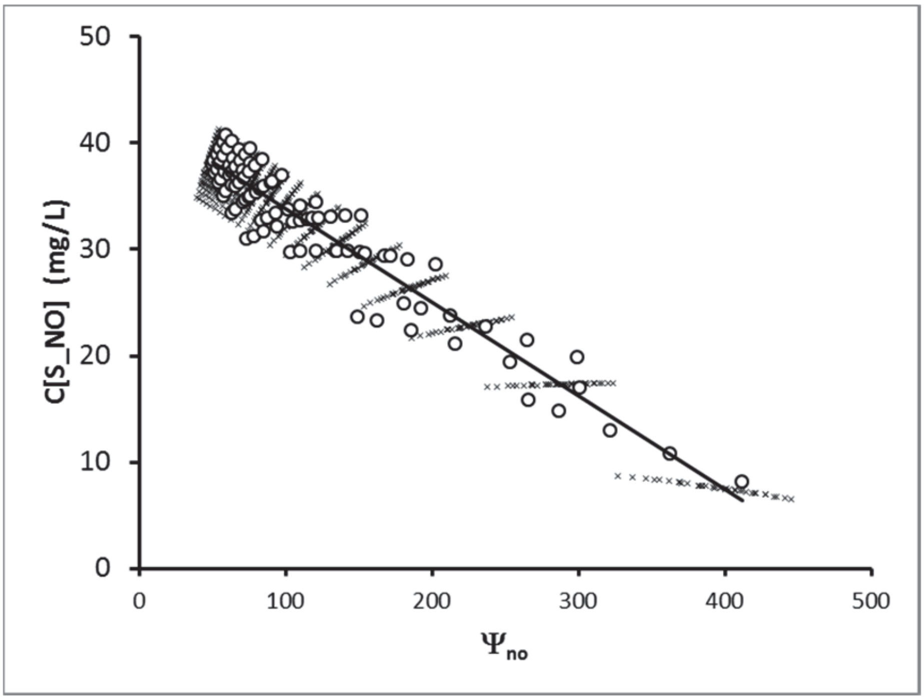


Figure 2.7: Correlation for nitrate concentration. The larger circles are from the reference ASM1 model, while the smaller points were generated by the steady-state model.

To avoid an iterative solution, an approximate relationship was derived from the reference solution results, as illustrated in Figure 2.7. The correlating term Ψ_{NO} is given by

$$\Psi_{NO} = 0.0878 \cdot \frac{K_{OA}}{K_{OA}+C[S_O]} \cdot C[X_{BH}]$$

This is a simplified version of the anoxic heterotrophic growth rate equation, with the missing terms lumped into the 0.0878 coefficient.

2.1.5 Steady-state model test on plant data

The data used to set up the model, although loosely based on data from the Mariannridge WWTP, was generated by the manipulating parameters of the ASM1 dynamic model. As a check that the resulting model does, in fact, provide a reasonable representation for monitoring the WWTP, it was run against the available plant data from 2007. This was a very rough test, because the measured data came from laboratory records without any information about operating parameters on the plant such as the aeration or the sludge flow.

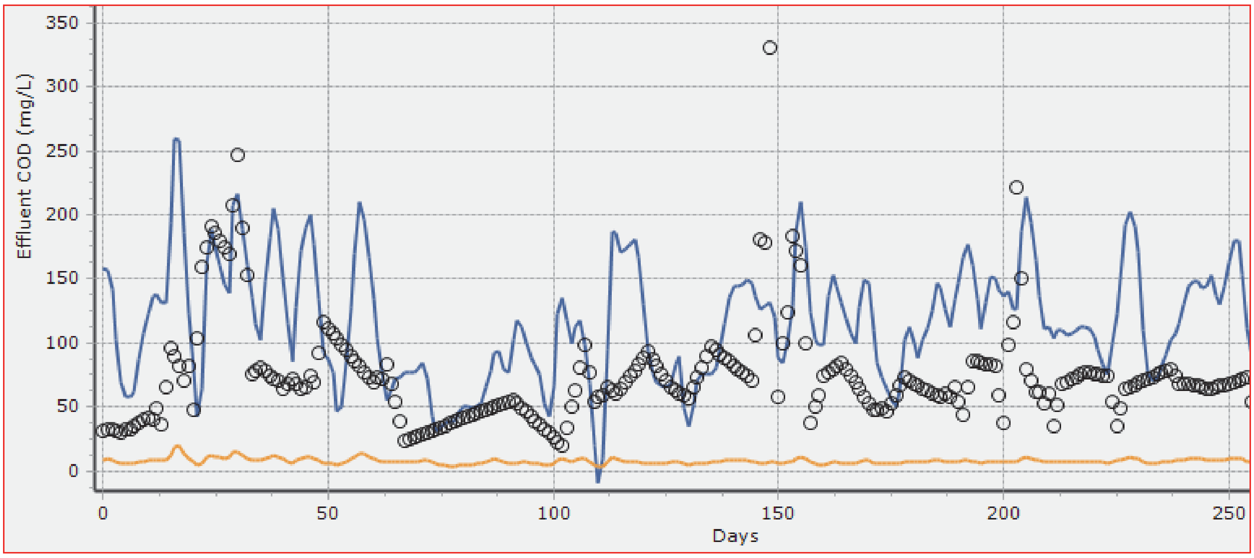


Figure 2.8: Comparison of measured and modelled effluent COD for the Mariannridge (2007 data) WWTP

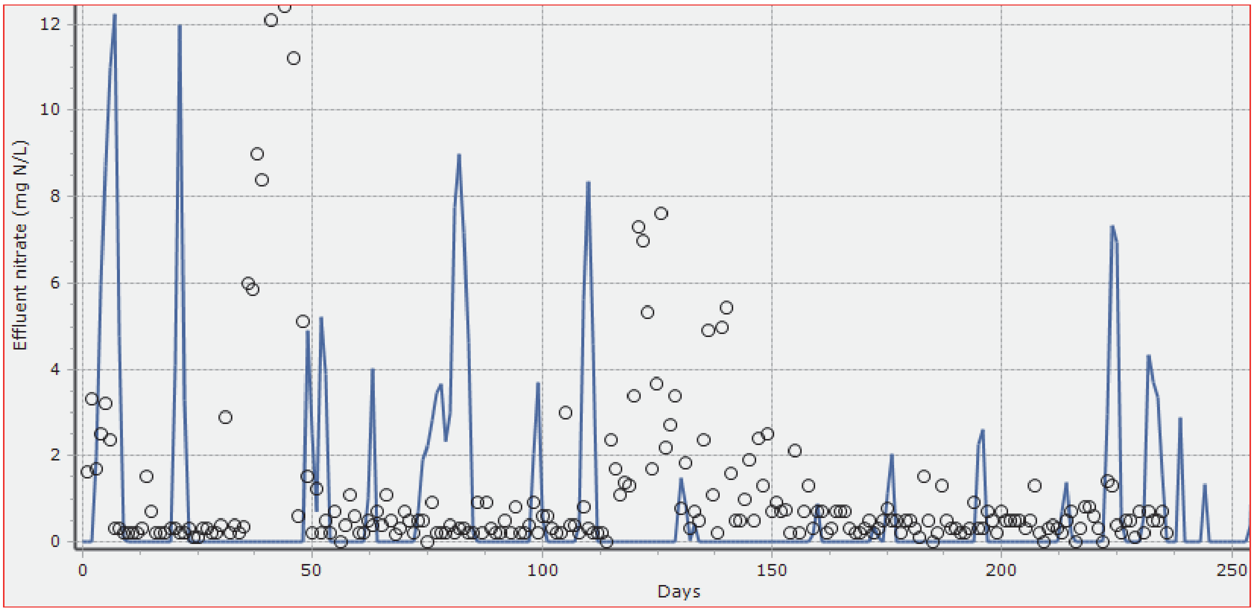


Figure 2.9: Comparison of measured and modelled effluent nitrate for Mariannridge WWTP

As can be seen from figures 2.8 and 2.9, there is a reasonable correspondence on average between predicted and measured effluent COD and nitrate, though obviously not in detail.

2.1.6 Discussion

The aim of this model is to take advantage of the accumulated experience embedded in WEST implementation of the ASM1 model, while reducing and simplifying the data requirements, so as to make it accessible to plant operators. These preliminary results are promising; however practical application may well uncover further issues that need to be addressed.

The ASM1 model is widely regarded as the most reliable wastewater treatment model, because it has been so widely used, and because of its relative simplicity. However, it is restricted to aerobic COD and nitrogen removal, and does not address phosphorus removal or anaerobic treatment. This is suitable for many of the smaller WWTPs in eThekwini, but another approach will be needed for those that are larger and more complex.

A related MSc project at the University of Cape Town (Wu, 2015) adapted the UCT spreadsheet design model for monitoring purposes, and when this became available it was deemed to be more appropriate for the Green Drop application, not least because it was implemented in Excel, which would be more accessible than WEST to operators of small treatment plants. This is discussed next.

2.2 The UCT Plant-Wide Steady State Design Model

Steady state modelling is a powerful tool for the design and optimization of wastewater treatment plants and is relatively easier to use and less "data hungry" than dynamic modelling. Steady state modelling can also be used to estimate the treatment capacity of a plant to treat a given wastewater (specifically with respect to the biological and separation process capacities), which is one of the requirements of the Green Drop programme (KPA (7) Wastewater Treatment Capacity) (Department of Water Affairs, 2015).

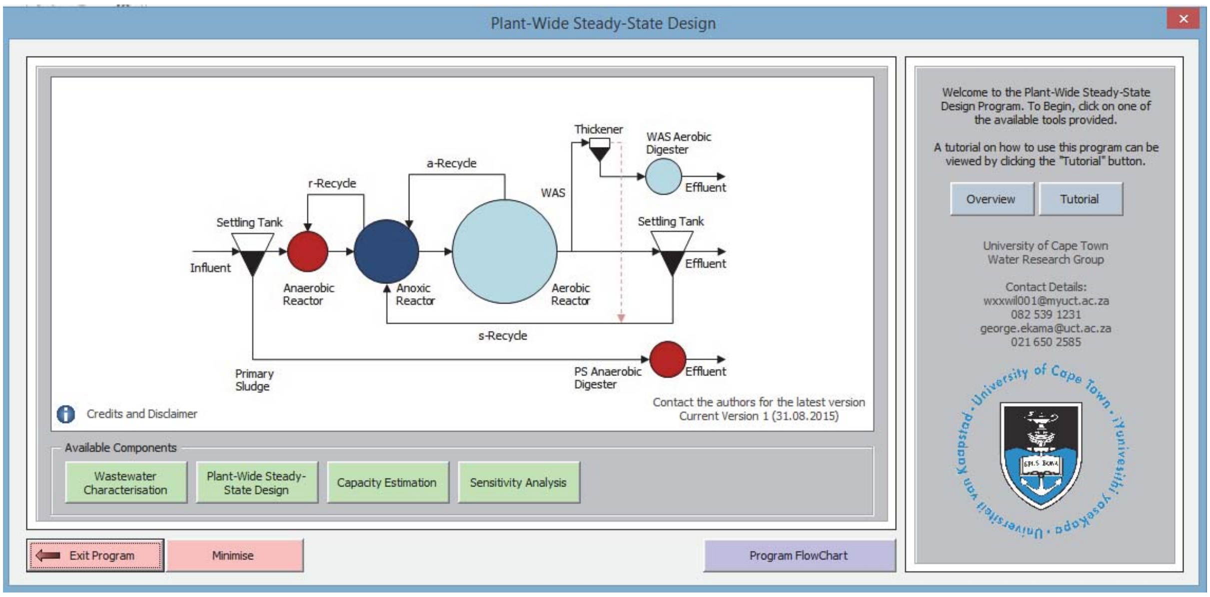


Figure 2.10: PWSSD User interface Main Menu

The Plant-Wide Steady State Design programme (PWSSD) (Wu, 2015) is an Excel/VBA implementation of the plant-wide steady-state model developed at UCT which has an interactive interface which guides the user through setting up a raw wastewater profile and entering the necessary information to either design a new plant or estimate the capacity of an existing plant to treat the wastewater. Although the model development was not formally tied to this WRC project, it was an outcome of a collaborative strategy in which UCT would provide the theoretical development, and this project would provide opportunities for practical application. The underlying steady state model equations on which PWSSD is based are described in detail in Henze et al (2008).

The PWSSD user interface currently offers four different tools as shown in Figure 2.10: i) wastewater characterisation tool, ii) design tool, iii) capacity estimation tool and (iv) sensitivity analysis (currently only for the secondary settlers).

2.2.1 Wastewater characterisation

The steady state model requires a fully characterised profile (soluble and particulate, biodegradable and unbiodegradable fractions, settleable and unsettleable, full C, H, N, O, P stoichiometry for all components, etc.) as an input. In the Wastewater Characterisation tool, the user selects from one of several data entry methods to generate the fully characterised wastewater profile, based on the level of data available. The available wastewater characterisation methods are summarised in Table 2.2.

Table 2.2: PWSSD wastewater characterization methods

Method	Data Level	Summary
1. Direct Input	Rich	Requires that the user already has a full characterised wastewater profile. A diurnal flow profile (DFP) can be applied, or at very least, a peak TOD amplitude factor must be specified.
2. DFD Data	Very good	Used to generate DFPs for wastewater characteristics from timed measurements.
3. Characterisation Tree	Good to moderate to poor	Can be used if designer does not have enough data for method 1
4. Grab/Composite Sample Reconciliation	Poor	Method still under development
5. Preloaded	Very poor	User can select the most appropriate wastewater profile from a set of preloaded profiles, or generate a new profile by modifying one of the preloaded profiles with available measurements. All preloaded profiles include DFPs

2.2.2 Capacity Estimation

The capacity estimation tool uses the steady state model to calculate the wastewater loads a given plant can treat. In the context of a plant audit, the calculated estimates can be compared to the original design capacities, actual plant loads and performance to determine whether the plant is over- or underloaded. The calculated estimates can also be used in an iterative manner to determine the accuracy of the model inputs. This is useful in determining the range of operating parameters such as sludge age and recycle ratios, and as well as to cross-check influent data and peak factors.

The capacity estimation tool estimates the maximum AWDF a plant can handle based on the system configuration and wastewater characteristics based on four limiting scenarios:

1) The total area of the SSTs (secondary settling tanks)
2) A user specified maximum aerobic Xt (MLSS concentration)
3) The aeration capacity of the system
4) The maximum solids flux the sludge handling facilities can accept

The capacity estimation output also includes the secondary effluent composition calculated by the steady state model.

The capacity estimation tool requires six different sets of information:

i. A wastewater characteristics profile set up in the Wastewater Characterisation Tool.
ii. General information on the plant configuration
iii. Information about the design and operation of the activated sludge basin
iv. Data required to calculate the amount of nitrification and denitrification occurring
v. Information required to calculate the SST capacity including the peak flows and sludge settlability
vi. Information required to calculate the aeration capacity.

The capacity estimation page showing the results page is shown in Figure 2.11.

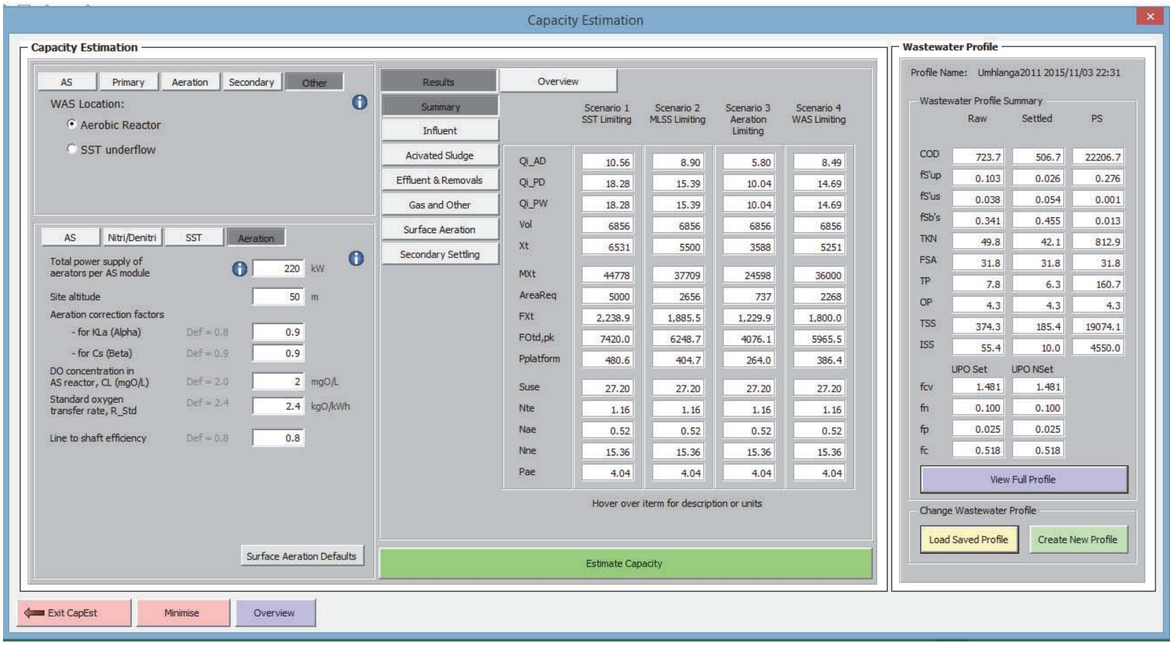


Figure 2.11: Capacity estimation page showing results

The ADWF calculated for each limiting scenario corresponds to a solution of the steady state model at which a specific criterion is met.

2.2.2.1 Scenario 1: SST limited

This scenario calculates the maximum ADWF (average dry weather flow) where the corresponding PWWF (peak wet weather) flow divided by the actual SST area is equal to the maximum applied overflow rate the clarifiers can handle. The maximum applied overflow rate is a function of the solids loading and sludge settlability (Takács and Ekama, 2008).

2.2.2.2 Scenario 2: MLSS limited

This scenario simply calculates the ADWF at which a user specified MLSS concentration is expected to occur. The steady state MLSS is a function of the influent characteristics and increases with influent loads.

2.2.2.3 Scenario 3: Aeration limited

This scenario calculates the ADWF for which the corresponding peak oxygen utilization rate (OURtd,pk) corresponds to the maximum aeration capacity available. OURtd,pk is a function of the diurnal variations in the influent COD and TKN. A peak TOD amplitude factor (TOD Amp) is either calculated from the diurnal variation in the TOD load or the TOD amplitude factor is specified as a raw water characteristic (Direct Input Method only) when the diurnal variations data is not available. A peak OURtd factor is estimated from the TOD amplitude factor using an empirical damping function (Musvoto et al, 2002).

2.2.2.4 Scenario 4: WAS limited

The waste activated sludge (WAS) limited ADWF is the solution of the steady state model at which the calculated WAS solids flux equals the maximum load the plant sludge handling facilities can manage. The WAS flux is a function of the influent loads and sludge age.

2.2.2.5 Effluent concentrations

In addition to the four capacity limiting scenarios, PWSSD also calculates the effluent COD, TKN, FSA, nitrate and orthophosphate concentrations. In the steady state model, the effluent concentrations do not depend on the plant flowrate and therefore are the same for all four scenarios.

The predicted effluent composition can be compared to measured plant data, however, it is important to understand the assumptions built into and limitations of the steady state model when trying to interpret them. For example, the steady state model assumes that all the biodegradable soluble influent COD (VFA and FBSO) is consumed in the AS reactor and all the particulates are removed in the SSTs. Therefore, the COD in the effluent is equal to the USO (soluble unbiodegradable organics) in the influent, which is specified by the user in the wastewater characterisation. Therefore, the effluent COD result does not provide any new information.

The calculations for effluent nitrogen and phosphorous are described in detail in Henze et al (2008) Chapters 5 and 7 respectively. Nitrogen removal is also explored in Section 3.6 of the current report.

The use of the capacity estimation tool is investigated in Chapter 3 and its potential application in the Green Drop programme is discussed.

2.3 The Plant Wide Dynamic Model: PWM_SA

The UCT three phase (aqueous-gas-solid) plant-wide dynamic model (PWM_SA) was developed for simulation of nutrient removal activated sludge systems coupled with the anaerobic, aerobic and anoxic-aerobic digestion of sewage sludge including waste activated sludge (WAS) produced by Biological Excess Phosphorus Removal (BEPR) plants.

The three phase plant-wide dynamic model is based on strict material mass balance and was developed by linking:

(i) a variation of ASM2 (Henze et al., 1995) for activated sludge nitrogen and phosphorus removal and aerobic or anoxic – aerobic digestion;

(ii) a variation of the anaerobic digestion model by Sötemann et al. (2005a).

The main extensions to the original models are through their integration within a three phase mixed weak acid/base chemical and physical processes models of the inorganic carbon, ammonia, acetate, propionate and phosphate systems.

More information on the calibration of the two models can be sourced from Ikumi et al. (2013).

2.3.1.1 Components

The model has a single set of components to represent both aerobic processes. Its particulate components include precipitates formed during anaerobic digestion, sewage particulate organics and biomass (with their storage products such as polyphosphate and poly-hydroxy-alkanoates included as separate components) from activated sludge and anaerobic digestion units. Carbon dioxide and methane are included as the two main gases that are evolved during transformation reactions for treatment of municipal waste. The remaining components are all soluble, including three dissolved sewage organics, thirteen dissolved ionic components, three dissolved gases (oxygen, hydrogen and nitrogen) and water.

Each component's elemental formulation is included – most components (e.g. dissolved ionic components, gases and precipitates) have distinct chemical formulations, which enables the direct calculation of their molar and material (COD, C, H, O, N and P) masses. However, the organic components (i.e. the seven organism groups and the sewage FBSO, USO, BPO, UPO) were given parameterized (variable) compositions in the general form ($C_XH_YO_ZN_AP_B$), so their compositions can be entered as model inputs. Therefore, the elemental molar ratios (i.e. the X, Y, Z, A and B values) of their elemental formulation were coded as model parameters to cater for the variability in sewage characteristics.

Table 2.3: The components used in the PWM_SA model

Name	Empirical formula	Description
H_2O	H_2O	Water
S_H	H^+	Hydrogen ion
S_Na	Na^+	Sodium
S_K	K^+	Potassium
S_Ca	Ca^{2+}	Calcium
S_Mg	Mg^{2+}	Magnesium
S_NH_x	NH_4^+	Ammonium
S_Cl	Cl^-	Chloride
S_VFA	CH_3COO^-	Acetate
S_Pr	$CH_3CH_2COO^-$	Propionate
S_CO_3	CO_3^{2-}	Carbonate
S_SO_4	SO_4^{2-}	Sulphate
S_PO_4	PO_4^{3-}	Phosphate
S_NO_x	NO_3^-	Nitrate
S_H_2	H_2	Dissolved hydrogen
S_O_2	O_2	Dissolved oxygen
S_U	$CH_{Yu}O_{Zu}N_{Au}P_{Bu}$	Unbiodegradable Soluble Organics
S_F	$CH_{Yf}O_{Zf}N_{Af}P_{Bf}$	Fermentable Biodegradable Soluble Organics
S_Glu	$C_6H_{12}O_6$	Glucose
X_U_inf	$CH_{Yup}O_{Zup}N_{Aup}P_{Bup}$	Unbiodegradable particulate organics
X_B_Org	$CH_{Ybp}O_{zbp}N_{Abp}P_{Bbp}$	Biodegradable particulate organics
X_B_Inf	$CH_{Ybps}O_{Zbps}N_{Abps}P_{Bbps}$	Primary sludge biodegradable particulate organics
X_PAO_PP	$K_{kp}Mg_{mp}Ca_{cp}PO_3$	Polyphosphate
X_PAO_Stor	$C_4H_6O_2$	Poly-hydroxy-alkanoate
X_Str_NH4	$MgNH_4PO_4.6H_2O$	Struvite
X_ACP	$Ca_3(PO_4)_2$	Calcium Phosphate
X_Str_K	$MgKPO_4.6H_2O$	K-struvite

Table 2.3 continued: The components used in the PWM_SA model

Name	Empirical formula	Description
X_Cal	$CaCO_3$	Calcite
X_Mag	$Mg\ CO_3$	Magnesite
X_Newb	$MgHPO_4$	Newberyite
X_ISS		Influent inorganic settleable solids
X_OHO	$CH_{Yo}O_{Zo}N_{Ao}P_{Bo}$	Ordinary heterotrophic organisms
X_PAO	$CH_{Yo}O_{Zo}N_{Ao}P_{Bo}$	Phosphate accumulating organisms
X_ANO	$CH_{Yo}O_{Zo}N_{Ao}P_{Bo}$	Autotrophic nitrifying organisms
X_ZAD	$CH_{Yo}O_{Zo}N_{Ao}P_{Bo}$	Acidogens
X_ZAC	$CH_{Yo}O_{Zo}N_{Ao}P_{Bo}$	Acetogens
X_ZAM	$CH_{Yo}O_{Zo}N_{Ao}P_{Bo}$	Acetoclastic Methanogens
X_ZHM	$CH_{Yo}O_{Zo}N_{Ao}P_{Bo}$	Hydrogenotrophic methanogens
X_U_Org	$CH_{ye}O_{ze}N_{ae}P_{be}$	Endogenous residue
G_CO₂	CO_2	Carbon dioxide
G_CH₄	CH_4	Methane

Based on experimental evidence, the un-biodegradable organics in the wastewater and generated in the activated sludge reactor (endogenous residue) remain un-biodegradable in the anaerobic digester (Ekama et al. 2006a; Ikumi et al., 2013b). The composition assigned to the endogenous residue is the same as that of the live Ordinary Heterotrophic Organisms (OHO) and Phosphate Accumulating Organisms (PAO) biomass which produces it. The composition of these un-biodegradable organics remains unchanged throughout the WWTP. All organism groups (aerobic and anaerobic) were given the same elemental formulation. The polyphosphate (PP, $MgdKeCafPO_3$) and polyhydroxy alkanoates (PHA, $C_4H_6O_2$) which are normally stored within (hence are part of) PAOs are included in the model as separate components, to avoid complications that would result in extending the PAO biomass formula (such as merging their stoichiometric coefficients in transformations).

2.3.1.2 Ionic speciation

The ionic speciation routine (Brouckaert et al., 2010), contained in the PWM_SA model provides a general algebraic approach to modelling the very rapid ionic dissociation and ion pairing equilibrium reactions separately from the slower biological and physical processes and can be applied to any combination of mixed weak acid/base systems. Because the weak acid/base chemistry processes for precipitation and gas exchange are slow, they are included with the slow bioprocesses, which are modelled with kinetic equations.

The aqueous ionic species input to the model is determined from the measured influent conductivity, temperature, pH, ortho-phosphate (OP), free and saline ammonia (FSA), NO_3^-, SO_4^{2-}, H_2CO_3 alkalinity and volatile fatty acids (VFA), where the last two were measured with the 5-point titration of Moosbrugger et al. (1992). These measurements allow complete speciation of the OP, FSA, VFA, inorganic carbon (IC) and water weak acid/base systems including ion-pairing, which is executed during influent characterisation. All the ions of the 5 weak acid/base systems (and NO_3^-, SO_4^{2-}) do not produce sufficient ionic strength to obtain the measured conductivity, so NaCl is hypothetically added to the model influent characterisation pre-processor (cf. pHAlkalinityIn transformer), to match the conductivity to the measured value. The addition of sodium chloride adds the necessary ionic strength for correct adjustment of dissociation and stability constants and solubility products and establishes the initial charge concentration associated with the measured conductivity.

2.3.1.3 Inter-phase transfers

The PWM_SA model considers three phases (liquid, gas and solid) and so can simulate active gas exchange through liquid to gaseous phase evolution and multiple mineral precipitation from liquid to solid or dissolution from solid into liquid phase.

2.3.1.3.1 Liquid-Gaseous phase transfer

Six gases are considered in the model (i.e. CO_2, CH_4, H_2, NH_3, N_2 and O_2). Ammonia (NH_3) is known to is highly soluble (Sötemann et al., 2005a, b) and its evolution is effectively zero. Therefore (for simplicity), unlike the other five gases, NH_3 is not included as a model component but is calculated in the equilibrium speciation routine mentioned above.

Methane (CH_4) is relatively insoluble in water (at atmospheric pressure) and is not utilized in the biological or chemical processes. It is a model component and is modelled to be directly produced in its gaseous phase by the AD methanogenic processes (acetoclastic and hydrogenotrophic methanogenesis). Carbon dioxide (CO_2) is significantly soluble and is evolved relatively slowly, hence needs to be modelled with the CO_2 evolution process such as that presented by Musvoto et al. (1997) and Sötemann et al. (2005a)

2.3.1.3.2 Liquid-Solid phase transfer

In the anaerobic and aerobic digestion unit operations, (especially those treating sludge from biological excess phosphorus removal systems), magnesium, potassium and calcium can be present at sufficiently high concentrations for the occurrence of precipitation. The solids most likely to precipitate were identified by Musvoto et al. (2000) as struvite ($MgNH_4PO_4$), newberyite ($MgHPO_4$), amorphous calcium phosphate ($Ca(PO_4)_2$), calcite ($CaCO_3$) and magnesite ($MgCO_3$). The ionic speciation described above plays a significant role in the dynamics of multiple mineral precipitation. This is because, as free ions (usually the least protonated) get incorporated into the precipitating minerals, other ions of the same type, bound in ion pairs, get released into the aqueous solution in the process of maintaining equilibrium in the aqueous phase. This continues to happen and influences pH for as long as the ionic product of the relevant species concentrations exceeds the solubility product of the mineral.

2.3.1.3.3 Biological processes

ASM2 is a widely accepted model that is broadly applied in NDBEPR system design, operation and process optimisation for activated sludge systems.

It includes the biological growth and death processes for OHO, PAO and Autotrophic nitrifying organisms (ANO) biomass and predicts oxygen demand and sludge production together with storage and lysis of polyphosphate and poly-3-hydoxyalkanoates for PAOs for strictly aerobic P uptake BEPR. The ASM2 was selected as the basis for PWM_SA because it was calibrated against consistent set experimental results (Wentzel et al., 1989a,b; Clayton et al., 1991).

ASM2, which was developed for simulating activated sludge systems, can also be used to simulate the aerobic digestion as a continuation of the biomass aerobic endogenous process.

ASM2 model was modified by including the inorganic settleable solids (ISS) model of Ekama and Wentzel (2004), together with the mixed weak acid/base chemistry model described above. The integration of these models together with the set of universally selected components, to ensure complete compatibility in development of the plant-wide model, required converting the model process stoichiometry from COD-based to mass concentration-based. The kinetic and stoichiometric coefficients for the ASM2 rates were also evaluated and transformed to be compatible with the revised components and stoichiometric process coefficients in different units. In some cases the kinetic equations, together with their included parameters, were changed to make them consistent with the components of the PWM_SA model.

Also added to the ASM2 model, is the process of CO_2 stripping from the aerated reactor together with processes that cater for mineral precipitation and dissolution processes that could take place during the aerobic or anoxic-aerobic digestion of NDBEPR WAS. For the aerobic CO_2 striping process, the CO_2 is continuously generated with aeration, hence no equilibrium is achieved between the aqueous ($H_2CO_3^*$) and gas phase CO_2 (Sötemann et al., 2005a).

The anaerobic digestion (AD) model of Sötemann et al. (2005b) is similar to the IWA ADM1 model (Batstone et al., 2002) in that it includes the reactions mediated by the same four organism groups (acidogens, acetogens, acetoclastic methanogens and hydrogenotrophic methanogens), but with a single hydrolysis process acting on a generic organic material representing sewage sludge, i.e. $C_XH_YO_ZN_A$. This hydrolysis process directly generates generic carbohydrate (glucose) while maintaining COD, C, N, H and O mass balances also producing NH_3 and taking up H_2CO_3.

This PWM_SA model extends the ASM and ADM models by Sötemann et al. (2005a,b) by adding P and integrating it within a three phase mixed weak acid/base chemical and physical processes model of the inorganic carbon, ammonia, acetate, propionate and phosphate systems.

2.4 The PWM_SA Probabilistic Fractionator

The principle of the fractionation algorithm is to start with a general estimate of the wastewater composition based on literature or previous experience, and to modify the composition when one or more measurements are encountered. The component concentrations are the state variables in a set of differential equations in time, and the general estimate referred to above constitutes the set of initial values. The expected values for all measurements considered by the algorithm can be calculated from the model component concentrations at any point in time. The measurement values are input to the algorithm as time series. When input measurements are encountered, their values are compared to the corresponding predicted values. The differences between predictions and measurements generate derivatives in the differential equations which drive a weighted least-squares objective function towards its minimum. This way, any combination of measurements can be taken into account (including none at all) at any point in the time sequence, because if there is no information to cause a component concentration to change, it will remain at is current value.

A secondary mechanism uses correlations between measurements to generate estimates of certain measurements where the actual measurements are not available. These estimates are then used as second class measurements in the algorithm, with lower weightings in the objective function. So, for example, the un-biodegradable particulate fraction of COD is hardly ever measured, but is important for modelling purposes. The commonly measured total COD, total suspended solids and total Kjeldahl nitrogen provide no information on this fraction. However the fraction can be estimated from the total COD using a literature correlation. In this way, the algorithm will generate a value that at least conforms to what is expected from experience.

The adjustable parameters in this scheme are the initial values for the component concentrations, upper and lower limits for each component concentration, the factors in the correlations between measurements, the weighting factors for measurement and measurement estimates, and a time constant that sets how fast the fractionation responds to a new measurement.

The elemental contents of the components can also be considered as parameters, because they determine the contribution of each component to each predicted measurement. However these stoichiometric factors are parameters of the entire model, not specifically the fractionation model.

Table 2.4: Measurements considered by the algorithm. All units are g/m^3

Symbol	Description
COD	Total COD
COD_filtered	Filtered (soluble) COD
COD_oho	COD of micro-organisms (taken to be ordinary heterotrophic organisms)
COD_up	Un-biodegradable particulate COD
COD_us	Un-biodegradable soluble COD
FSA	Free and saline ammonia
ISS	Inorganic suspended solids
ON_up	Organically bound un-biodegradable particulate nitrogen
ON_us	Organically bound un-biodegradable soluble nitrogen
OP_up	Organically bound un-biodegradable particulate phosphorus
OP_us	Organically bound un-biodegradable soluble phosphorus
OrthoP	(soluble) ortho-phosphate
TKN	Total Kjeldahl nitrogen
TKN_filtered	Filtered (soluble) TKN
TOC	Total organic carbon
TOC_filtered	Filtered (soluble) TOC
TP	Total phosphorus
TP_filtered	Filtered (soluble) total phosphorus
TSS	Total suspended solids
VFA	Volatile fatty acids (as acetic acid).

Table 2.5: Measurement correlation parameters

parameter	Description
f_codf	Soluble fraction of total COD
f_codup	Un-biodegradable particulate fraction of total COD
f_codus	Un-biodegradable soluble fraction of total COD
f_iss	Inorganic fraction of total suspended solids
f_oho	Fraction of total COD contributed by OHOs (ordinary heterotrophic organisms)
f_fsa	Ratio of FSA/TKN (mg N/mg N)
f_tkn	Ratio of TKN to total COD (mg N/mg COD)
f_tknf	Ratio of filtered TKN to total COD
f_tp	Ratio of total phosphorus to total COD (mg P/mg COD)
f_tpf	Ratio of filtered total phosphorus to total phosphorus
f_vfa	Fraction of total COD contributed by VFAs (taken to be acetic acid)
f_tss	Ratio of TSS to total COD

The ionic composition is determined from the following set of measurements plus the FSA, OrthoP and VFA described above.

Table 2.6: Ionic composition measurements

Measurement	Description
pH_in	pH
Alkalinity_in	Alkalinity (mg CO_3/L) - total or carbonate acooring to AlkalinityOption
Conductivity_in	Electrical conductivity (mS/m)
K,Ca,Mg,SO_4	Potassium, Calcium, Magnesium, sulphate

The composition is eventually expressed in terms of the full set of PWM_SA components. However, a number of these are expected to be negligible in and influent wastewater stream and are set to zero.

Table 2.7: Components determined from the organic fractionation

Component	Description	Component	Description
s_NH	Ammonia	x_OHO	Ordinary heterotrophic organisms
s_VFA	Acetate	x_B_Inf	Biodegradable particulate organics
s_PO4	Phosphate	x_ISS	Inorganics suspended solid
s_U	Un-biodegradable soluble organics		
s_F	Biodegradable soluble organics		
x_U_Inf	Un-biodegradable particulate organics		

Table 2.8: Components determined from the ionic speciation

Component	Description (all units mg/L)
S_H	Total hydrogen
S_Na, S_K	Total sodium and potassium
S_Ca, S_Mg	Total calcium and magnesium
S_Cl, S_SO4	Total chloride and sulphate

Table 2.9: Components assumed to have zero concentrations

Component	Description (all units mg/L)
S_Pr, S_H_2	Propionate and dissolved hydrogen.
S_Glu, S_NOx	Glucose and nitrate
X_B_Org, X _U_Org	Organic particulates produced by biomass death
X_PAO_PP	Polyphosphate content of Phosphorus Accumulating Organisms
X_PAO_Stor	PAO storage component
X_Str_NH4, X_Str_K	Struvite and K_struvite
X_ACP, X_Cal, X_Mag	Calcium phosphate, calcite and magnesite
X_PAO … X_ANO	All microorganisms except OHOs
G_CH4 … G_N2	Gases

Two versions of the fractionator were developed. For dynamic modelling an in-line version was implement in WEST, and for the steady-state modelling, an Excel spreadsheet version was developed. These work on essentially the same principle, with differences in detail as to how the equations are solved, as a result of the solvers available in WEST and Excel.

3 Assessment of Steady-state Modelling

3.1 Introduction

In this exercise, the PWSSD programme introduced in Section 2.2 was used to estimate the capacity of a small WWTW using available raw water characteristics, design and operating data. The capacity estimation exercise attempts to determine whether a given plant should be able to consistently handle the raw waste loads it receives. If the plant is overloaded then a major upgrade may be required whereas if it is operating at or below capacity then incidences of poor performance can presumably be attributed to operating and maintenance issues. Capacity estimation can therefore help to determine the most appropriate and cost effective strategies for improving performance under the Green Drop programme.

The modelling exercise also explores the data requirements of capacity estimation, the most critical factors in determining the capacity and the impact of uncertainty in the data on the reliability of the estimates.

Most of the data required for the model should be available from the design documents and routine plant monitoring data which is already required for the Green Drop process. The exception is the fractionation of the raw water into the various soluble and particulate, biodegradable and unbiodegradable, settleable and unsettleable components required as inputs to the model. This case study therefore also explores the use the offline probabilistic fractionator implemented in Excel (Section 2.4) as a tool for addressing the gaps in the raw water measurements.

3.2 The Umhlanga WWTP

3.2.1 Plant description

The Umhlanga WWTP is an extended aeration plant serving the Northern Coastal area of the eThekwini Municipality. The Umhlanga plant was inherited by eThekwini without its original design documents therefore there is some uncertainty about it is original design capacity. Consequently, eThekwini is particularly interested in getting reliable and objective capacity figures for this plant.

Extended aeration is a simplified design typically used in smaller treatment works and package plants, usually consisting of degritting, aeration and secondary clarification. Extended aeration plants operate in the endogenous respiration phase of the growth curve which requires long aeration times (hydraulic retention time 18-36 h)and sludge ages (20-30 days) and limits it to low organic loads (0.05-0.15 kg BOD_5/kg MLVSS/d) (Tchobanoglous and Burton, 1991). Long sludge ages in extended aeration processes allow the production of relatively stable sludge (low residual biodegradable organics) from the aerobic basins without primary sedimentation or secondary sludge treatment (Ekama and Wentzel, 2008a).

At the Umhlanga plant, the flow of raw sewage to the plant is controlled by a gate valve which is usually set to maintain an average daily flow of 5-7 ML/d during normal operation with the rest of flow being diverted to the Phoenix works. The flow arrangements are described in greater detail in Section 3.3.2. The flow can be further throttled when one of the activated sludge basins is offline. The raw sewage is screened and degritted before entering the aeration basin which consists of 4 parallel aerated lanes A, B, C and D. Lanes A and B are divided into three equal compartments, while lanes C and D are not (Figure 3.1). Each lane has three 22 kW surface aerators spaced evenly along its length. The first aerator in each lane is operated at low speed while the remaining aerators are operated at high speed.

The mixed liquor goes to two clarifiers and the sludge produced is returned to the aeration basin by two screw pumps while the secondary effluent overflows into a pond system where it is disinfected with chlorine before being released to the river.

Mixed liquor is wasted directly from the aeration basins and was previously sent to a thickening unit consisting of a DAF unit followed by centrifugation of the DAF float as shown in Figure 3.1. However, the waste sludge is now sent to Phoenix WWTP for treatment instead.

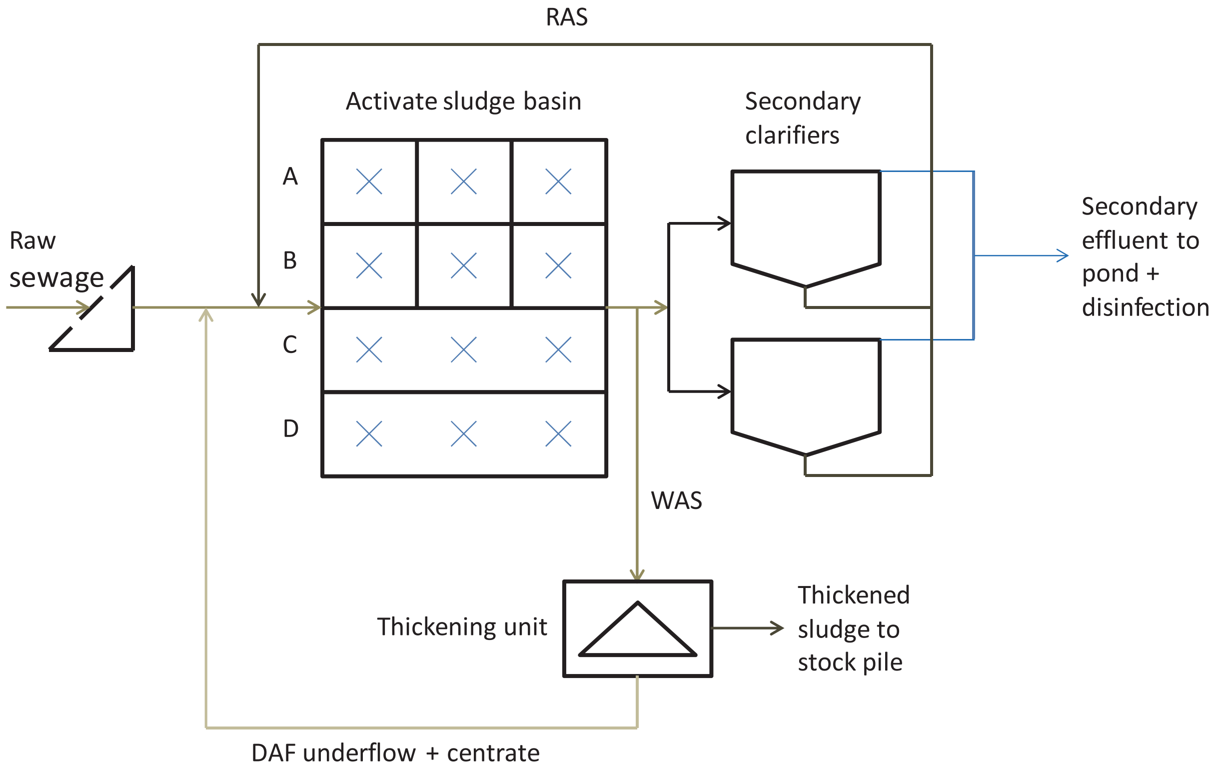


Figure 3.1: Umhlanga WWTP Plant Layout

Umhlanga furthermore has a known problem that untreated sewage is discharged into the pond system when the pumps for the raw water diversion line to Phoenix fail, however, this is not a an issue that can be addressed by steady state modelling.

3.2.2 Data Provided

The following data was initially provided for the Umhlanga WWTP

- Plant design data, dimensions, information on pumps and aerators
- For 1996: Average and range of inflow, rainfall, sludge wasting, laboratory analyses (raw sewage, secondary effluent, pond effluent, mixed and supernatant liquor, return activated sludge, DAF underflow and float)
- For 2009-2014:
  - Daily 12 hour composite samples analyses for screened sewage secondary effluent
  - Daily grab samples for pond effluent
  - Twice weekly 12 hour composite samples for mixed liquor suspended solids and nitrate, return activated sludge solids.
  - Operation data: including inflow, rainfall, DAF feed, centrifuge feed and cake

On the project team’s request, the following additional data on the flow arrangements and diurnal flow variations was provided:

1. Hourly flowrates collected between 6 a.m. and 5 p.m. for two non-consecutive weeks in June 2015
2. Data on the inflow split between Umhlanga and Phoenix for April and May of 2014.

3.3 Wastewater Characterisation

In PWSSD, each wastewater is generally characterized in terms of its flow weighted average (FWA) composition and diurnal variations.

3.3.1 Flow weighted average profile

The steady state model itself requires a very detailed wastewater characterization profile including many parameters which are not usually measured such as COD fractions. Therefore, as discussed in Section 2.2, the wastewater characterization tool allows the user several different data entry methods based on the richness or scarcity of the available information and the model generates estimates of the characteristics for which there is no direct data. The method which requires the least data input is the Preloaded method, in which the user can select from several preloaded profiles or set up a new profile based on limited data. Where data is missing, the closest preloaded profile can be used to estimate the missing fractions. This was the approach initially taken in this exercise because the raw wastewater characteristics available from routine monitoring were insufficient for any of the other methods. The preloaded profile selected as a template for Umhlanga was the profile for the Macassar WWTP, a 35 ML/d extended aeration plant with a low concentration feed.

Subsequently, the probabilistic fractionator (Section 2.4) became available, making it possible to generate a fully characterized wastewater profile based on the raw water monitoring data available for Umhlanga WWTP. The direct input method then became the simplest and most practical method for inputting the wastewater profile.

The particulate content of the wastewater is an important part of its characterisation and raw suspended solids data was only available from April 2011, therefore only data from this point onwards was considered for the characterisation exercise. An audit usually considers a year’s worth data and since the period April 2011 - March 2012 had the most complete data set, this was chosen for the capacity estimation exercise.

Table 3.1 compares the preloaded Macassar profile, the Umhlanga profile generated by scaling the missing measurements to the Macassar values, and the profile generated by the static (Excel based) probabilistic fractionator and the Direct Input Wastewater Characterisation method. The measurements used were actually twelve hour composites, however, in the absence of detailed information on the actual diurnal variations, it was assumed that these were a reasonable approximation of the flow weighted averages.

Note that the wastewater characterisation tool requires entries for both the settleable and non-settleable fractions of each type of particulate for the primary settler model. Since the Umhlanga WWTP does not have a primary settler, the split between these fractions is not important and the same ratios of settleable to non-settleable BPO (biodegradable particulate organics), UPO (unbiodegradable particulate organics) and ISS (inorganic suspended solids) as Macassar were used.

In Table 3.1 the missing fractions in the Umhlanga profile were calculated either by scaling them to the COD in the same ratio as Macassar or using the probabilistic fractionator. The TSS and TKN were calculated outputs rather than inputs to the Wastewater Characterisation Tool. Since TSS measurements were available, the calculated TSS can be used as a basis for comparing the two profiles. Umhlanga has a higher COD/TSS ratio than Macassar so simply scaling up the particulate COD fractions results in an overestimation of the total particulates. The probabilistic fractionator does a better job of reconciling the total COD and TSS although it calculates the average total COD to be slightly less than the measured values.

Table 3.1: Average wastewater characteristic profiles

	Macassar WWTP	Umhlanga April 2011 - Mar 2012, average measurements	Umhlanga April 2011 - Mar 2012, scaled to Macassar	Umhlanga April 2011 - Mar 2012, probabilistic fractionator
Total COD, mg COD/L	709.0	**737.5**	**737.5**	729.2
VFA, mg COD/L	30.8		32.1	30.8
FBSO, mg COD/L	114.5		119.1	151.2
USO, mg COD/L	35.5		36.9	27.2
BPOset, mg COD/L	178.2175		185.4	173.7
BPOsus, mg COD/L	279.0724		290.3	271.8
UPOset, mg COD/L	58.35446		60.7	61.3
UPOsus, mg COD/L	12.54554		13.0	13.2
FSA, mg N/L	40.7	**27.8**	27.8	31.7
OP, mg P/L	4.5	**3.8**	3.8	4.4
ISS, mg ISS/L	66.7		69.4	55.5
% Settleable ISS	65.7		65.7	65.7
Alkalinity, mg $CaCO_3$/L	350.0		350	350
pH	7.0		7.3	7.3
TSS, mg TSS/L	415.4	**405.5**	432	397.9
TKN, mg N/L	58.5		46.4	50

VFA = volatile fatty acids; FBSO = fermentable biodegradable soluble organics; USO = un-biodegradable soluble organics; BPOset = settleable biodegradable particulate organics; BPOsus = non-settleable biodegradable particulate organics; UPOset = settleable un-biodegradable particulate organics; UPOsus = non- settleable un-biodegradable particulate organics; FSA = free and saline ammonia; OP = orthophosphate; ISS = inorganic suspended solids; TSS = total suspended solids; TKN = total kjeldahl nitrogen

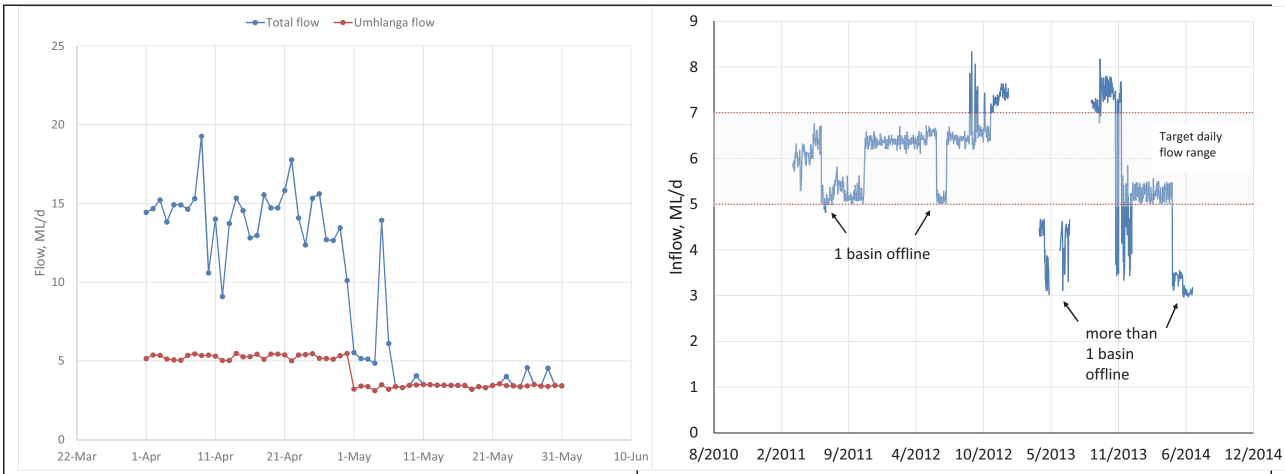
3.3.2 Umhlanga Flow Arrangements and Diurnal Variations

In addition to the flow weighted average composition of the wastewater, the capacity estimation tool also requires the following peak factors:

1. Peak wet weather and peak dry weather flow factors (PWWF and PDWF) for the secondary settler calculations. The SST area limited capacity is calculated based on the PWWF which is required to be equal or greater than PDWF.
2. A peak (hourly) TOD amplitude factor based on which the peak oxygen demand is calculated in the aeration limited scenario.

Diurnal variations in flow and mass loadings are a function of the size of the catchment and water consumption patterns with smaller catchments typically having lower total flows but higher peaking factors. However, the Umhlanga WWTP is atypical in this respect because the inflow can be regulated with the excess flow being diverted to the larger Phoenix WWTP. Figure 3.2 shows the inflow to Umhlanga as a fraction of the total flow in the trunk sewer for April and May 2014 while Figure 3.3 shows the inflow to Umhlanga for April 2011 to June 2014.

The flow into Umhlanga is regulated by a gate valve which the operators typically set to achieve a totalized daily flow of 5 to 7 ML/d. From Figures 3.2 (a) and (b) this clearly eliminates most of the day to day flow variation with most of the daily flows falling in the 5 to 7 ML/d range.



(a) Total sewer flow and Umhlanga inflow April and May 2014	(b) Umhlanga inflow April 2011 to June 2014

Figure 3.2: Umhlanga WWTP daily flows

The flow into Umhlanga can be throttled when one or more of the activated sludge basins are offline as shown in Figure 3.2. Spikes in the inflow could potentially occur when the diversion line pumps fail however the excess flow is generally sent straight to pond system. Alternately spikes may occur due to errors in setting the influent valve. Nonetheless, it is clear from Figure 3.2 that the peak flows do not depend on seasonal variations in this particular case. Therefore the ADWF was calculated as the average daily flow when all four activated sludge basins were in operation.

The throttling of the inflow also reduces the diurnal flow variations. The operators report that when the daily flow is set 5-7 ML/d, the peak (hourly) flow observed is 350 to 450 m^3/h (8.4 to 10.8 ML/d). Figure 3.3 shows hourly flowrates collected during the day shift over two non-consecutive weeks in June 2015. The measured flows are compared to the typical ranges reported by the operators. The average daily flows were on the low end of the expected range (~ 5 ML/d) and most of the peak hourly flows were < 350 m^3/h. Based on the limited data, assuming a peak flow of 450 m^3/h (10.8 ML/d) for the capacity estimation exercise appears to be a reasonable and conservative approach.

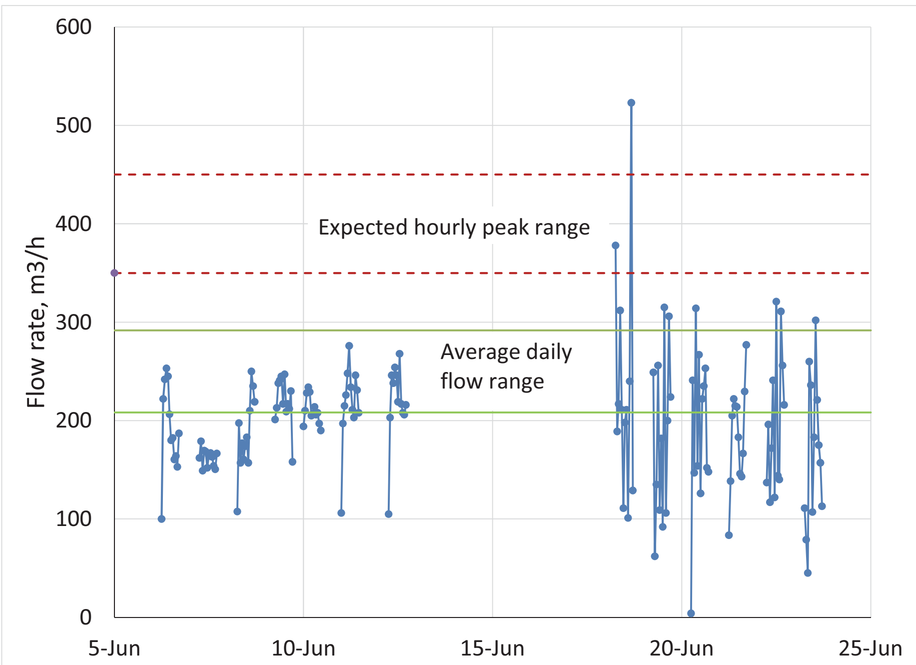


Figure 3.3: Diurnal flow variation June 2015

In addition to the peak flow factors, the daily peak TOD load is required which is calculated as a function of the daily variations in the influent COD and TKN loads. The calculation of the peak TOD amplitude factor for the Macassar profile is illustrated in Figure 3.4. Data on diurnal variations in raw wastewater composition are not routinely collected and this information is not likely to be available in a capacity estimation exercise. Alternately, the Direct Input wastewater characterisation method allows the user simply to specify the peak TOD amplitude factor directly which was the approach taken in this study. The peak oxygen uptake rate (OURtd, pk) calculated by the model is related to the peak TOD amplitude factor by Equation 3.1.

$$OURtd\ peak\ factor = 1 + damping\ factor \times TOD\ peak\ amplitude\ factor \tag{3.1}$$

Table 3.2 summarises the average flows and peak factors used in the capacity estimation exercise.

Table 3.2: Flows and peak factors for April 2011 - March 2012

ADWF	6.2 ML/d = 260 m3/h	
PDWF	10.8 ML/d = 450 m3/h	PDWF factor = 1.73
PWWF	10.8 ML/d = 450 m3/h	PWWF factor = 1.73
Peak TOD amplitude factor	1.20	Model default value
Damping factor	0.28	Model default value for no primary settling
Peak OURtd	1.34	Calculated from Eqn 3.1

3.4 Capacity Estimation Inputs

The data requirements for capacity estimation were summarized in Section 2.2.2 and the capacity estimation window was shown in Figure 2.11. This section describes the capacity estimation inputs used in the Umhlanga exercise and how they were obtained. Gaps in the available information and their impact on the results are also discussed.

3.4.1 Plant Configuration

The top left panel consists of a menu of options for describing the plant configuration. The following options were selected:

- Modified Ludzack-Ettinger (MLE) process (“AS” page)
- Secondary settling tank, but no balancing or primary settling tanks (“Physical” page)
- Sludge wasting from the aerobic reactor (“Other” page)

The model representation of the Umhlanga plant is shown in Figure 3.4. Note that model includes only the AS reactor and secondary clarifier. The ponds and thickening unit are not considered. The MLE configuration was selected because it provided the closest approximation of the Umhlanga configuration of the available options, however, Umhlanga is not actually an MLE system, because the anoxic zone is not in a separate tank to the aerobic zone and there is no pumped recycle from the aerobic to anoxic zones. This complicates the modelling of the nitrogen removal as discussed in Section 3.6.

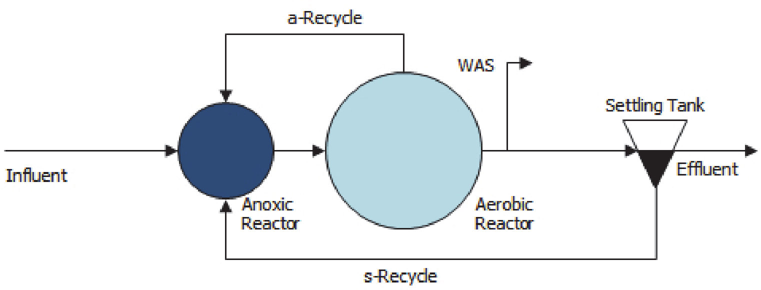


Figure 3.4: Model representation of Umhlanga WWTP in the PWSSD programme

3.4.2 Activated sludge window

Table 3.3 lists the design and operational inputs in the activated sludge (AS) window

Table 3.3: Inputs for the AS window

Reactor volume per module	6856.239 m^3	
Number of modules	1	
Aerobic Xt	3607 mg TSS/L	Table 3.4
Operating sludge age	18.5-25 d	Section 3.4.2.1
Temperature	16°C	Assumed minimum temperature based on temperature data for Phoenix WWTP
Primary anoxic fraction, fx1	0.15-0.7	Section 3.4.2.3
Sludge (s) recycle	≥ 1	Section 3.4.2.2
Maximum TSS wasting (WAS flux)	2 750 kg/d	Max observed sludge wasting rate X MLSS upper operating limit

3.4.2.1 Sludge Age (Rs) and Reactor MLSS (Xt)

The sludge age determines the mass of the sludge in the reactor. Longer Rs will increase the mass of sludge and load to the clarifiers, thus decreasing the capacity of the system calculated in the aerobic Xt and SST area limited scenarios. Sludge age also determines if nitrification occurs. The Rs must be higher than the minimum Rs for nitrification. The minimum Rs is calculated in the Nitri/Denitri window (discussed in Section 3.4.1.3).

When sludge is wasted from the aerobic reactor (hydraulic control), the sludge age (Rs) can be estimated as (neglecting solids leaving with the effluent):

$$Rs = \frac{V_{AS}}{Q_W} \qquad \text{Eqn. 3.2}$$

V_{AS} Total activated sludge reactor volume

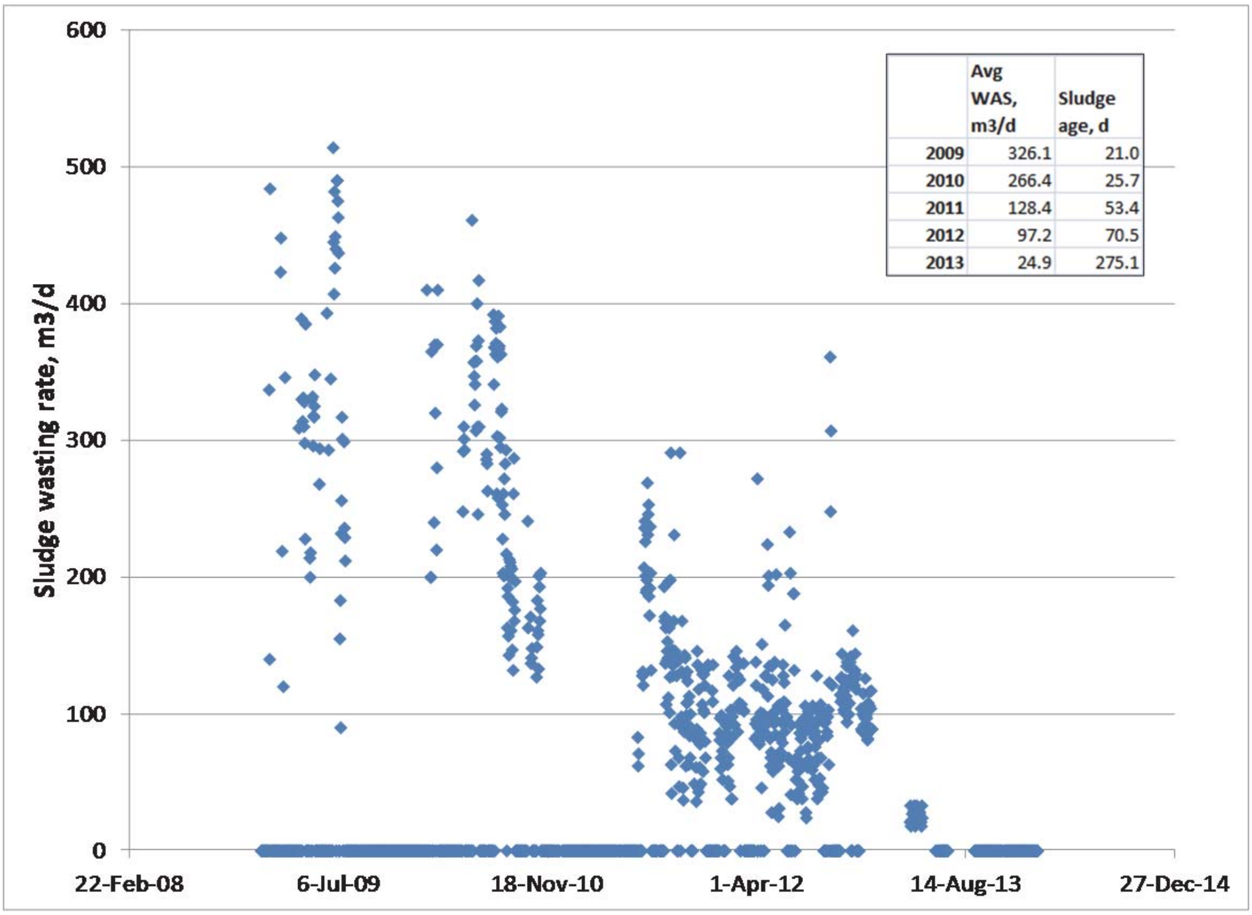
Q_W Waste sludge flow

For the 1996 data, the average daily sludge wasting rate was calculated to be 295 m^3/d based on sludge being wasted an average of 285 min/d at a flow of 62 m^3/h. Given a total reactor volume of 6856 m^3, this yields an average sludge age of 23.3 d which is in the typical range for extended aeration (15- 30 d).

From January 2009 - April 2013 data, non-zero sludge DAF WAS feeds were recorded for 563 out of 1422 days for which operational data was provided. It is not clear what happened on the other 859 days. Possible reasons for shutting down the DAF plant include:

i. Waste sludge was being diverted to Phoenix WWTP
ii. One or more of the AS channels were being reseeded after maintenance or an upset
iii. The operators were struggling to maintain the MLSS in the AS basin

However, it seems unlikely that wasting was suspended 60% of the time during this period and it is assumed that in most cases, it was simply not recorded. The sludge wasting rate appeared to decline over the monitoring period, possibly because the sludge was increasingly being diverted to Phoenix.



	Avg WAS, m3/d	Sludge age, d
2009	326.1	21.0
2010	266.4	25.7
2011	128.4	53.4
2012	97.2	70.5
2013	24.9	275.1

Figure 3.5: Waste sludge feed to DAF

Figure 3.5 shows the recorded wasting rates from 2009 to 2013 as well as the calculated annual averages (based on non-zero values). The average rates for 2009 and 2010 correspond to SRTs which fall in the expected 15-30 days, however, the calculated SRTs for 2011 are excessively long and do not appear to be realistic. There was no sludge wasting data after April 2013 and it is assumed that Umhlanga began diverting all its waste sludge to Phoenix. The difficulty is that for the period selected for the capacity estimation exercise (April 2011 to March 2012), the recorded DAF feed rate had already declined to the point that it was not possible to obtain reliable sludge age estimates and therefore an alternate method of estimating sludge age had to be found.

The reactor MLSS (Xt) calculated by the steady state model is a function of the influent wastewater characteristics and the sludge age. If the wastewater characterisation is adequate and the sludge age is known, then the Xt limited capacity (expressed as ADWF) which corresponds to the observed reactor Xt should equal the observed average flow (ADWF). If there is a significant discrepancy between the measurements and model results, it is generally assumed that there are problems with the waste characterisation. Table 3.4 lists the average ADWF and Xt for the data period used in the capacity estimation exercise and the two years subsequent. Only days in which all four basins were operational were included in the averages.

Table 3.4: ADWF and average Xt for four basins in operation

	ADWF, ML/d	Xt, mg/L
April 2011 - March 2012	6.24	3607
April 2012 - March 2013	6.70	4048
April 2013 - March 2014	5.64	2719

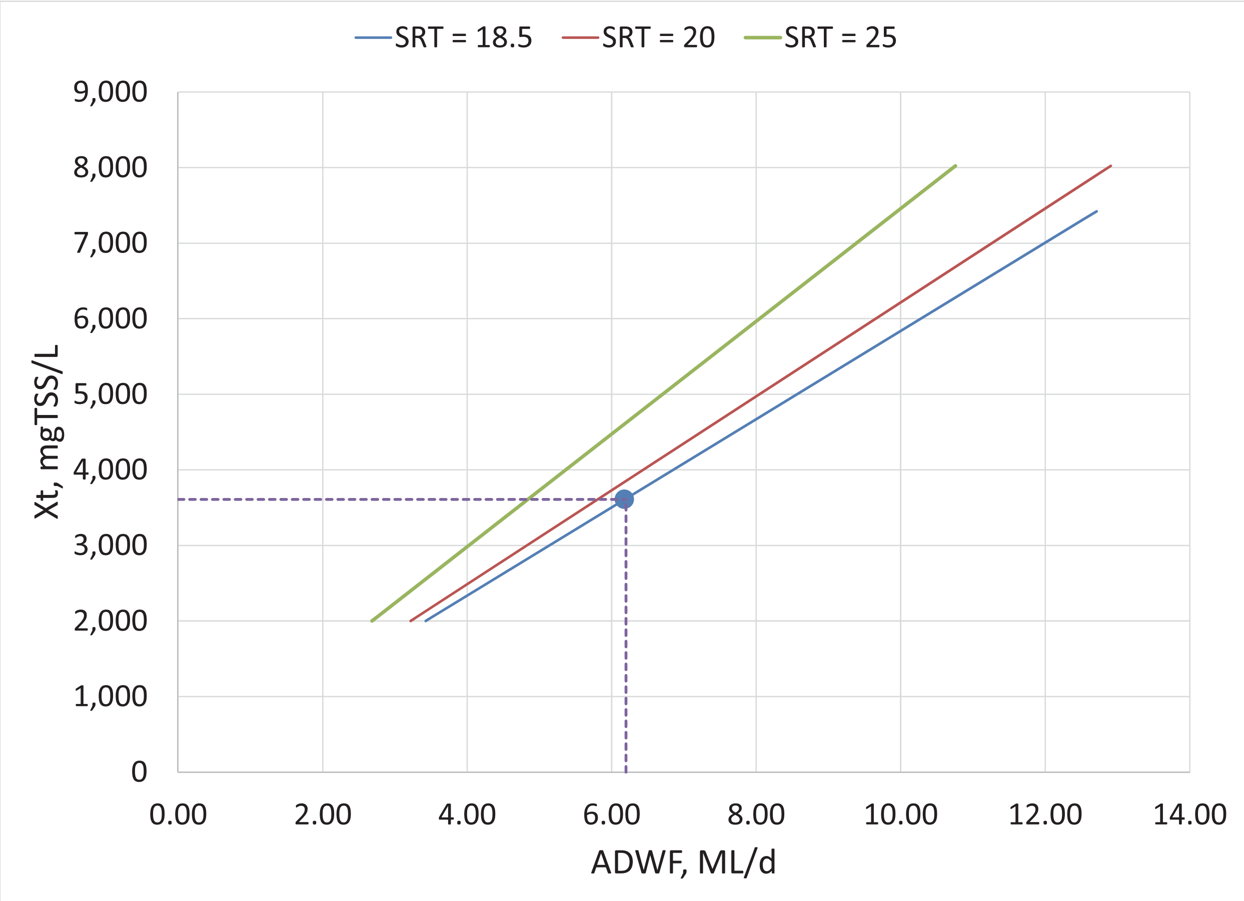


Figure 3.6: Estimating the average sludge age using the steady state model

In this case, since the sludge age was unknown, the Xt limited scenario capacity calculation was used to determine at which sludge age the measured MLSS and ADWF in Table 3.4 matched the model calculations, as shown in Figure 3.6. The steady state model calculated that the observed average MLSS of 3607 mg TSS/L would occur at the observed average flow of 6.2 ML/d at a sludge age of ~ 18.5 d. The effect of increasing the sludge up to 25 days was also investigated as this is the one parameter which affects the overall capacity that the operators have control over.

3.4.2.2 Sludge recycle ratio

The sludge recycle ratio affects nitrogen removal by denitrification which in turn affects oxygen demand and the aeration limited capacity estimate. There is no facility for measuring return sludge flow and the recycle sludge is pumped back to the basins using screw pumps which are well known to be difficult to set accurately and for delivering inconsistent flows. However the sludge recycle rate (s-Recycle) at steady state can theoretically be estimated from the ratio

$$\text{s-Recycle} = \frac{Xt}{(X_{RAS} - Xt)} \qquad \text{Eqn. 3.3}$$

Xt MLSS concentration

X_{RAS} TSS of the return activated sludge

Table 3.5 shows the estimation of the s-Recycle rate for the various available data sets using equation 3.3. The corresponding estimated RAS flow rates are also shown. The averages in Table 3.5 were calculated with all available pairs of Xt and XRAS measurements including for days with less than four basins in operation hence the average Xt and flow values are slightly different to those in Table 3.4.

Table 3.5: Sludge recycle rates estimated from average Xt and X_{RAS}

Data set	Xt, mg TSS/L	X_{RAS}, mg TSS/L	s-Recycle ratio	Average inflow, ML/d	Estimated RAS flow, ML/d
January 2009 - March 2011	3337	6319	1.1	5.93	6.64
April 2011-March 2012	3588	6896	1.1	5.90	6.40
April 2012-March 2013	4050	5992	2.1	6.50	13.6
April 2013-March 2014	3145	4818	1.9	5.46	10.3

Typical sludge recycle ratios for small treatment plants are 0.5 to 1.5 (Tchobanoglous and Burton, 1991, p. 546). The estimated s-recycle ratios for January 2009 - March 2012 fall well within this range however from April 2012 the ratios and estimated sludge recycle flows are higher than expected due to low measured RAS suspended solids concentrations. There is not enough to information to determine whether the higher RAS flows are even possible and whether they may instead be a problem with the RAS samples. This may be something the Umhlanga staff need to investigate further. In the steady state model, the sludge recycle rate only affects the rate of denitrification. The effect of varying the recycle ratio is investigated in Section 3.6.2.

3.4.2.3 Anoxic fraction, fx

The anoxic fraction of the AS basin volume is required to determine the plant's capacity for denitrification as well as the total aerated volume. As discussed in Section 3.2, the AS basins at Umhlanga consist of four parallel channels with three aerators each. At present, the first aerator in each channel is operated at low speed while the last two are operated at high speed. Unfortunately there is no dissolved oxygen (DO) measurement anywhere in the plant and the impact on the overall aeration rate of having the four aerators on low speed is unknown. The preliminary assumption was that the first third of the basin is anoxic due to high COD and low oxygen transfer. However, as discussed in the results section, it appears that the plant is aeration limited and the actual anoxic volume fraction is may be larger than a third. However, there is a splash zone even for the first row of aerators so the anoxic zone could also theoretically be less than a third. Therefore anoxic fractions of 0.15 to 0.7 were investigated.

3.4.3 Nitri/denitri page

The following information was entered in the Nitri/denitri page in the bottom left panel:

Table 3.6: Information required for the nitrogen removal calculations

MuAm20 (μ_{Am20})	0.3-0.5/d	Nitrifier maximum specific growth rate at 20°C
Sf	1.2	Nitrification safety factor, default
a-recycle	≥ 0.1	Recycle from aerobic to anoxic zone. Section 3.4.3.2
DOa	0.5-2 mg O/L	Dissolved oxygen in a-recycle. Default = 2 mg O/L. Section 3.4.3.2
DOs	0.1-1 mg O/L	Dissolved oxygen in s-recycle. Default = 1 mg/L. Section 3.4.3.2

3.4.3.1 Nitrifier maximum specific growth rate

The nitrifier maximum specific growth rate is generally considered to be a characteristic of a particular wastewater rather than a kinetic constant with values ranging from 0.3 to 0.7 being reported in the literature (Ekama and Wentzel, 2008b). However, wastewater specific growth rate data is unlikely to be available in the context of a plant audit so the user will generally select the default value of 0.45/d which is purposefully set at the low end of the range to ensure a conservative estimate of the capacity for nitrification.

As is discussed in Section 3.5 It appears likely that the aeration capacity is insufficient to maintain an average dissolved oxygen concentration of 2 mg/L in the activated sludge reactor and this would tend to decrease the nitrification rate. Therefore the effect of decreasing (μ_{Am20}) to account for oxygen limitations was also investigated.

3.4.3.2 Recycle to the anoxic zone

The Umhlanga plant does not have a recycle line between the aerobic and anoxic zones, however, the Capacity Estimation tool does not allow it to be set to zero because non-zero values are required for the denitrification calculations. Furthermore, the mixing action induced by the aerators will result in some internal recycling between anoxic and aerobic regions. The a-recycle ratio was initially set at 0.1 and then increased to determine its impact on denitrification. The calculation of the denitrification capacity also requires the dissolved oxygen in the a- and s-recycles to be specified. Values of DOa = 0.5-2 mg O/L and Dos = 0.1-1 mg O/L were investigated but were not found to have any impact on the results in this case because the system appeared to be underloaded in terms of denitrification capacity (Equation 3.11 in Section 3.6 Table 3.12b).

3.4.4 SST Window

The secondary settler calculations require information on the SST design, peak flows and sludge settleability as listed in Table 3.7.

Table 3.7: Information required for SST capacity calculation

SST area per SST	397.6 m^2	
Number of SSTs	2	
SVI	100-200 ml/L	Section 3.4.4.1
SST Flux rating	0.8	Default
PWWF factor =PWDF factor w.r.t ADWF	1.73	See Table 3.2

3.4.4.1 SVI

Sludge volume index (SVI) is one of the measures of the settleability of the activated sludge solids which is used in the SST model. SVI data was available for the 1996 data set but not for the 2009-2014 data set (30 minute settling tests are still carried out periodically but the results are not captured electronically). The average SVI for lanes A, B, and C in the 1996 data set was 157 ml/L (range 44-227). The average for lane D was 290 mL/L (reported range 87-1854) which appears to be skewed by the high maximum value (which is clearly an error since SVI cannot be greater than 1000 mL/L) and was therefore excluded. Clarifier performance appeared to be better in 2011 to 2014 than in 1996 (average ~ 25 NTU in 1996 compared to < 11 NTU in 2011 to 2014) therefore the later SVIs were probably < 157 ml/L. The effect of SVI values from 100-200 ml/L was investigated. Note that SVI values > 200 ml/L do not accurately reflect sludge settleability and should not be used.

3.4.5 Aeration window

The following information is required to calculate the aeration capacity of the plant:

Table 3.8: Information required for aeration capacity calculation

Site altitude	50 m	Assumed
Alpha factor	0.9	Correction factor accounting for impact of impurities on KLa. Recommended range 0.8-0.9. Higher value selected because long sludge ages in extended aeration tend to improve aeration efficiency (Stenstrom and Rosso, 2008)
Beta factor	0.9	Default value. Correction factor for the effect of impurities on solubility of oxygen.
CL	2 mg/L	DO concentration in reactor. Default is 2 mg/L
Standard oxygen transfer rate, R_Std	1.2 kg O_2/kWh	Typical range for high rate aerators is 0.9-1.3 (Stenstrom and Rosso, 2008) R_std corresponds to overall transfer efficiency of ~ 0.8
Line to shaft efficiency	0.8	Default
Total power supply of aerators per AS module	220-264 W	Section 3.4.5.1

3.4.5.1 Total aerator power

The Umhlanga AS basins are equipped with twelve 22 kW surface aerators for a total aerator power of 264 kW. However, as discussed previously, four of the aerators are run at low speed, the impact of which on both the aeration capacity and power consumption is currently not clear. Assuming the first row of aerators run at half power, the total power available is 220 kW and this was used as the initial input for the aeration limited capacity calculation.

Note that the aeration capacity is also directly proportional to the R_Std and line power. Therefore if the aeration capacity is limiting, it is important that these values are known as accurately as possible. In some cases, they can be obtained from the equipment specifications if available.

3.4.6 Reference case inputs

Table 3.9 lists the initial inputs used in the capacity estimation exercise. The effect of varying the listed parameters on the capacity estimates was subsequently investigated and the results compared to the reference case.

Table 3.9: Reference Case Inputs for April 2011 - March 2012

Aerobic Xt	3607 mg TSS/L	Average for April 2011 - March 2012
Operating sludge age	18.5 d	Adjusted to match measured and predicted ADWF for Xt limited scenario
Primary anoxic fraction, fx1	0.33	
Sludge (s) recycle	1.1	From Table 3.5
a-recycle	0.1	
DOa = CL	2 mg O/L	Default
DOs	1 mg O/L	Default
SVI	157 ml/L	Average for 1996
Aerator power supply	220 kW	Four of twelve aerators operating at half power

3.5 Capacity Estimation Results

3.5.1 Preliminary reference case results

The capacity estimates for the reference case are shown in Table 3.10. Note that each limiting scenario corresponds to the solution of the steady state model for the particular set of inputs and the given constraint as discussed in Section 2.2.2. The results for the SST area limiting scenario is the solution of the steady state model in which the required SST area at the PWWF is the actual SST area. The clarifiers should be able to handle peak flows of up to 11.082 ML/d and aerobic Xt up to 3 536.9 mg/L in this scenario. The aerobic Xt results correspond to the solution in which the aerobic Xt is equal to the user specified value, i.e. 3607 mg TSS/L. The aeration limited output is the model solution in which the calculated aeration requirement at the daily peak oxygen utilization rate, OURtd,pk, equals the actual power available, which in this case is assumed to be 220 kW.

Table 3.10: Reference case output April 2011 - March 2012

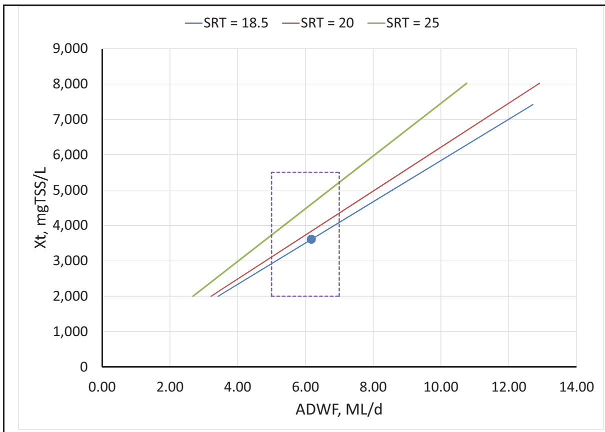
	SST area limited	Xt limited	Aeration limited	WAS rate limited
ADWF (Qi_AD), Ml/d	7.34	6.18	4.84	12.71
PWDF (Qi_PD), Ml/d	12.70	10.69	8.38	21.99
PWWF (Qi_PW), Ml/d	12.70	10.69	8.38	21.99
MLSS (Xt), mg TSS/l	4284	3607	2827	7420
Total SST area required (at PWWF), m^2	795.2	494.6	273.4	5,595.3
Oxygen utilization rate (OURtd), mg O_2/L/h	38.45	32.37	25.37	66.59
Peak oxygen utilization rate, (OURtd,pk), mg O_2/L/h	46.70	39.32	30.81	80.88
Required aeration capacity, kW	333.5	280.7	220.0	577.6

From Table 3.10, it appears that the Umhlanga WWTP is ultimately aeration capacity limited (maximum ADWF = 4.45 ML/d, PWWF = 7.69 ML/d). Moreover it appears that aeration capacity is insufficient to meet the current peak and average OURtd (the average OURtd at 6.2 ML/d is slightly higher than the peak OURtd at the aeration limited capacity). Table 3.10 also shows that, with this set of conditions, aeration capacity would still be limiting if all three aerators were operated at full power (264 kW available compared to 307 kW required at ADWF = 6.2 ML/d).

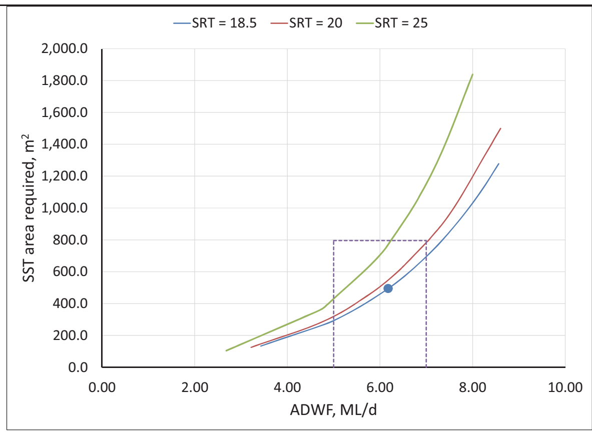
3.5.2 Effect of sludge age

The reactor MLSS is strongly dependent on the sludge age and therefore so are the MLSS, SST and WAS limited capacities. Sludge age is also an important aspect of the plant operation that the operators have control over so strictly speaking the plant's capacity should also be defined in terms of a range of feasible sludge ages. The effect of varying the sludge age from 18.5 to 25 days on the reactor MLSS, required SST area, required aeration power and WAS solids flux is summarized in Figure 3.6.

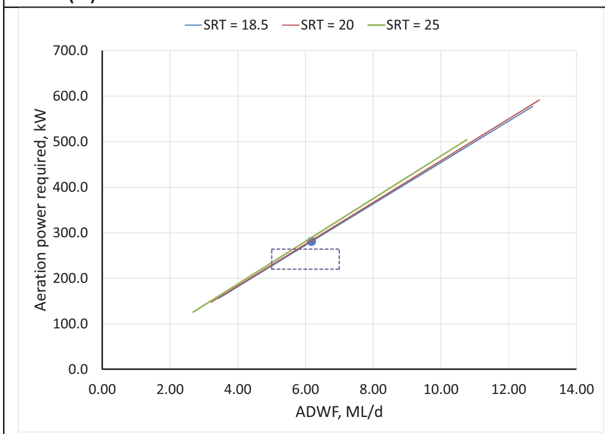
Increasing the sludge age increases the MLSS (X_t) at a given flowrate since solids are being retained in the system for longer. Therefore, the flowrate at which a given MLSS occurs (MLSS limited scenario) decreases. Similarly the solids loading to the SSTs increases, reducing the flow which a given SST can handle (SST area limited). Overall sludge production decreases with increasing sludge age so the WAS limited capacity actually increases. The aeration limited capacity also decreases with increasing sludge age but the effect is small because the aeration requirement is primarily determined by the raw wastewater characteristics.



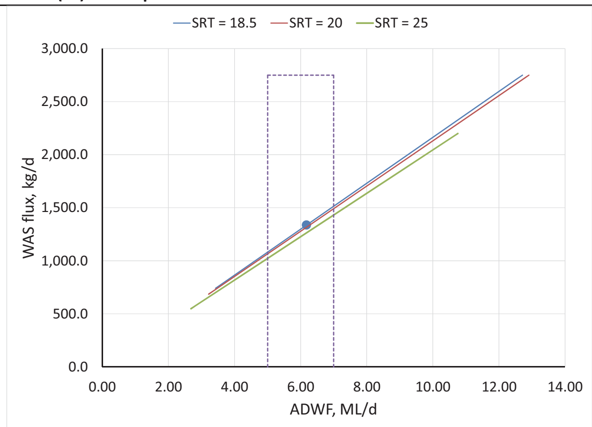
(a) Reactor Xt



(b) Required SST area



(c) Required aeration power

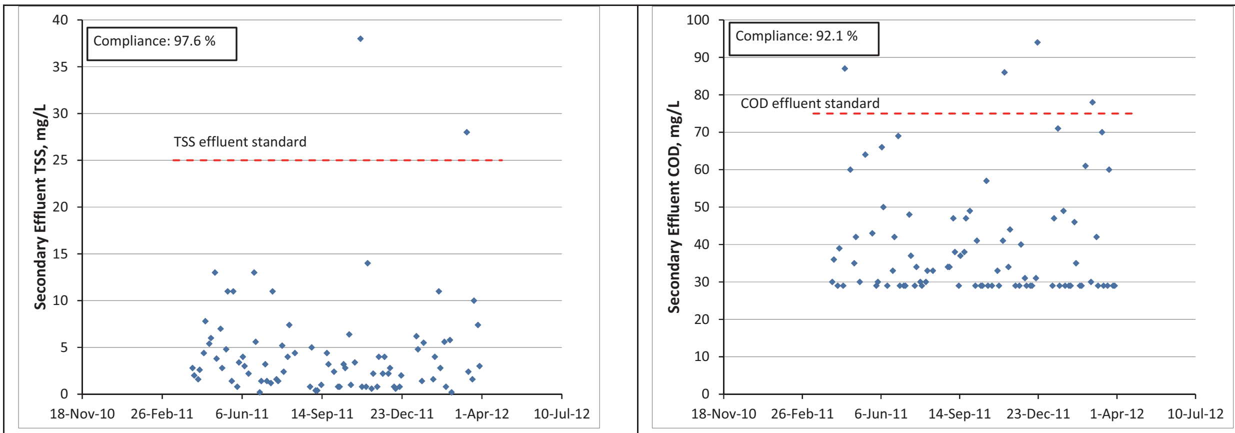


(d) WAS flux

Figure 3.7: Capacity estimation summary charts

3.5.3 Effect of SVI

The clarifiers' capacity and performance is strongly dependent on the settlability of the activated sludge solids. The SVI measurements used were from 1996 when the clarifiers appeared to be performing less well than in 2011-2012 therefore the average SVI was probably less than the 157 ml/L used in the initial set up. Figure 3.8 shows the effluent COD and TSS for April 2011 to March 2012, which would have been primarily determined by SST performance.



(a) Secondary effluent TSS (b) Secondary effluent COD

Figure 3.8: Clarifier performance April 2011 - March 2012

The default value in PWSSD is DSVI~SVI = 100 ml/L. Table 3.11 shows the effect of varying SVI from 100-200 ml/L on the SST area limited capacity.

Table 3.11: Effect of sludge settleability on SST area limited capacity (PWWF factor = 1.73)

	SVI = 100 ml/L	SVI = 157 ml/L	SVI = 200 ml/L
ADWF (Qi_AD), Ml/d	9.83	**7.34**	5.95
PWWF (Qi_PW), Ml/d	17.01	12.70	10.29
MLSS (Xt), mg TSS/l	5739	4284	3472

The SST limited capacity increases to 9.83 ML/d (34% increase) for SVI = 100 ml/L or decreases to 5.95 ML/d (19% decrease) for 200 ml/L. SVI or some other measure of sludge settleability is therefore potentially a critical parameter in establishing the plant capacity. Since this data was available onsite, it would have been available in an actual plant audit but since SST performance did not seem to be limiting in this case (based on both the capacity estimates and Figure 3.8), the matter was not pursued any further.

3.5.4 Factors affecting the aeration limited capacity

In this exercise, the aeration system appeared to be the factor limiting the overall plant capacity. However, the caveat is that it was calculated based almost entirely on estimates, literature and default values. In Table 3.7, the aeration capacity is directly proportional to the standard oxygen transfer rate, line to shaft efficiency and total aeration power. During an audit it should be possible to obtain more reliable estimates of these parameters from the equipment product literature, which can usually be found online if necessary.

The aeration capacity is also inversely proportional to the peak OURtd factor which was calculated using the default peak TOD factor. Figure 3.9 shows the relationship between the peak TOD amplitude factor and peak OURtd factor.

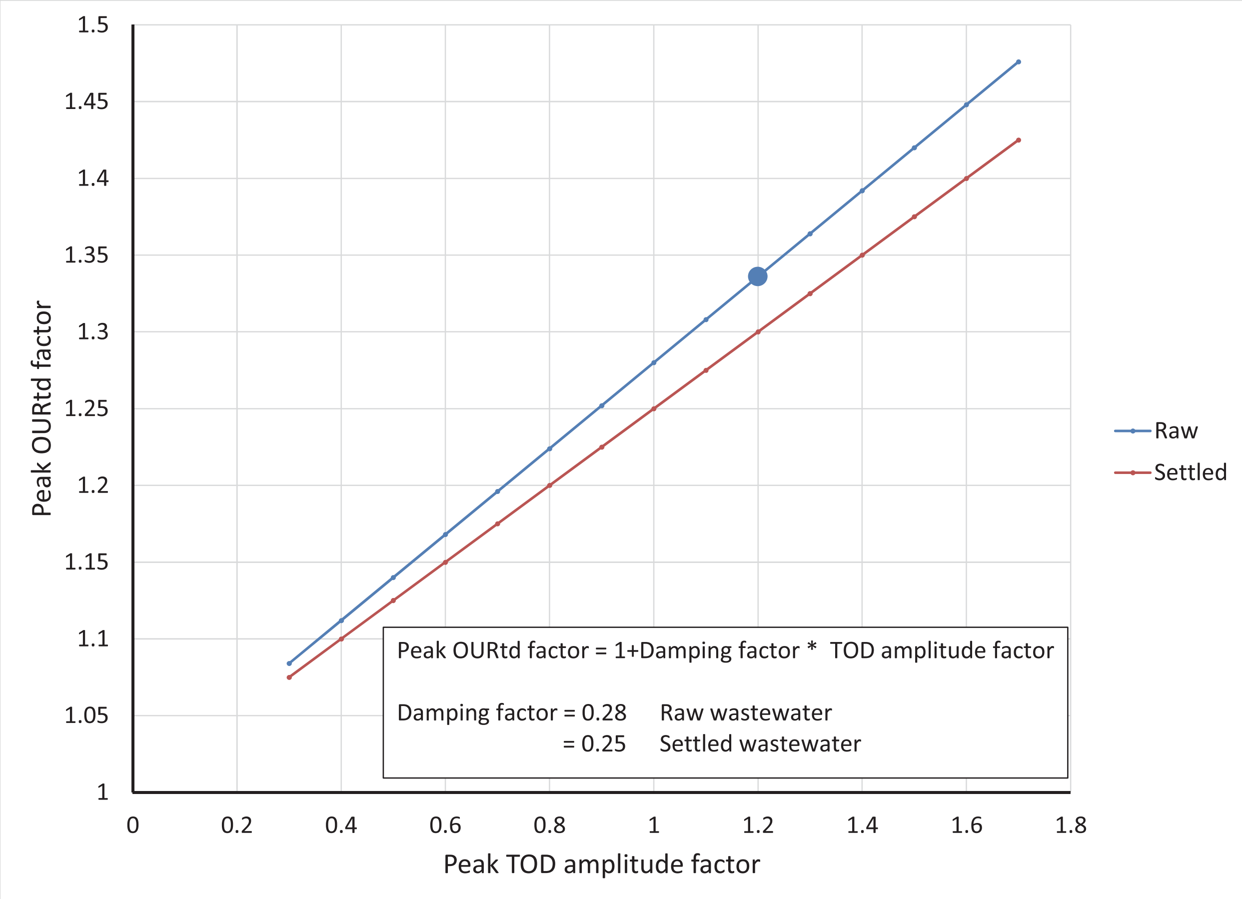


Figure 3.9: Peak OUR factor as a function of peak TOD amplitude factor

The default TOD amplitude factor used in this example was 1.2 which is set high to ensure a conservative estimate of the aeration capacity and corresponds to an OURtd peak factor of 1.34. In comparison, the TOD amplitude factors for the eight preloaded wastewater profiles currently provided with PWSSD range from 0.4 to 1.7. These correspond to raw wastewater OURtd peak factors ranging from 1.11 to 1.48 or a 20% range in overall aeration limited capacity.

In practice, the diurnal variation profiles required to calculate the peak TOD are unlikely to be available in a plant audit and are expensive and time consuming to measure. However, dissolved oxygen measurements in the activated sludge basins can provide an indication of whether the aeration capacity is sufficient or not. Umhlanga WWTP unfortunately does not have dissolved oxygen measurements at present however they should be available at most plants since they are an important tool for plant control. Checks on dissolved oxygen levels can also easily be carried out during onsite visits using field testing equipment.

3.6 Nitrification/denitrification

In addition to the capacity estimates for the various limiting scenarios, the steady state model also calculates the plant nitrification and denitrification capacities, effluent TKN, FSA and nitrate concentrations and % nitrogen removal. Table 3.12 lists the effluent concentrations and denitrification state for the reference case (Tables 3.8 and 3.9). PWSSD also calculated phosphorous removal but since effluent phosphate is not measured at Umhlanga, these results will not be discussed.

Table 3.12: Effluent concentrations and removal efficiencies

Effluent COD (Suse)	27.2 mg COD/L
Effluent TKN (Nte)	1.20 mg N/L
Effluent FSA (Nae)	0.57 mg N/L
Effluent Nitrate (Nne)	15.27 mg N/L
Effluent TP (Pae)	3.96 mg P/L
AX1 Nitrification/Denitrification	Underloaded
%N Removal	67.1
%P Removal	48.7

Figure 3.10 shows the model nitrification/denitrification results as a function of sludge age and recycle to the anoxic zone (a+s) and compares them to the measured effluent ammonia and nitrate. The measured effluent results are summarized as box and whisker plots which show the minimum, maximum, median and 25th and 75th percentiles for each parameter. The nitrification capacity (effluent nitrate concentration which would be observed in the absence of denitrification) is also shown as are the effluent standards for ammonia and nitrate.

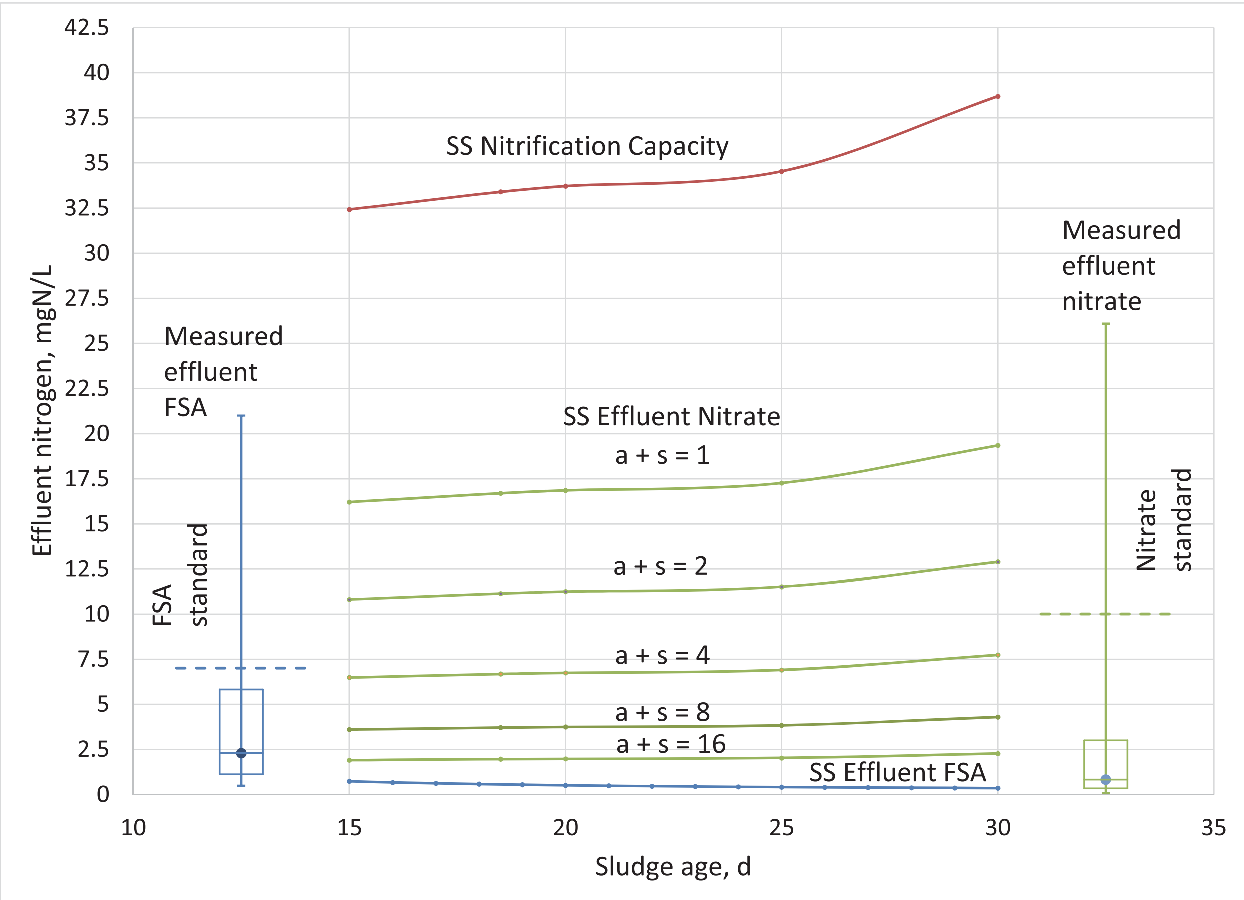


Figure 3.10: Nitrogen removal summary

In the absence of nitrification, the model calculates the effluent ammonia would be ~ 50 mg N/L therefore based on the measured effluent results shown in Figure 3.10, both nitrification and denitrification must be occurring. However, the model appears to under predict the effluent ammonia and over predict the effluent nitrate except at very high combined recycle ratios.

As discussed in Section 2.2.2.5, it is important to understand the steady state model assumptions and limitations in interpreting the results. A comprehensive treatment of nitrification/denitrification and the development of the steady state model equations is provided in Ekama and Wentzel (2008b). The equations used by PWSSD to calculate the effluent nitrogen concentrations are listed in Table 3.13.

Table 3.13a: Nitrification model equations for the Modified Ludzack-Ettinger system

Minimum sludge age for nitrification	
$$Rs,min = \frac{1}{\left(\mu_{AmT}/S_f\right)(1-f_{xt}) - b_{AT}}$$	Eqn. 3.4
Effluent ammonia for no nitrification, Rs < Rs,min	
$$Nae = N_{ti} - N_s - N_{ousi}$$	Eqn. 3.5
Nitrogen content of sludge	
$$Ns = \frac{f_n MX_v}{Q_i R_S}$$	Eqn. 3.6
Effluent ammonia for complete nitrification Rs >Rs,min	
$$Nae = \frac{K_{nT}\left(b_{AT} + {}^{1}/_{R_S}\right)}{\mu_{AmT}(1-f_{xt}) - \left(b_{AT} + {}^{1}/_{R_S}\right)}$$	Eqn. 3.7
Effluent TKN	
$$N_{te} = N_{ae} + N_{ousi}$$	Eqn. 3.8
Nitrification capacity = effluent nitrate in the absence of denitrification	
$$Nne = N_c = N_{ti} - N_s - N_{te}$$	Eqn. 3.9
Rs = sludge age, d μ_{AmT} = maximum nitrifier growth rate at temperature T S_f = nitrification safety factor f_{xt} = primary anoxic fraction b_{AT} = nitrifier endogenous respiration rate, 1/d Nae = effluent ammonia concentration, mg N/L N_{ti} = influent TKN, mg N/L N_s = nitrogen content of sludge, mg N/L N_{ousi} = soluble unbiodegradable organic influent nitrogen, mg N/L f_n = nitrogen content of VSS mg N/mg VSS MX_v = mass of volatile suspended solids in activated sludge reactor, kg Q_i = Influent flow, ML/d K_{nT} = nitrifiers half saturation constant at temperature at T, mg N/L N_{te} = effluent TKN Nne = effluent nitrate concentration, mg N/L N_c = nitrification capacity	

Table 3.13b: Denitrification model equations for the Modified Ludzack-Ettinger system

Primary anoxic denitrification potential	
$$D_{p1} = S_{bi}\left[f_{Sb's}\frac{(1-f_{cv}Y_{Hv})}{2.86}+\frac{K_2 f_{xt} Y_{Hv} R_s}{1+b_H R_s}\right]$$	Eqn. 3.9
Effluent nitrate concentration for overloaded denitrification system	
$$N_{ne} = N_c + \frac{aO_a}{2.86}+\frac{sO_s}{2.86}-D_{p1}$$	Eqn. 3.10
Effluent nitrate concentration for underloaded denitrification system	
$$Nne = \frac{N_c}{a+s+1}$$	Eqn. 3.11
D_{p1} = primary anoxic denitrification potential, mg N/L S_{bi} = Influent biodegradable COD, mg COD/L $f_{Sb's}$ = readily biodegradable COD fraction with respect to S_{bi} f_{cv} = COD to VSS ratio of sludge, mg COD/mg VSS Y_{Hv} = Yield of OHOs (ordinary heterotrophic organisms) in terms of biomass, mg VSS/mg COD K_2 = specific rate of denitrification in the primary anoxic reactor due to degradation of slowly biodegradable (particulate) COD, mg NO_3-N/mg OHOVSS/d b_H = OHO endogenous respiration rate, 1/d a = MLSS recycle ratio from aerobic to anoxic reactor s = return activated sludge recycle ratio O_a = dissolved oxygen in a-recycle, mg O/L O_s = dissolved oxygen in s-recycle, mg O/L	

3.6.1 Nitrification and effluent ammonia

From Eqn. 3.7, the effluent ammonia is essentially independent of the influent nitrogen load under steady state conditions, however, in reality, the slow growing nitrifiers do not adapt well to rapidly varying loads. As a result, not only does the effluent FSA fluctuate due to normal diurnal variation, but the average nitrification efficiency is known to drop under cyclical flow and load conditions compared to steady state conditions (Ekama and Wentzel, 2008b). Consequently, even when there is good data for the model inputs, the average effluent concentration observed at a plant will generally be higher than that predicted in Equation 3.7. Furthermore, this effect is most pronounced at low Rs to Rs,min ratios and can be mitigated to some extent by operating at long sludge ages.

It is therefore important to understand that PWSSD is not designed to produce accurate predictions of the average effluent ammonia and the user should not attempt to adjust the model to fit the measurements. Instead, the analysis of the nitrification results should focus on how close the plant is operating to Rs,min. Rs,min is the minimum sludge age at which a nitrifier population can be maintained in the activated sludge system. At lower sludge ages, the nitrifier growth rate cannot keep up with the rate of wasting and the nitrifiers wash out. From Equation 3.4, Rs,min depends on the maximum nitrifier growth rate μ_{AmT}, endogenous respiration rate b_{AT} and aerobic fraction $(1-f_{xt})$ (since nitrifiers are obligate aerobes). μ_{AmT} and hence Rs,min are strongly dependent on temperature and so Rs,min is always calculated at the minimum expected temperature in the Nitrification/Denitrification window.

The nitrification safety factor Sf is a design parameter, specified in the aeration window (Section 3.4.3), which is introduced to ensure that the minimum sludge age the plant will be operated at is always higher than Rs,min. During design, the value of Sf should be selected to reflect (Ekama, 2015, Personal Communication)

1) Uncertainty in the maximum nitrifier growth rate
2) The desired degree of damping of *Nae* in response to varying influent nitrogen loads

In this particular example, neither the actual sludge ages, μ_{Am20} (which tends to be wastewater specific), $(1 - f_{xt})$ nor the original design Sf are known with any certainty which makes interpretation of the effluent ammonia results difficult.

The overall compliance with the effluent ammonia standard is only 75.6% which suggests the plant could be operating too close to Rs,min. However, one would expect this to be primarily an issue during the colder months and from Figure 3.11, there is no evidence that ammonia removal is worse during winter.

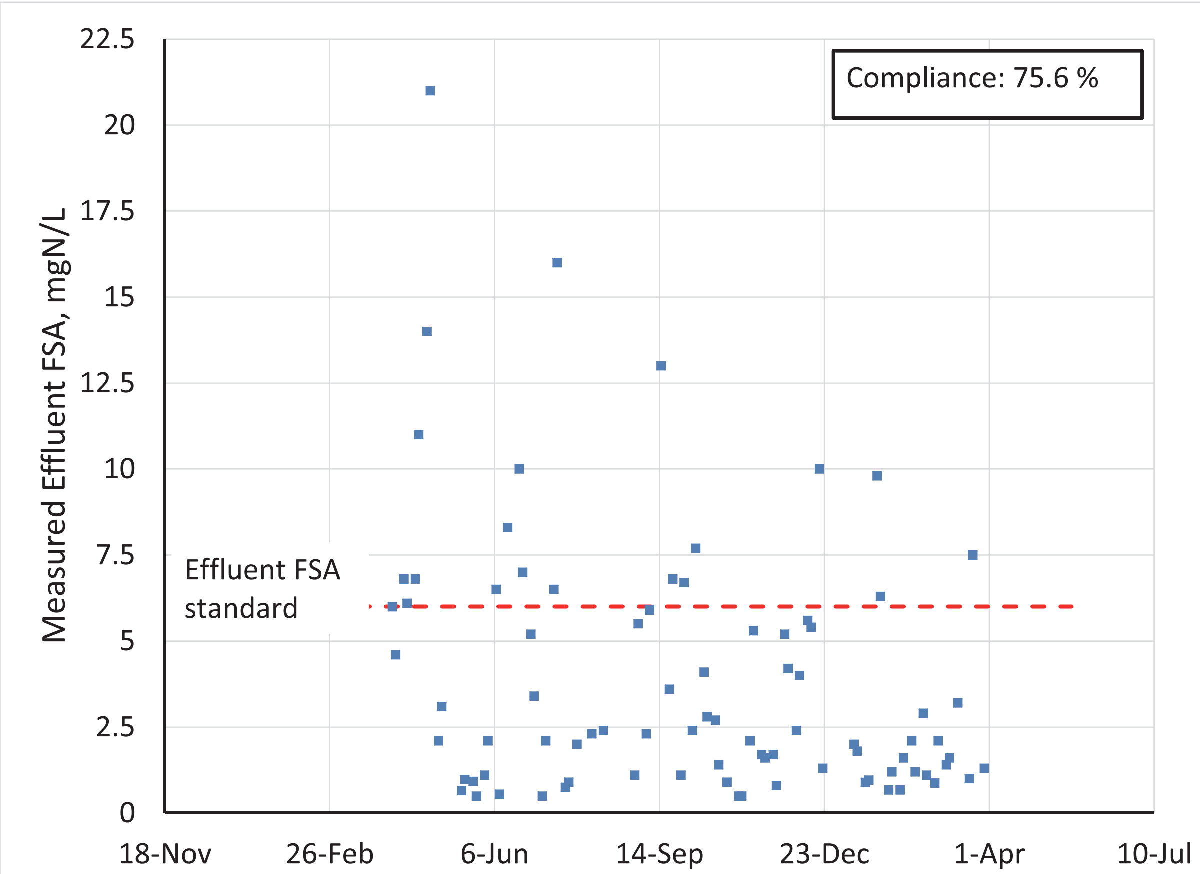


Figure 3.11: Effluent ammonia measurements for April 2011 - March 2012

In Section 3.5, the capacity estimation results indicated that the aeration capacity of the plant may be insufficient to meet peak oxygen demand and this could have an impact on nitrification efficiency. The steady state model does not calculate the actual oxygen concentrations in the reactor and nitrification model (Equation 3.7) assumes that dissolved oxygen is not limiting.

However, the user can make adjustments to the values of μ_{AmT20} and f_{xt} to investigate the potential effects of insufficient aeration. μ_{AmT20} is insensitive to high dissolved oxygen levels however reduced nitrification rates are observed at low dissolved oxygen concentrations (Ekama and Wentzel, 2008b). Insufficient oxygen transfer would presumably also tend to result in larger values of f_{xt} and lower rates of nitrification. Since the effects of the two parameters on the nitrification rate could not be separated, the sensitivity of Rs,min and effluent ammonia on the lumped parameter $\mu_{AmT}(1 - f_{xt})$ was investigated. The values of $\mu_{AmT}(1 - f_{xt})$ for μ_{AmT} = 0.3-0. 5 and f_{xt} = 0.15-0.7 at 16°C are listed in Table 3.14 assuming a minimum wastewater temperature of 16°C. The reference case assumed operating point is shown in bold.

Table 3.14: Values of $\mu_{AmT}(1-f_{xt})$ at 16°C

μ_{Am20}	μ_{AmT} \ f_{xt}	0.15	0.3	0.33	0.4	0.45	0.5	0.55	0.6	0.65	0.7
0.3	0.189	0.160	0.132	0.126	0.113	0.104	0.094	0.085	0.075	0.066	0.057
0.35	0.220	0.187	0.154	0.147	0.132	0.121	0.110	0.099	0.088	0.077	0.066
0.4	0.252	0.214	0.176	0.169	0.151	0.138	0.126	0.113	0.101	0.088	0.075
0.45	0.283	0.240	0.198	**0.190**	0.170	0.156	0.141	0.127	0.113	0.099	0.085
0.5	0.314	0.267	0.220	0.211	0.189	0.173	0.157	0.141	0.126	0.110	0.094

Figure 3.12 shows Rs,min as a function of $\mu_{AmT}(1-f_{xt})$ at 16°C and 22°C for Sf values of 1 (no margin of safety) and 1.2 (default value).

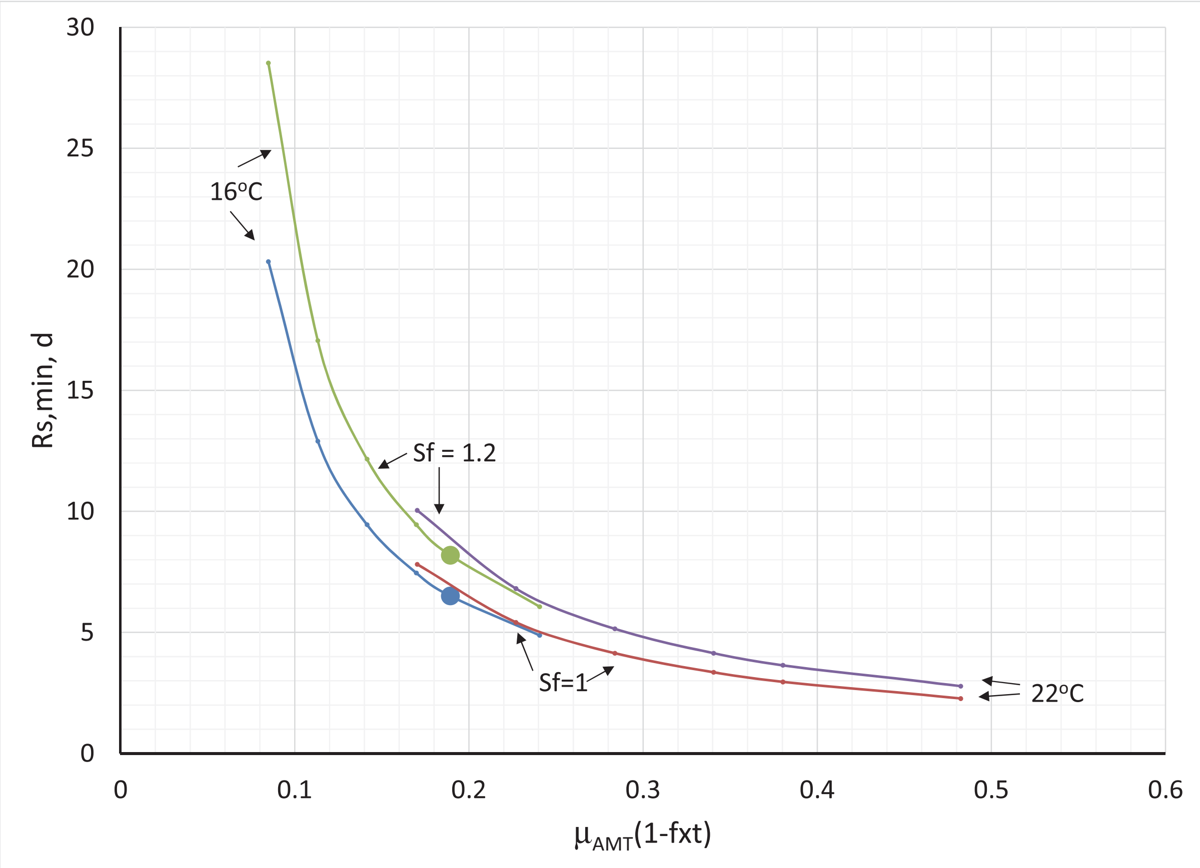
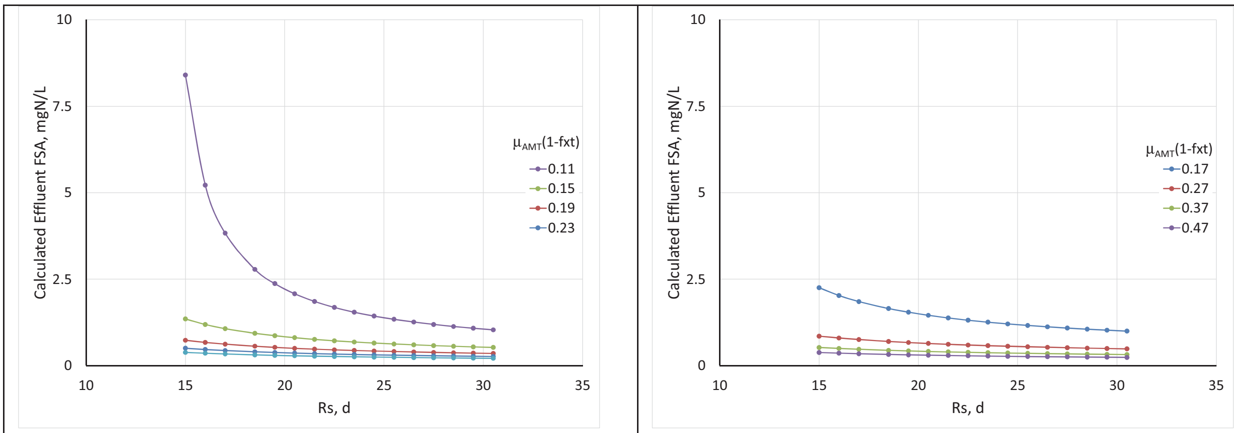


Figure 3.12: Rs,min as a function of temperature and $\mu_{AmT}(1-f_{xt})$

In Figure 3.12, the curves for 16°C and 22°C correspond to the same range of values of $\mu_{Am20}(1-f_{xt})$ however μ_{AmT} increases and hence Rs,min decreases with temperature. From Figure 3.12, Rs,min < 18.5 d requires $\mu_{AmT}(1-f_{xt})$ > 0.11 therefore only values of $\mu_{AmT}(1-f_{xt})$ > 1.1 are considered feasible. The feasible region is indicated by the shaded blocks in Table 3.13. Figure 3.13 shows the steady state effluent ammonia (Equation 3.7) as a function of Rs and $\mu_{AmT}(1-f_{xt})$ at 16°C and 22°C.



(a) Steady state effluent ammonia at 16°C

(b) Steady state effluent ammonia at 22°C

Figure 3.13: Steady state effluent ammonia as a function of sludge age

From Figures 13.3(a) and (b), steady state effluent ammonia is not very sensitive to either Rs or $\mu_{AmT}(1 - f_{xt})$ except for Rs close to Rs,min where nitrification efficiency decreases. Therefore it is assumed that the discrepancy between the predicted and measured ammonia values are primarily due to diurnal variations in influent TKN load or process disruptions not accounted for in the model. Nevertheless lower values of $\mu_{AmT}(1 - f_{xt})$ due to insufficient aeration capacity would correspond to a less robust nitrifier population which would be more sensitive to load variations and upsets. However, without a much more detailed analysis as well as reliable sludge age and dissolved oxygen data it is not possible to conclusively link the frequent noncompliance in the effluent ammonia to the aeration capacity and at best one can say that these issues should be investigated further to ensure more consistent nitrification.

3.6.2 Denitrification and effluent nitrate

Denitrification occurs when nitrate generated in the aerobic zone is recycled to the anoxic zone. In PWSSD, the assumption is that the anoxic and aerobic zones are separate completely mixed reactors and there is a pumped MLSS recycle line (a-recycle) between them in addition to the RAS recycle line (s-recycle) as shown in Figure 3.4. In the case of Umhlanga, however, the anoxic and aerobic zones are just different regions in the same basin and the only pumped recycle is the RAS which was estimated to be ~ 1.1 times the inflow.

From Figure 3.10, the steady state predicted effluent nitrate at a combined recycle ratio of ~ 1 is ~ 17 mg N/L which is much higher than the nitrate measurements, 92.5% of which comply with the 10 mg N/L effluent standard as shown in Figure 3.14. Therefore, there must be a significant amount of exchange of nitrate between the anoxic and aerobic zones due to internal mixing and assuming a~0 is obviously not valid. However, it is not clear what a reasonable value for the a-recycle would be in this situation.

Surface aerators induce a substantial amount of vertical mixing with high DO levels at the surface being depleted as the fluid is sucked down towards the bottom of the basin before being drawn up through the impeller again. This may produce an equivalent effect to a pumped recycle to a separate anoxic tank depending the oxygen levels near the bottom of the basin. However, the effect of rapidly cycling between aerobic and anoxic on the efficiency of denitrification is not known. Furthermore, if this effect was generally sufficient to ensure consistent denitrification then presumably there would have been no need for the development of the MLE process. From Figure 3.10, an effective combined recycle ratio in excess of 16 would be required to achieve the observed measured nitrates in the secondary effluent and this seems unrealistic.

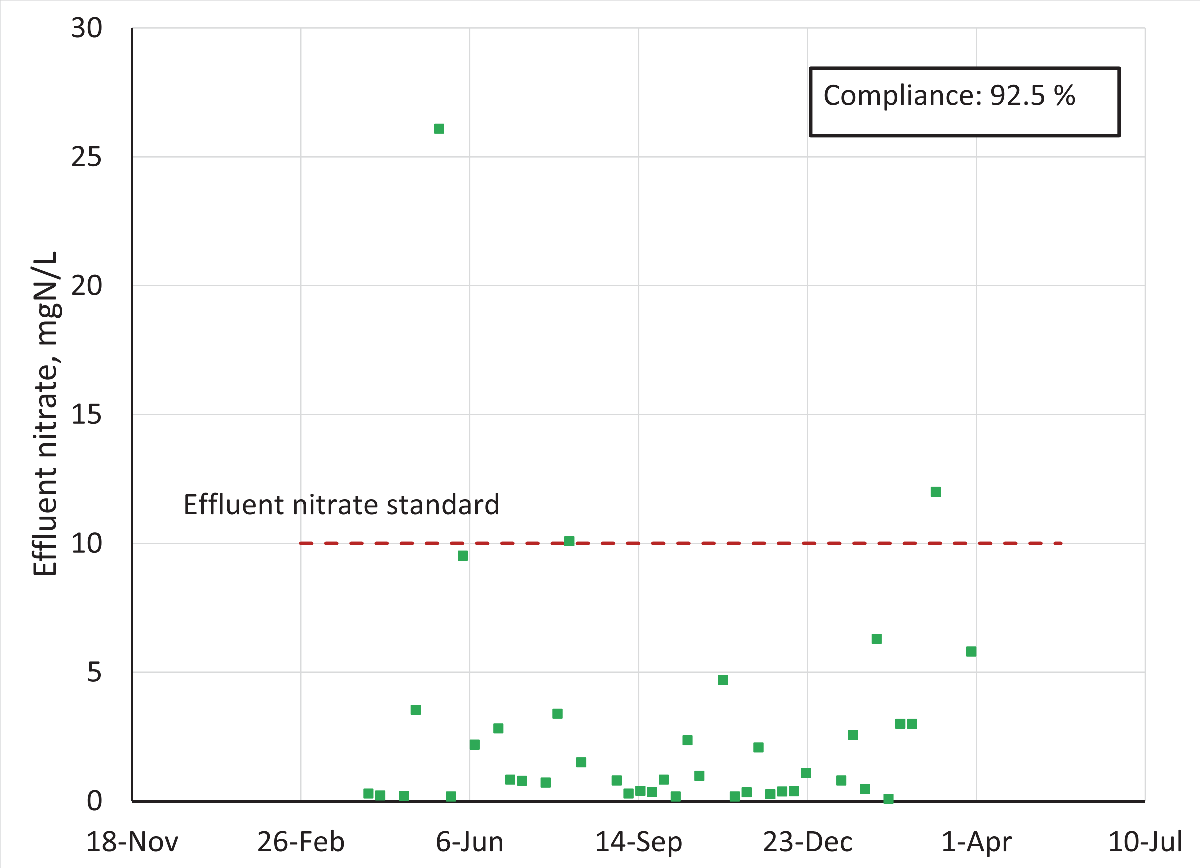


Figure 3.14: Effluent nitrate

Other factors which could contribute to the discrepancy between measured and predicted nitrates include:

1. Inaccurate characterisation of the wastewater resulting in the overestimation of the nitrification capacity in Equation 3.9. The values of Nti and Nousi were calculated using the probabilistic fractionator
2. Some denitrification does occur in the secondary settlers. However, excessive denitrification the settlers tends to cause rising sludge problems and there is no evidence of this in the effluent TSS data.
3. Improper handling of the samples allowing denitrification to continue between collection and analysis.

While these issues warrant further investigation, it should be pointed out that the plant's overall compliance depends on the pond effluent concentration not the secondary effluent and further denitrification may also occur in the ponds.

3.6.3 Effect of denitrification on aeration limited capacity

Denitrification not only reduces effluent nitrate but also reduces the overall oxygen demand of the activated sludge system since influent COD utilization continues under anoxic conditions using NO_3 as the electron acceptor, reducing the of the oxygen demand by 2.86 mg O/L per 1 g N/L of NO_3 denitrified (Ekama and Wentzel, 2008b). Figures 3.15 show the increase in aeration limited capacity and the reduction in both peak and average aeration power required as the combined recycle (a+s) recycle and hence denitrification increase.

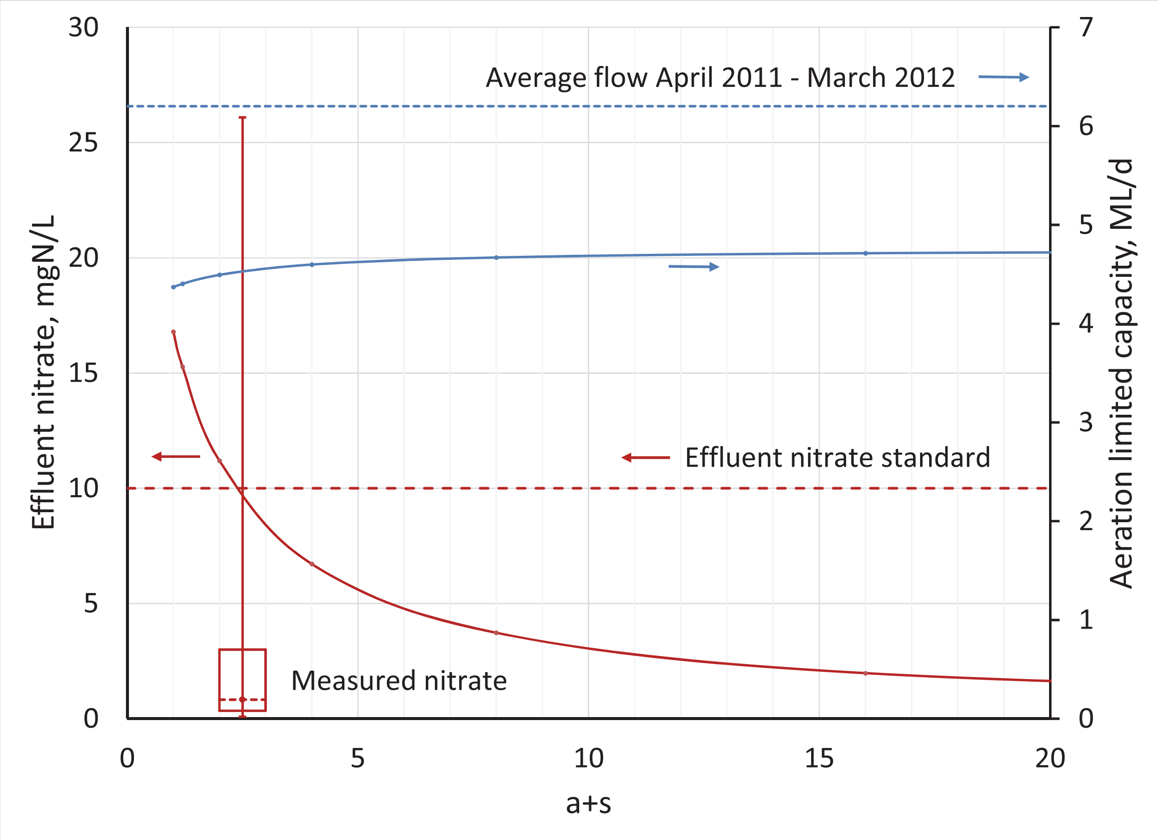


Figure 3.15 (a): Aeration limited capacity and effluent nitrate as a function of combined recycle to the anoxic zone

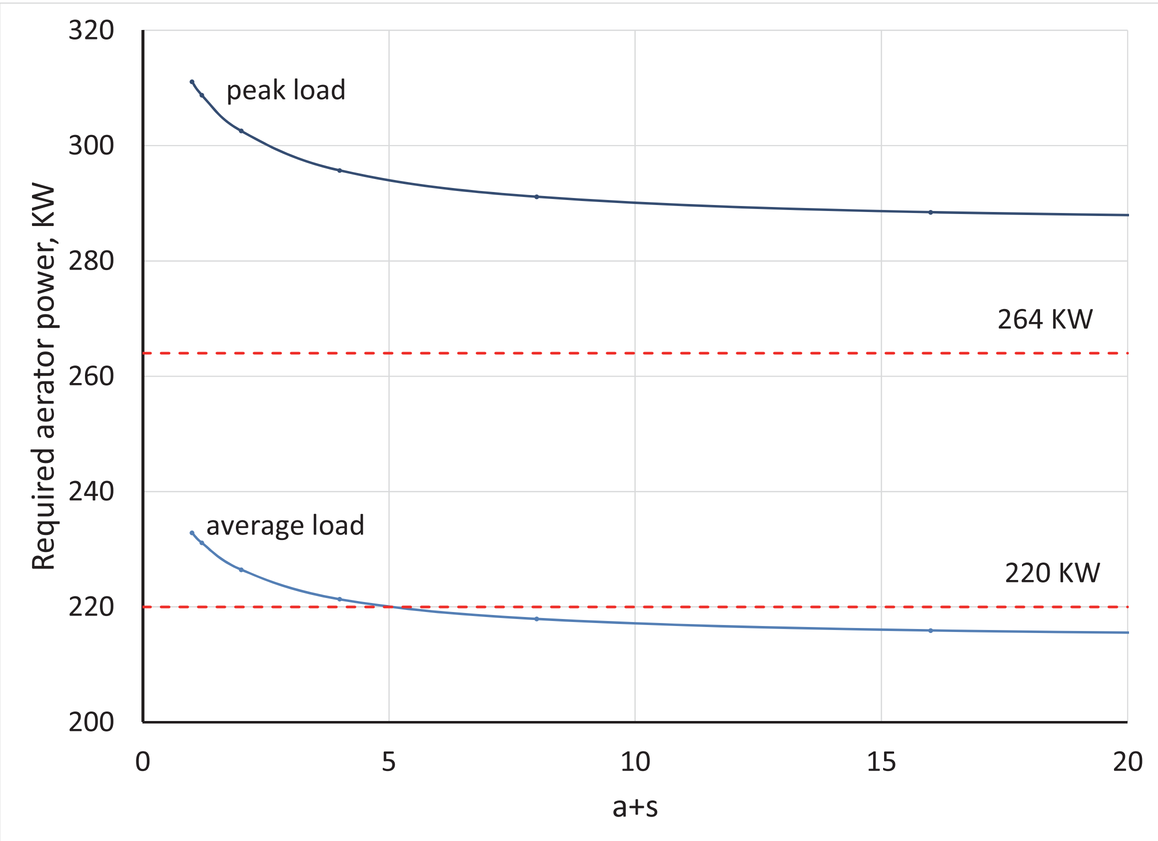


Figure 3.15 (b): Required aeration power as a function of combined recycle to the anoxic zone

3.7 Discussion

3.7.1 Green Drop requirements

The current Green Drop requirements (Department of Water Affairs, 2015) already include the following:

- comparison of flow (ADWF) and COD load with documented design flow (KPA 7 (a) Design Capacity)
- long term capacity planning (KPA 7(c)).
- Water Service Institutions (WSIs) are required to undertake annual Process Audits (KPA 8 Wastewater Asset Management (8 (a)) Process Audit) including the following
    - Description of the treatment plant hydraulic and design capacities (4)
    - Design capability of plant, compared to performance delivered by plant (10)
    - Expected and actual performance modelling of individual process units under design and operational conditions, using operational analysis data (11)

3.7.1.1 Design capacity and capacity planning

The capacity estimation feature of PWSSD can be used to estimate WWTP capacity not only in terms of ADWF and organic load but also in terms of its biological nutrient removal capacity. This could be particularly important for inland plants where effluent standards tend to be stricter. The capacity estimation tool can be especially useful for plants for which design documents are missing, plants where the influent characteristics have changed so that the original design assumption are no longer valid, and plants which need to meet new and stricter effluent standards

PWSSD can be used to determine which of the major unit processes (aeration tank, secondary settlers, aerators, sludge handling) limits the overall capacity, and how the capacity of each unit is affected by the operating conditions (sludge age and recycles in particular). It can therefore be used in treatment capacity planning and as a preliminary step in the design of upgrades.

3.7.1.2 Process audits

Most of the effort involved in the capacity estimation exercise is in collecting the necessary input data. This should be done as part of the WSI annual process audit. As such it is also an excellent tool for checking the completeness and consistency of the operational data and highlighting issues which require further attention. The advantage of PWSSD is that it includes all the major unit operations typically found at wastewater treatment plant therefore this modelling can be carried out simultaneously to the capacity estimation exercise.

Once all the inputs have been collected and reported as part of the process audit, a Green Drop inspector can very quickly set up and run the tool to check the results reported by the WSI as well as compare the capacity estimates and predicted effluent quality with the monitoring data submitted. Strictly speaking, the steady state model cannot predict plant performance under typically variable conditions but it can predict how close a plant is operating to its limits. If a plant is operating below capacity then poor performance can presumably be attributed to maintenance and operation issues

3.7.2 Data requirements

Most of the data required for the capacity estimation exercise should in principle be available from the annual process audits required of the WSI. This would include design information, equipment specifications, monitoring and operational data.

The exception is the detailed wastewater characterization information required, including the detailed fractionation of the various soluble and particulate components and the diurnal variation profiles. The various data input methods as well as the probabilistic fractionator developed during the project can generate detailed profiles from standard monitoring data, which should be adequate for typical domestic wastewaters. For wastewaters with a significant industrial component which affects plant performance an investment in more detailed wastewater characterization would be warranted.

However, all plants are subject to diurnal variations and the generation of representative diurnal variation profiles for the influent wastewater requires a substantial sampling and analysis campaign carried out on multiple days. It is accepted that this type of data will not be available at most plants. In the capacity estimation exercise, the diurnal variations affect the SST and aeration calculations. The SST area limited capacity is determined by the peak observed flow (wet or dry weather) and usually the operators would know what this is (Wu 2105, personal communication) whether or not it is routinely recorded.

On the other hand, the aeration limited capacity depends on the peak oxygen demand. In PWSSD, the peak OUR factor is an empirical function of the peak TOD amplitude factor which is calculated from the peak COD and TKN concentrations in the diurnal variation profiles. In the absence of this data, the user will have to use a conservative default value of the TOD amplitude factor as was done in the Umhlanga case study and this may result in an over-estimation of the peak oxygen demand. However, it should be pointed out that the relationship between peak TOD and peak OUR is only approximate, and an accurate value of TOD amplitude factor does not guarantee an accurate estimate of the peak OUR.

The most direct means of determining whether a plant is operating above or below its aeration capacity is to analyze the dissolved oxygen levels in the activated sludge basins and compare it to the power consumption of the aerators. If the dissolved oxygen targets can be maintained with the aerators operating at below their maximum power then the plant is operating within its capacity; however, if the DO levels drop below their targets even with the aerators running at full power, then the plant is operating beyond its capacity. If the plant does not have onsite DO monitoring then the diurnal variations in DO levels should be measured using field testing equipment during the audit process. The TOD amplitude factor in the model can then be adjusted to match the observed aerator performance.

3.7.3 Steady state model limitations

The Umhlanga case study highlights some of the limitations of the steady state model especially with respect to nitrogen removal. As discussed in Section 3.6, the steady state model does not account for the effect of diurnal variations on nitrification efficiency and therefore the model cannot be calibrated by trying to match the predicted effluent ammonia to the measured values. There are also a limited number of plant configurations from which the user can choose. The simplistic mixing model used in the denitrification calculations did not appear to adequately represent the transport of nitrate between the anoxic and aerobic zones. Nevertheless, a large discrepancy between measured and predicted results does indicate a need for further investigation.

It is important to remember that the capacity estimation tool is intended to predict whether or not the plant is operating close to its limits (both for optimal and adequate performance) and not really to predict what its actual performance will be. The steady state model also assumes relatively stable operation over a period of time and cannot predict the plant's capacity to handle shock loadings and process disruptions (the only exception being temporary increases in flow due to wet weather).

3.7.4 Importance of field trials

The reliability of the capacity estimates generated by PWSSD is strongly dependent on the quality of the inputs as well as the extent to which the model assumptions deviate from the actual operating conditions. The Umhlanga case study was carried out almost exclusively only with data that had already been captured electronically and this was insufficient to resolve a number of questions, especially regarding the sludge age, dissolved oxygen profiles and aeration capacity. It is assumed that many of the data gaps could have been addressed during onsite inspections and interviews with the operators during a process audit, however, this needs to be tested in full scale field trials. The process of trying to gather data required by the steady state model is likely to be a useful exercise in itself because it would draw attention to deficiencies in plant operating procedures and monitoring and also potentially help operators and plant managers understand the operation of their plant better. However, there is no guarantee it would lead to accurate and reliable capacity estimates. There are many reasons why the data available to an audit team might be inadequate, including:

- Missing or incomplete operating and monitoring records;
- Uncalibrated instruments;
- Poor sampling procedures and inappropriate sample handling;
- Missing documentation;
- High staff turnover, with the result that operators cannot answer questions about the plant's history and typical operation;
- Measurements taken during onsite inspections may not be representative of typical operation;
- The plant configuration and operating procedures may not conform to the model assumptions.

It is also assumed that once a model has been set up and a capacity estimation exercise completed for one year's process audit, much less effort will be required to repeat the exercise in subsequent years, especially if the problems highlighted in the first audit have been addressed.

Field trials are therefore required to assess:

1) Barriers to obtaining the necessary input data;
2) The amount of additional time, effort and resources required to obtain useful results;
3) Whether the enhanced capacity estimation capabilities of PWSSD actual does lead to improved plant operation and long term capacity planning.

3.8 Conclusions

The PWSSD programme is a powerful, flexible and freely available software tool which can simultaneously estimate the capacities of all the major unit processes, including the capacity for biological nutrient removal, as a function of the raw wastewater characteristics and operating conditions. Figures 3.7 and 3.10 provide a useful graphical summary of the results. The capacity estimates can be compared to the measured plant performance as required by Green Drop and the results used to assess whether poor performance is due to design limitations or operating and maintenance issues.

The model is easy to set up, the data requirements are much less than those required for dynamic modelling, and most of them should be available from the annual process audits that WSI's are already required to undertake as part of the Green Drop programme. However, field trials are required to assess how well this will work in practice.

It also important to understand the limitations of steady state modelling: it cannot be used to predict the effect of shock loadings and process disruptions and it does not account for the effect of normal diurnal variations on biological nutrient removal efficiencies. Furthermore, while it can predict whether the influent load exceeds the capacity of the various units including the aerators and secondary clarifiers, it cannot predict what the effect on the plant performance will be.

4 Assessment of Dynamic Modelling

Steady state modelling is a useful tool for design and capacity estimation but has limited ability to predict the response of plants to typical variations in raw water quality and loads. Furthermore, the performance of a plant under average loads is not necessarily the same as the average performance of the same plant under variable loads. Dynamic modelling, on the other hand, requires more complex simulation tools, substantially more data for calibration and validation, and is much more difficult to implement.

Two modelling investigations were undertaken using the WEST dynamic simulation platform. The treatment plants investigated were the eThekwini Water and Sanitation's Phoenix WWTP and Umgeni Water's Darvill WWTP. The Phoenix WWTW was 25 ML/d plant which was operating at capacity, while the Darvill WWTW was a 65 ML/d which was generally operating at above capacity. Both treatment works were in the process of being upgraded.

The principle objective of the studies was to assess whether it would be feasible to use such models as part of the Green Drop programme. The original concept was that the model would provide a quantitative benchmark to measure actual plant performance against, so as to be able to distinguish between the different kinds of constraints which might limit the overall performance of the plant. In terms of WWTP modelling practice, this is a relatively limited objective.

It was subsequently realised that dynamic modelling could be a powerful tool for generating quantitative data for risk assessment exercises. Risk assessment is a major focus of the Green Drop Programme, although it has largely been limited to semi-quantitative approaches.

Since considerable effort is required to obtain the data required for a comprehensive dynamic model, it was assumed that the concept was more likely to be taken up if data requirements could be minimized. Thus, a sub-objective was to rely on plant data available from routine process monitoring as far as possible. This led to investigating methods for screening and filtering plant records to identify and remove errors and inconsistencies as far as possible. Two approaches were tried: the first involving manual correction assisted by spreadsheet calculations, the second an automatic filter added to the dynamic model which fits an internally consistent composition for the influent wastewater to measurements as they appear in the time sequence.

A secondary objective of the investigations was to gain experience with the application of the new PWM_SA (Plant-Wide Model, South Africa) developed during WRC project K5/1822. PWM_SA was jointly developed by the Water Research Group (WRG) at UCT and the PRG at UKZN and provides an integrated representation of a plant containing both aerobic and anaerobic processes. This model was incorporated into the 2014 release of the WEST modelling platform provided by DHI (Danish Hydraulic Institute).

In the case of the smaller and simpler Phoenix plant, the model development and calibration proved reasonably successful using the established IWA ASM2 model, which, however, was only able to represent the aerobic section of the plant. A comparison with a PWM_SA version also showed satisfactory agreement, which provides a measure of confidence in the new model. However, the anaerobic sludge digesters were not included in the model.

The Darvill WWTP presented a much more complex problem. Not only is its process configuration more complex, but it is subject to more complex and variable loading from industrial contributions to the wastewater it received. It was also significantly overloaded, and was consequently undergoing a major upgrade. While it was hoped that it would provide a good example for testing the Green Drop concept (i.e. how the model would identify overloading), calibrating the model proved to be very difficult, and indeed was not satisfactorily achieved.

4.1 The Phoenix WWTP

4.1.1 Plant description

Phoenix WWTW is a conventional activated sludge plant designed to treat 25 ML/d of wastewater. The influent to this plant is in the order of 23 ML/d currently and is predominantly domestic sewage. Raw sewage enters the plant at the head of works where it is screened and de-gritted. The resulting grit and detritus is disposed of via land fill. The screened sewage is pumped up to the next process via two Archimedean screws.

The screened sewage is then fed to two Primary Settling Tanks (PSTs) where primary sludge (raw sludge) is drawn off from the underflow at a rate of about 200 m^3/d (approximately 1% of the inlet flow), and a consistency of between 1.5-2.5% solids. This sludge is fed to the Primary Anaerobic Digesters (PAD) for further treatment. The overflow from the PSTs is fed to the Activated Sludge Plant (ASP). De-sludging is manually controlled by the operator, and is carried out twice daily.

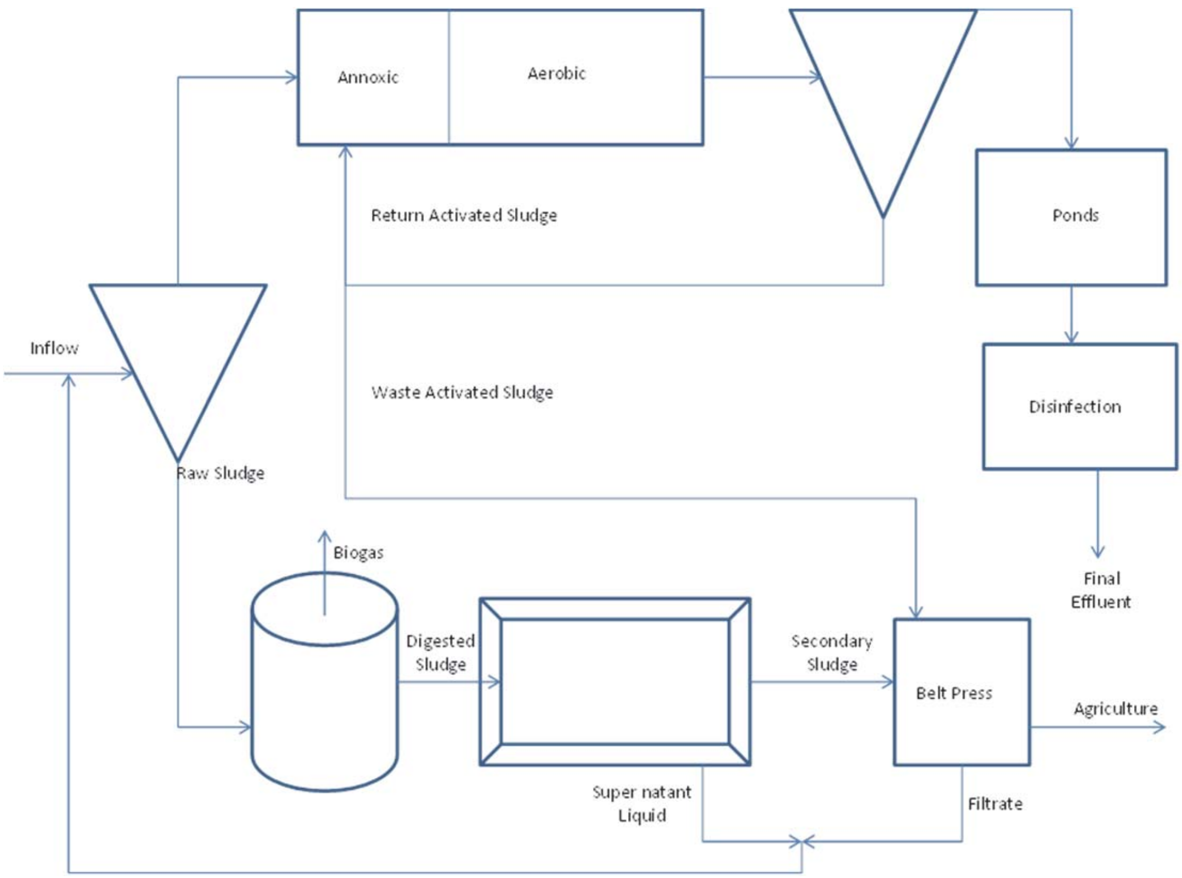


Figure 4.1: Schematic Process Flow Diagram of Phoenix WWTW

Figure 4.2: Aerial Photograph of Phoenix WWTW

The Mixed Liquor from the ASP overflow is split to Secondary Settling Tanks (SSTs) where the Mixed Liquor Suspended Solids (MLSS) is settled and drawn off via hydro-siphons. The bulk of this sludge is returned to the ASP as Return Activated sludge (RAS) and approximately 250 m^3/d is taken off as Waste Activated Sludge (WAS). The WAS is sent directly to the dewatering plant for processing. The overflow from the SSTs flows to a system of 5 maturation ponds before it is finally disinfected with gaseous chlorine and discharged to river. A portion of this effluent is used as a plant utility stream for washing, etc.

The raw sludge from the PST underflows is fed into two PADs. These are high rate mesophilic anaerobic digesters. They are operated at a temperature of approximately 37 °C and a retention time of approximately 24 days. Biogas is collected and stored in a gas holder. A portion of this gas is used to heat water and the rest is flared. The hot water in turn is used to heat the sludge in the digesters via double pipe heat exchangers. Mixing in the digesters is achieved via drawing sludge from the bottom and returning it to the top of the unit or the bottom. These reactors automatically overflow to the Secondary Digesters when fed.

The sludge from the PADs is allowed to settle in the Secondary Digesters. Supernatant Liquor (SNL) is drawn off the top of these tanks and returned to the PSTs. The Digested Sludge(DS) is drawn off the bottom of this tank and is set to the dewatering plant.

The dewatering plant consists of a Linear Screen (LS) or gravity belt which is used as a thickener coupled to a Belt Press (BP). The feed to the dewatering plant is fed mixed with a polyelectrolyte solution to coagulate it and then spread onto the LS. From there it falls onto the belt press and is dewatered. Filtrate from the BP and LS is returned to the PSTs and the resulting dewatered cake is sent to agriculture for disposal. The feed to this plant alternates between WAS and DS as required. Typically the plant processes DS in the morning and WAS in the afternoon.

4.1.2 Available Plant Data

It was decided at the onset of the project to assess how much progress could be made just using the existing plant monitoring data without taking additional measurements or tests. The data available at most plants is generally geared towards compliance monitoring the operation. As such, the frequency of certain determinants will probably not be adequate for fully calibrating a dynamic model. Nevertheless, it may be possible to use the monitoring data to calibrate the model to a reasonable level and perhaps reduce the additional measurements or tests required substantially.

When looking at the data at Phoenix WWTW in relation to applying the PWM the available data at each unit operation node would have to be assessed to determine its applicability.

In general there appears to be sufficient data to get an average indication of the performance of the simulation on the liquid treatment train. The frequency of monitoring would probably not be sufficient for a more rigorous dynamic calibration and additional data collection will be required.

The data for the solids treatment train is predominantly focused on physical determinants and additional data collection will be required for calibration of the anaerobic digestion model.

4.1.2.1 Flows:

The quantity of raw sewage entering the plant is measured on a daily basis, and totalized records are available electronically. The rate of wasting, de-sludging, digester feed and internal recycles are also measured daily and this can be used to construct a fairly representative flow balance around the plant.

4.1.2.2 COD:

The raw sewage COD is measured at the inflow 3 times a week on a Monday, Wednesday and Friday. COD is measured 2 times a week in the primary effluent (PST overflow) on a Monday and Friday. It is measured 3 times in the secondary effluent (SST overflow) on a Monday, Wednesday and Friday. All of these samples are 24 hour composites. The data is useful in the initial stages of calibration and setting up of the simulation as they provide a measure of the average removal rates and performance of the liquid side of the plant. Further sampling would probably be required when refining the model calibration.

4.1.2.3 Nitrogen:

Data for ammonia is measured twice weekly on a Monday and Friday in the raw sewage, primary and secondary effluents on a 24hr composite basis. As with the COD data, this allows a basic average calibration, but would probably require more information for the later stage of calibration. In addition, while the ammonia load in the SNL is measured, the measurements are sporadic. Nitrate/Nitrate levels are measured twice weekly in the secondary effluent and activated sludge basin. This should also be sufficient for a basic calibration.

4.1.2.4 Phosphorus:

Phosphorous is the most sparsely measured determinant and is only done weekly in the raw sewage (composite) and a weekly grab in the final effluent (after the ponds). A comprehensive phosphorus balance across the system will require additional measurement.

4.1.2.5 pH and Alkalinity

pH data is frequently measured at almost all process nodes and sufficient data can be gathered to compare to the model outputs. Unfortunately alkalinity is only measured in the Anaerobic Primary Digesters. As such some additional monitoring would be required in order to calibrate the weak acid/base chemistry model of the simulation.

4.1.2.6 Solids and Volatile Acids

Measurements of total solids and % Ash and Organics are measured twice weekly through most nodes of the solids treatment train. Volatile acids are only measured in the PADs. Some additional measurement will therefore be required to calibrate the anaerobic digestion model.

4.1.3 Dynamic Simulation of the Phoenix WWTW

4.1.3.1 Introduction

The plant was designed for 25 ML/day, and it was a conventional activated sludge plant capable of nitrification and COD removal. With development in the surrounding areas, it was in the process of being upgraded to a capacity of 50 ML/day. In addition, new licence requirements from the Department of Water and Sanitation required the plant to be retrofitted for Biological Nutrient Removal (BNR).

In view of the above the following objectives were adopted:

- To develop a calibrated dynamic model of the Phoenix WWTW;
- Using existing plant data as far as possible;
- Model COD removal, Nitrification/Denitrification as well as Biological Phosphorus Removal;
- Apply the Plant Wide Model to PWMSA to Phoenix WWTW.

The model arising out of this process could then be used to confirm the design, optimise operation and troubleshoot the BNR retrofit.

4.1.3.2 Data Gathering

Data gathering fell into two categories:

- Data required for the plant configuration and layout;
- Data required to calibrate the model.

The quality of the model depends largely on the quality of the data used to develop and calibrate it. Fortunately, owing to the ongoing upgrade at the plant, detailed up to date information was available with respect to the plant configuration. One would require detailed information on the feed to the plant and its characterization as well as data that would allow for mass balances across the entire plant as well as individual unit operations. Depending on the ultimate intended use of the model the data requirements can be onerous, and often become a barrier to entry for modelling for many municipalities. It is for this reason that it was decided to attempt to develop the model for PWWTW without additional sampling using only pre-existing plant data. In the case of PWWTW most of the monitoring and measurement data were captured on eThekwini's Laboratory Information Management System (LIMS) and thus easily available in an electronic format.

4.1.3.3 Challenges with Existing Data

Unfortunately, available data at the plant is often insufficient, or not of high enough quality, to produce a high resolution calibrated model. This largely stems from the differences in the requirements for operational monitoring versus modelling. As stated above, in order to produce a calibrated model, one would require detailed information to allow for mass balances to be calculated across the plant and various unit operations for all species that one would be interested in. In addition, the fractionation of COD, Nitrogen and Phosphorus in the influent is required. All of this information would be required at the highest frequency possible (again dependent on the desired model resolution)

In practice however, monitoring is broadly driven by legislative requirements, costs and personnel capacity. In the case of PWWTW, some analyses such as phosphorus were not required to be measured by DWS and therefore data pertaining to phosphorus is limited throughout the available data set. Certain analyses are not measured throughout the plant as this would be cost prohibitive and would not strictly add value to the operation of a particular unit operation. For example, the only analyses performed on across the dewatering plant would be for solids content because this is the only variable that is relevant for the operation of this unit. Quality in terms of phosphorus, ammonia, etc. would be 'nice to haves' for modelling but add little value to the press operations.

Likewise, the filtrate from the dewatering plants and the supernatant liquor from the digesters flow to a common sump from where it is returned to the head of works. The total flow of this mixture is only taken from this point as it is the most cost effective and practicable place to measure this flow. This makes direct evaluation of a mass balance across the digesters difficult at best.

Frequency, timing and grab vs. composite samples also have implications for the representativity of the available data. Frequency of sampling is limited by operational and laboratory capacity. Sampling and analysis errors are also not uncommon and frequently go undetected.

4.1.3.4 Data Reconciliation

While the above challenges are present, they are not insurmountable and the data can be reconciled by the process described below to yield a useable data set for the purposes of model calibration.

As previously stated, the data for PWWTW was readily available in an electronic format and thus lent itself to ease of manipulation. Were this not the case, the first step would have been to capture the data electronically. From here one would gather all available data into one spreadsheet and arrange it chronologically and sequentially (in terms of the process configuration).

This data would then be scanned for a data dense region, i.e. a date range where the most data are available and the most measurements consistently made. The selected region would then be scanned for obvious errors, outliers and inconsistencies.

Checking the consistency of the data can be time consuming and involves some understanding of the requirements of the model as well as plant operations. Consistency checking of data usually falls into two categories: intra-sample consistency and inter-sample consistency.

Intra-sample consistency is concerned with checking the internal consistency of all the analyses in a particular sample. For example, a raw sewage sample should have some agreement between the total COD and the raw suspended solids. An example from the PWWTW showed a raw sewage sample where the total COD was 180 mg/L whereas the suspended solids for this sample was 340 mg/L. this would mean, that if one applied a rudimentary fractionation to this sample that the soluble COD of the raw sewage would have to be negative. This is clearly not possible, and since different analyses were performed by different sections in the laboratory, it could have been that when the sample was split, it was not done representatively. Another possibility is that one or both analyses were captured incorrectly. Rather than discard the entire sample and lose usable data one would look at the trends within other raw sewage samples to infer which analysis was incorrect. This analysis could then be discarded, or preferably estimated to be in-line with the consistency of the other samples. From the table below one can see that the COD is probably erroneous and could be safely estimated at around 500-700 mg/l.

	RAW_COD	Raw_SusSol
	mg/l	mg/l
2011-05-06	922	**398**
2011-05-09	508	**366**
2011-05-11	778	
2011-05-13	718	144
2011-05-16	630	253
2011-05-20	240	**340**

Inter-sample consistency revolves around checking the internal consistency of all the samples in a particular day. This generally requires an understanding of the plant layout and how the process works. For example, the raw sewage COD should not be less than the primary effluent COD for given day because it is expected that COD reduction would occur across the primary sedimentation tanks.

RAW_COD	PRI_COD
mg/l	mg/l
560	700

So when a situation such as the above arises, it is inconsistent with the rest of the samples. Here possible causes for the error could be unrepresentative sampling or potentially a plug of COD that was missed during the sampling of the raw sewage. Given that the samples are all 24 hour composites and the residence time in the primary treatment unit operation is far less than this, the latter becomes less likely. Thus, in a similar manner to dealing with intra-sample inconsistencies, the values for the primary effluent sample could be discarded or estimated based on the range of available data for that sample. It must be noted that while it is reasonable to do so, not all such outlier cases are as a result of sampling or measurement errors. On occasion process upsets or operation errors could yield such an inconsistent result. The model should therefore be able to take into account such situations. For example, should there have not been desludging from the tanks due to a failure at the desludging pump station then this result would be valid. It would however still be consistent with the lack of feed flow to the digesters and perhaps an increase in MLSS or final effluent COD. So while the sample would be inconsistent with upstream data it would be consistent with downstream data that would help identify this as an upset condition. In addition, such events would have an intra-sample consistency. Other more obvious inter-sample inconsistencies would be the swopping of MLSS and RAS/WAS values. This is fairly common place and usually occurs as a result of either operational staff using the incorrect bottles or the data being swopped around at the lab. Such an error is so evident that these values can be switched with a high degree of confidence.

Other uses of inter-sample consistency checking would be to estimate missing or false zero data. For example if the wasting rate for a given day was blank or zero but the dewatering plant was shown to have run for the usual allotted hours and there are WAS cake samples for the day it would be relatively safe to assume that the wasting rate was not recorded for the day but wasting did take place at more or less the average rate for the period.

By carrying out the above exercise for all samples within the selected range, the data is reconciled and smoothed and can then be used for calibration of a model. In addition, site-specific 'consistency ratios' can be identified for future use with other data ranges from this plant. It must be noted that this approach is valid only for plants where there is not a high degree of variability in the inflow and characteristic of the raw sewage. Thus it would be more suitable for domestic WWTWs and tends to fail for industrial applications (where the feed is highly variable).

From this the various required input files can be generated. Not all inconsistencies will be readily identified and several iterations of this process may be required during the course of the model calibration.

4.1.3.5 Setting up the Simulation

Using the available drawings the plant layout was constructed in WEST as pictured below:

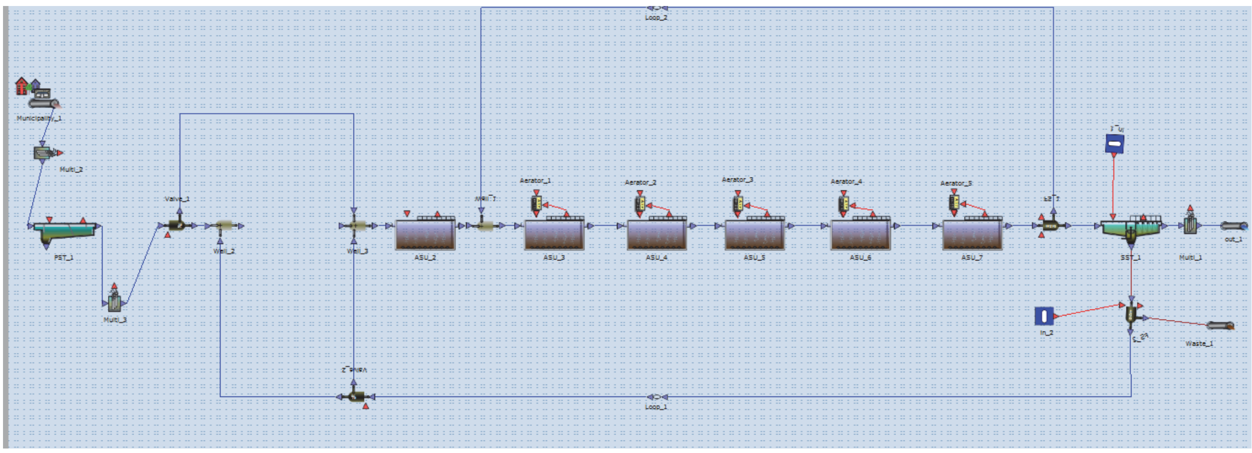
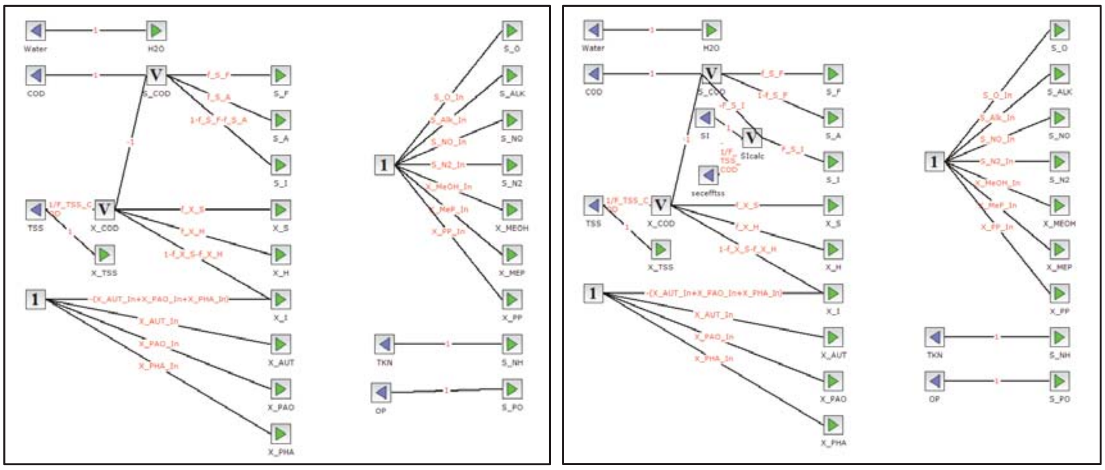


Figure 4.3: WEST ASM2 model layout for Phoenix WWTW

As previously stated, PWWTW is a conventional activated sludge with primary settling followed by a two lane activated sludge basin consisting of six cells. All the primary settling tanks were modelling as one single point settler and the both lanes were condensed into a single train for the purposes of the model. The first cell contains only a mixer and is set to function as an anoxic selector for the purposes of denitrification. The subsequent 5 cells are aerobic with identical aerators. In theory these 5 cells could have been modelled as one large cell with the total volume combined but this was not done so as to facilitate ease of modification of the process for the BNR retrofit. Various recycle lines were also inserted and their respective flows set at zero for this purpose. The Aerator Model in WEST was used to simulate the surface aerators rather than estimating a KLA for each cell. All the secondary settling tanks were modelled as one single point settler with inputs for actual plant data for recycles and wasting rates. Various sensors were inserted for tracking the models performance and to calibrate it. At this point only the liquid treatment train was modelled.

4.1.3.6 Influent Characterization and Calibration

While the ultimate goal of setting up the simulation would be to apply PWM SA to the PWWTW, the fractionation model for this model base is inherently complex. It was thus decided to use an ASM2 model base to estimate the feed fraction as well as gain an appreciation for any additional data requirements. ASM2 is a well-known and understood model base that has been successfully applied to many plants and therefore would be more reliable to initially estimate fractions.



(a) Standard ASM2 fractionation (b) Modified ASM2 fractionation

Figure 4.4: ASM2 fractionation model

Figure 4.4 shows the standard ASM2 fractionation on the left and a modified ASM2 fractionation on the right. As can be seen, there is an input for the soluble inert fraction of the COD (SI) present in the modified version. While no S_I was measured in the raw sewage, this value was estimated from the COD of the final effluent which was presumed to be predominantly un-biodegradable.

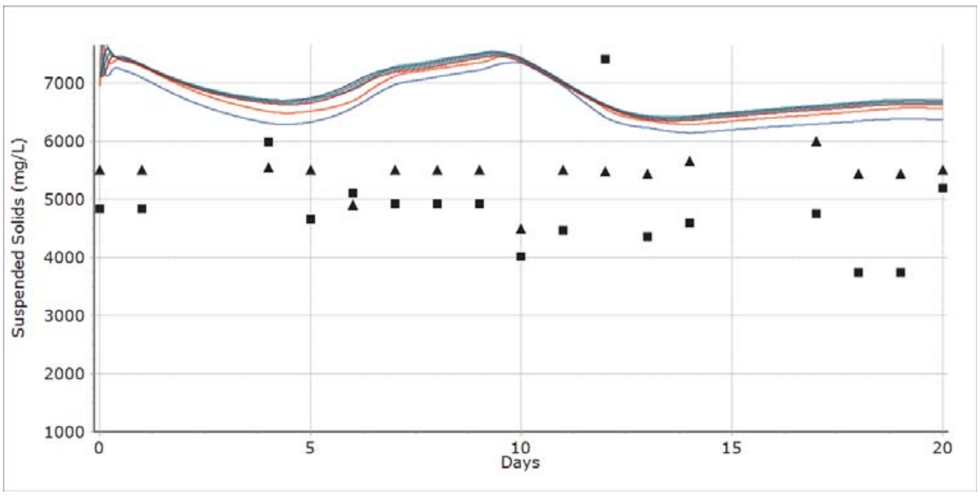


Figure 4.5(a): Activated sludge reactor suspended solids – initial fit

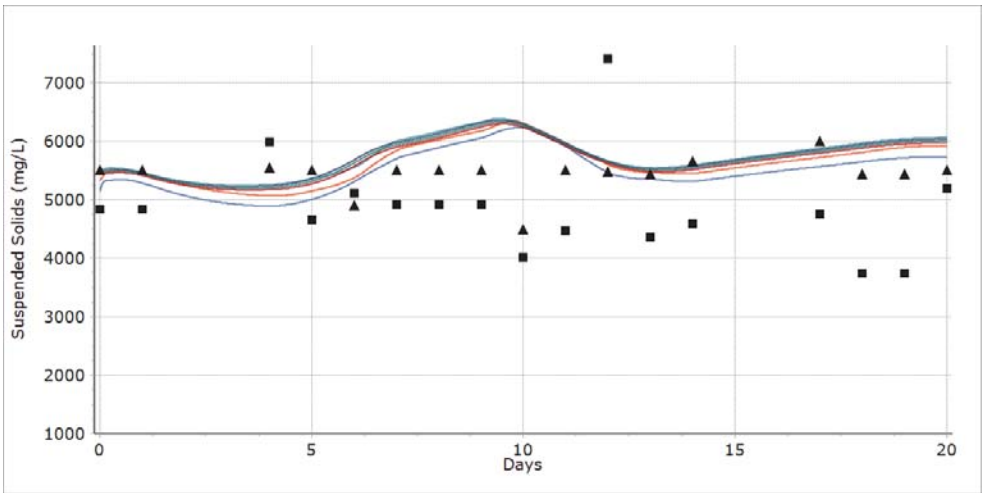


Figure 4.5(b): Activated sludge reactor suspended solids – adjusted fit

Once the fractionation model was setup, the model was calibrated using an iterative process of adjusting various parameters in the fractionation model. It must be noted that aside from adjusting the temperature parameters none of the standard kinetics were modified. An important part of the calibration was ensuring that the initial conditions in the reactor were aligned with the initial conditions of the actual plant data. For example, in Figure 4.5 the simulation was initialized first using a steady state approximation and then dynamically adjusted till the initial suspended solids conditions in the reactor were aligned with the plant data.

Through the course of several adjustments the feed characterization was estimated to a satisfactory degree and the results of the final effluent COD and ammonia especially correlated well with the final effluent data as can be seen in Figure 4.6. It is noted that the final effluent COD produced by the simulation follows the pattern of the actual plant data very well in Figure 4.6(a). This is not altogether surprising, as the influent and effluent data are "coupled", since the final effluent was used to calibrate the influent data. This is useful to setup up the simulation and estimate various parameters. However, this procedure is not viable for simulation for the purposes of testing modifications or optimization, as the final effluent quality would be an unknown in those scenarios. It is therefore required that the feed data be "decoupled" from the final effluent data. This is achieved by using the model data to estimate an average fraction for the SI, and using this as a parameter in the standard ASM2 fractionation model.

The results of this can be seen in Figure 4.6 (b) (after some further calibration tweaks):

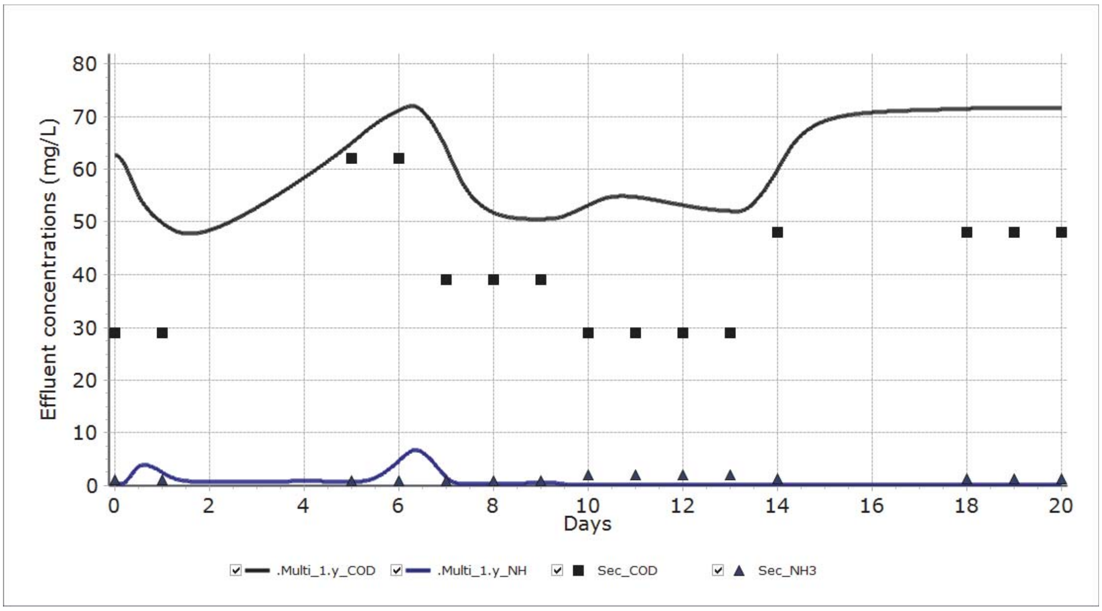


Figure 4.6(a): Final effluent COD and ammonia – initial fit

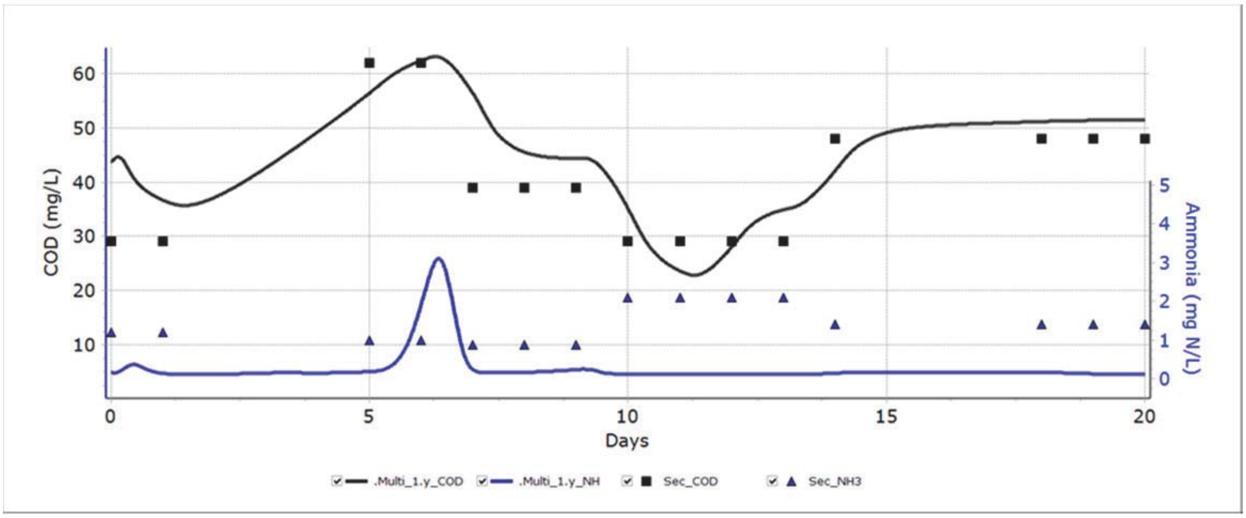


Figure 4.6(b): Final effluent COD and ammonia – final fit

From the above, it is noted that the final modelled COD almost perfectly replicates the onsite actual data. Again this is not surprising, since the model was calibrated on this data set. In order to validate these results the simulation needs to be tested against an independent data set (i.e. that was not used in the calibration). The results of which can be seen in Figure 4.7 below:

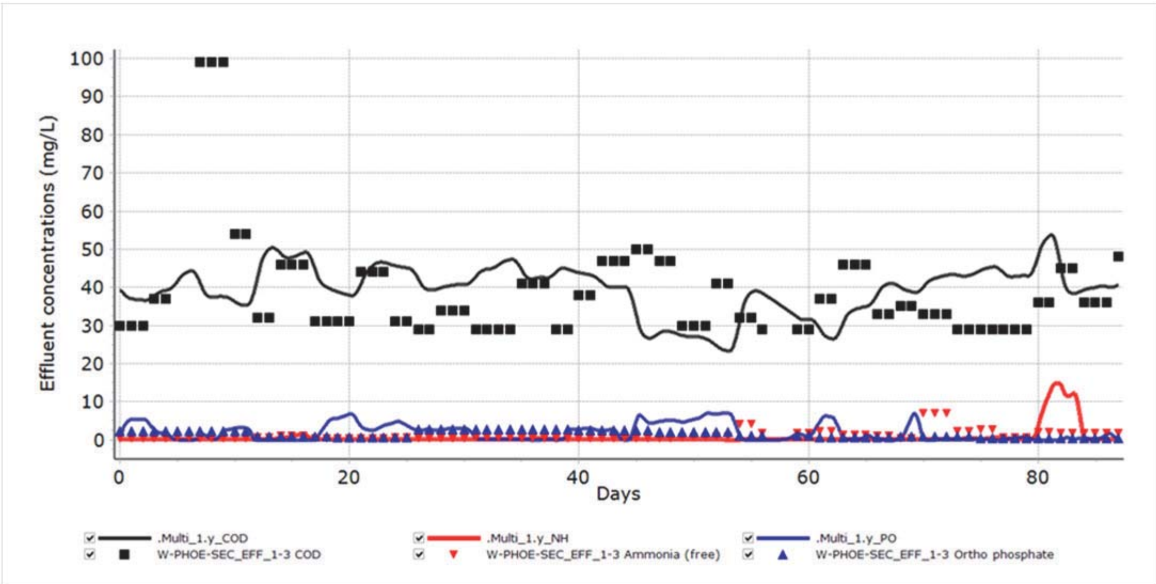


Figure 4.7: Model validation using an independent data set (final effluent COD and ammonia)

From the above it is noted that the model does indeed reflect an accurate simulation of the plant. It must be noted that the initial deviation of the final effluent COD from actual data was as a result of suspended solids in the final effluent as a result of an operational upset. Currently the point settler model used to simulate the secondary settling cells was not able to account for this. It is however safe to assume that the biological model is sound and the feed characterization is representative.

4.1.4 Application of the PWM SA model

Once satisfied with the ASM2 simulation outputs the plant was modelled using the PWM SA.

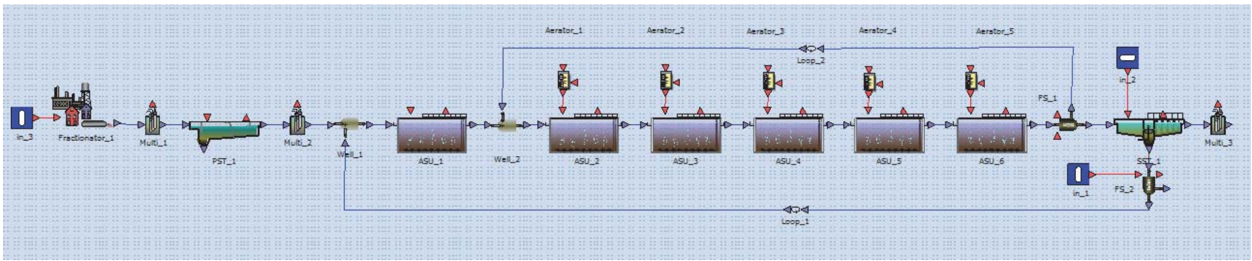


Figure 4.8: PWM_SA model layout for Phoenix WWTW

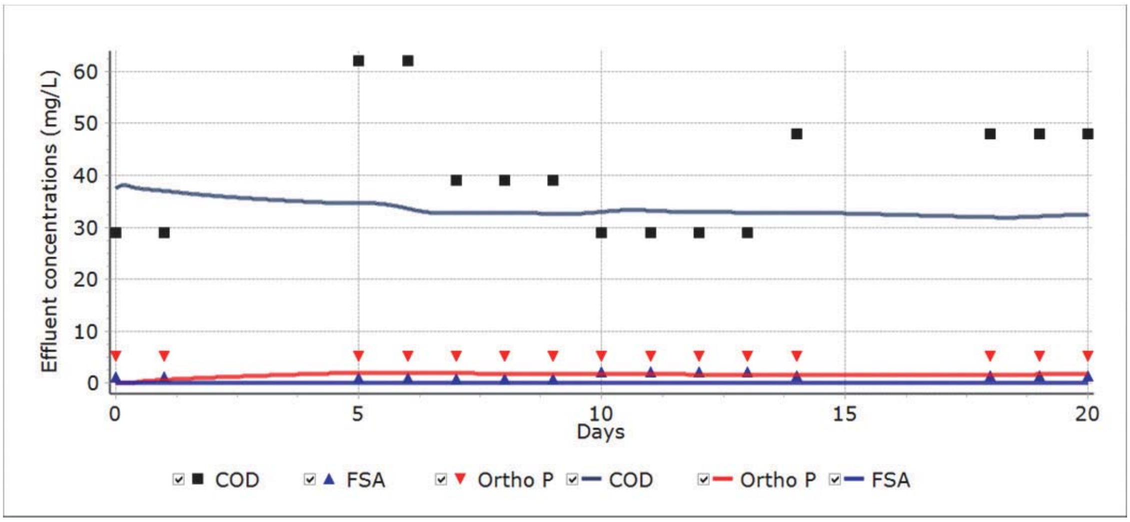


Figure 4.9: Simulated effluent concentrations for Phoenix WWTW using the PWM_SA model.

The PWM SA simulation yielded results that were similar to the ASM2 simulation. It is noted that the simulation outputs were smoother as a result of the fractionator effects. A key difference to note is the inclusion of a fractionator model developed during the project that performed the data reconciliation and smoothing function similar to that described in section 4.3.2.3. The input data for this simulation was extracted from the ASM2 model and the fractionator parameters were adjusted to reflect the feed fractionation obtained from the previously described ASM2 model. The results from this simulation can be seen in Figure 5.7.

4.1.5 Application to risk assessment

It takes a great deal of effort to setup and calibrate a dynamic simulation of an existing plant as can be seen from the preceding sections. However, once achieved, such a simulation is a powerful tool for a plant engineer. It provides a virtual 'sandbox' of the real plant, where different scenarios can be tested without risk to the actual plant's performance or stability. One can effectively run the plant to failure within the simulation to test its operating limits, without environmental or social impacts. This opens up many opportunities for process optimization, troubleshooting, etc. One such application is *consequence analysis* in risk assessments.

As part of the Green Drop Assessments, WWTPs are require to have a Wastewater Risk Abatement Plan (W_2RAP) to identify, quantify, prioritize and mitigate risks pertaining to the plant and its operation. At the heart of this plan is a risk assessment of the plant and its various unit operations. In most cases the risk assessment is semi-quantitative, and defines risk as the product of a scored probability and consequence of a failure or event. The event probabilities are objectively known from historical data and/or manufacturers guidelines. The consequence scores tend to be subjective, as they are drawn predominantly from the experience of people involved with operation and maintenance of these plants. WWTPs can differ dramatically from each other, even if they are identically designed, due to local conditions, raw effluent characteristics and variances in biological characteristics. This leads to disagreement as to the actual severity of a failure's consequence, and uncertainty in quantifying the associated risk.

A dynamic simulation of such a plant allows the risk assessment team to simulate such failures and quantify their impacts more objectively. As an illustrative example, consider a WWTP that processes its waste activated sludge (WAS) through a beltpress dewatering unit for disposal. Failure of this unit prevents the plant from wasting activated sludge and causes a build-up of suspended solids in the system. The failure rates for certain key components are documented in the manufacturer's manuals and thus the probability of failure is known. It is also known that the lead time to procure and install replacement parts is approximately two weeks. There is disagreement about the consequence of such a breakdown. Some operators believe that the build-up of solids will cause the reach non-compliance within the two weeks, and insist that this is a high impact event. Others believe that the breakdown will have a minimal effect on the overall plant performance. The typical decision will be to err on the side of caution and ensure that spares are purchased and held in stock, incurring significant cost at the expense of other risk areas due to a limited capital pool. The uncertainty can be resolved by simulating the failure of the belt-press and assessing the outcome.

The Phoenix WWTP model was used to simulate a belt-press being out of operation for two weeks. Figure 4.10(a) shows the response of the suspended solids in the activated sludge basins to a failure of the sludge de-watering system between days 18 and 31. The simulated effect on the effluent quality is shown in Figure 4.10(b).

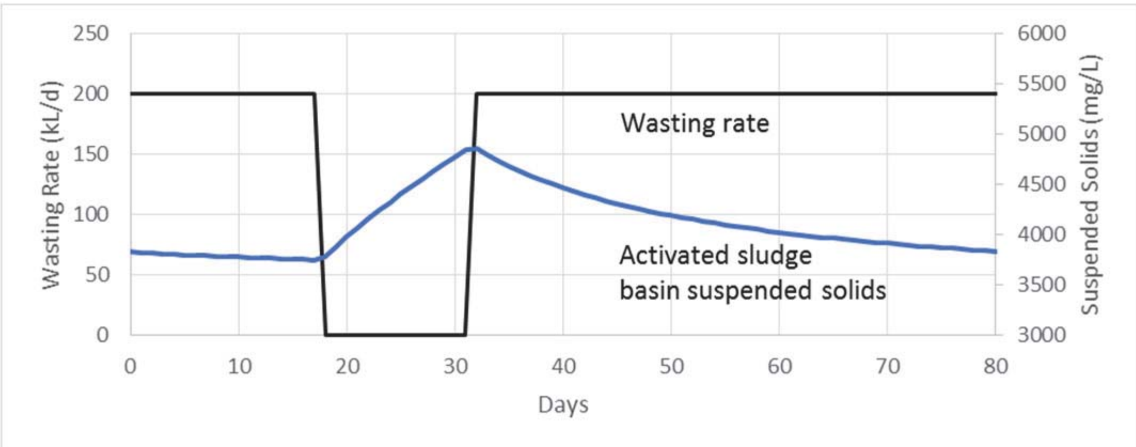


Figure 4.10(a): Simulated de-watering system failure – activated sludge basin response.

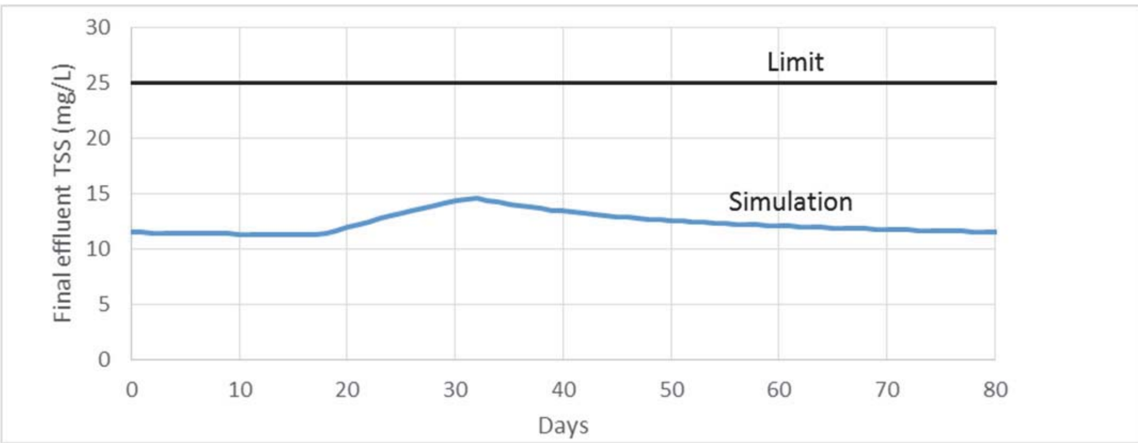


Figure 4.10(b): Simulated de-watering system failure – final effluent response.

The simulated failure predicts that the plant would operate for the two week period without exceeding the compliance limit. Thus a belt-press failure could safely be deemed to constitute a low risk, and funding could be released to mitigate other risk areas.

Thus, a dynamic simulation has the potential to add objectivity to the W_2RAP and steer it towards a fully quantitative assessment.

4.2 The Darvill WWTP

4.2.1 Background

In this case study, an attempt was made to use the new PWM_SA model to model Umgeni Water's Darvill WWTP. The Darvill Wastewater Treatment Plant is a 65 ML/d Biological Nutrient Removal (BNR) plant which treats domestic and industrial wastewater from the city of Pietermaritzburg and discharges treated effluent into the Umsunduzi River. Biological treatment at Darvill consists of an activated sludge plant and anaerobic digestion of the primary sludge.

Over the years, there have been a number of upgrades and changes to the process configuration and operation of the plant in response to increasing flows and organic loads as well as more stringent discharge standards. In 2014, the plant was operating at above its design capacity and plans were underway to upgrade it to 100 ML/d.

As discussed at the beginning of this chapter, one of the objectives of this case study was to assess the application of dynamic modelling to an overloaded plant and one with a significant industrial component in its influent. Furthermore, Umgeni was able to supply a wealth of data on the operation and performance of the plant, beyond what is typically available, which made this case study appear very promising initially. However, the plant has a number of significant problems and challenges which made it extremely difficult to model.

Based on the preliminary results it was concluded that this type of modelling study was far too complex to be carried as part of a routine audit and therefore it was not worth continuing with this particular case study for the purposes of the current project. However, the work undertaken on the Darvill model did highlight a number of challenges in modelling plants, especially those operating above capacity and receiving significant quantities of industrial effluent. These are listed below. Additional information on the model structure and the attempts to calibrate it are provided in Appendix A.

4.2.2 Modelling challenges and limitations in the Darvill case study

4.2.2.1 Challenges related to data availability

All modelling studies will be limited by the type, quantity and quality of data available. The type and quantity of data plants are required to collect for monitoring purposes is typically insufficient for modelling. The probabilistic fractionator was designed to convert available influent measurements into a self-consistent COD and element balanced set of model components, however, the current versions do not correct for the effect of diurnal variations on the calculations of the daily loads.

In the Phoenix case study, the influent data was available in form of composite samples which are more representative of the average daily loads, while only daily grab samples were available for Darvill. It was hoped that the equalization tank at Darvill would provide sufficient damping of the normal daily variations in settled sewage data used in the calibration of the activated sludge system model. However, there was still a lot of scatter in the measurements and it was not clear if this was due to grab sample timing, measurement errors or actual variations in the daily loads. There were similar concerns about the dissolved oxygen measurements which were also single daily measurements.

More work is therefore required to resolve the issues related to diurnal variability if the proposed modelling tools are to be widely applied.

4.2.2.2 Challenges related to overloaded plants

When a plant is overloaded with respect to its aeration capacity, it is expected that the dissolved oxygen levels in the activated sludge basins will vary with the influent loads. However, it turns out to be quite difficult to model this situation. The profile is very sensitive to the assumed oxygen transfer coefficients (Kla's), which probably also vary with changes in operating conditions and concentrations. Biological processes such as nitrification are strongly dependent on the dissolved oxygen levels therefore it is essential to get this aspect of the modelling right. Future work should focus on a more robust approach to modelling this type of situation.

The Darvill WWTP was no longer operating as it was originally designed or as described in the operating manual. For example, most of the settled sewage bypassed the anaerobic selector, aerators have been installed in the anoxic zone and phosphorous removal was being achieved using chemical precipitation. This made it harder to make reasonable assumptions about the plant operation and performance where data were missing.

Processes with highly variable performance are inherently more difficult to model than processes that are relatively stable. Extreme situations are more likely occur which may fall outside the range of validity of the various models and typical modelling assumptions.

4.2.2.3 Challenges relating to the industrial content of the influent

The fractionator tool defaults (set of components, stoichiometry and typical fractions) have been set up based on wastewater characterisation data for typical domestic wastewaters and may not handle wastewaters with a significant industrial component as well. It may therefore be necessary to adjust the PWM_SA components' stoichiometry or possibly even define new industry specific components with their own stoichiometries and degradation kinetics for some plants receiving industrial effluent. This would of course require a substantial modelling effort as well as additional measurements and would not be feasible as part of an audit.

In Darvill's case, a particular concern is the high oil content of the raw water. While domestic wastewater always contains some level of FOG (fats, oils and grease), edible oil processing effluent is a significant fraction of the Darvill influent meaning that oil levels will be higher than typical and quite variable depending on what is happening at the industrial facilities. Vegetable oil has a higher COD/g and zero nitrogen content compared to the default PWM_SA components. Another problem with FOG in the influent is that it does not mix well with water, so getting representative samples may be challenging. Furthermore, FOG decreases aeration efficiency and intermittently high FOG levels may have contributed to the difficulties in modelling the oxygen transfer rate.

4.2.3 Conclusions and recommendations

While the Darvill WWTP modelling exercise itself was not successful, it highlighted a number of important challenges relating to modelling plants which are overloaded and/or receiving significant amount of industrial effluent, as well general issues relating to the type of data available, which need to be addressed in future work.

- The available plant data will not always be sufficient for model calibration even with the use of the probabilistic fractionator.
- Model calibration should ideally be carried out using composite measurements for the influent. Since these are not always available, additional tools will have to be developed to estimate daily loads from grab samples if the propose modelling tools are to be widely applied and/or more detailed wastewater characterization studies will have to be carried out on a case by case basis.

- Oxygen transfer coefficients vary with process conditions therefore attempting to model an overloaded aeration system with fixed Kla’s does not work very well. More robust methods for modelling oxygen transfer in overloaded systems need to be developed.
- More work is required on the characterization and modelling of wastewaters with significant industrial effluent content. If the industrial contribution to a municipal wastewater is reasonably consistent, it may be sufficient to simply adjust the PWM_SA component stoichiometries as necessary, however, if the industrial effluent loads are variable and significantly different in composition to the domestic contribution, then additional model components may have to be defined.
- Precipitation with alum should be added to PWM_SA. The speciation sub-model is already in place and the relevant components, species and reactions just need to be added. Chemical precipitation for phosphorous removal and improved sludge settleability is expected to be commonly required at overloaded plants.

5 Discussion

The overall goal of this project was to investigate and assess the potential for using wastewater treatment process models to support the goals of the Green Drop programme, namely to help develop core competencies in the municipalities, and over time, improve the level of wastewater management in South Africa. In assessing the modelling tools developed and investigated in the study, it was important to consider both whether they were useful and whether they were practical in the Green Drop context. The capabilities and limitations of the modelling tools were explored in the Case Studies presented in the previous chapters. However, the input of other practitioners in the field, especially those with experience of the Green Drop process, was essential for evaluating both the usefulness and practicality of the tools. The discussions of the reference group and participants at the dissemination workshop (Appendix C) were in fact key to developing several of the main conclusions and recommendations emanating from this project. This section summarises the main areas of discussion.

5.1 Advantages and barriers

The main barrier to the implementation of modelling in the Green Drop process and in wastewater management sector is lack of capacity. This is most acute in in the smaller municipalities and rural areas but is also a challenge for the larger municipalities and organisations like Umgeni Water who also struggle to recruit and retain qualified and experienced engineers. WWTP modelling, in particular using the dynamic simulators, has a significant learning curve and already overstretched process engineers often have difficulty allocating sufficient time to the task.

On the other hand, once a model is set up and calibrated, it can be used to improve design, operation and risk assessment, helping to avoid costly design mistakes and process disruptions, more than justifying the initial investment of time, effort and other resources, and helping to achieve the overall goals of Green Drop.

Furthermore, when properly used, models and modelling can be an invaluable interactive tool for learning the fundamentals of wastewater treatment and for training in plant operations. Thus modelling can be incorporated into university level as well as engineer-in-training courses. Engaging in modelling exercises can also help operations staff gain a greater understanding of their treatment plants and their limitations.

Therefore the recommendation coming out of the both the workshop and reference group was that the modelling tools should be made available to those larger municipalities and other organizations that have the capacity to use them, whether or not modelling becomes a formal part of the Green Drop requirements. There was also strong support for a planned WISA Specialist Group to promote and support the use of modelling in the Wastewater Management sector. The hope is that over time, a critical mass of expertise in modelling can be built up, eventually making it possible to extend the benefits of modelling to the smaller municipalities.

5.2 Data requirements and reconciliation

Confidence in model outputs depends strongly on the quality and completeness of the data input to the model. This was particularly evident in the Umhlanga case study where some critical operations data including dissolved oxygen levels and sludge ages were missing, and in the Darvill case study where the grab sample data and dissolved oxygen measurements may not have adequately represented the daily average loads.

As discussed in the Tools and Case Studies sections, the data requirements of modelling are greater than that of compliance monitoring and therefore the data that is routinely collected at plants may not be sufficient to meet modelling objectives. The cost and effort to obtain the additional measurements is an additional barrier to use of modelling. Furthermore, as discussed in Chapter 4, the quality of the data available is highly variable and the bulk of the modelling effort often has to go into getting the available data into a useful form and generating estimates for data that is missing.

Getting the wastewater characterization right is essential since all the biological treatment models depend on the composition and fractionation of the influent. The wastewater characterisation tool in the steady state model (PWSSD) and the probabilistic fractionator developed as part of this project are a first step towards reducing the need for additional measurement campaigns as well as determining what the minimum data requirements are in various scenarios. However, they do have their limitations:

1. The fractionators may not work as well when the wastewater has a significant industrial component, i.e. deviates from a typical domestic wastewater.
2. The tools developed so far only deal with data reconciliation for the influent. Issues such sampling errors and missing operation data still have to be addressed.
3. The tools developed currently do not have the ability to estimate daily loads based on grab sample data. This is a problem for plants which collect single daily grab samples as opposed to composite samples for monitoring purposes.

Future work on the modelling tools to be disseminated should include the following:

a) Developing guidelines on the minimum data requirements for modelling various types of plants with different treatment objectives, e.g. requirements for biological nitrogen removal, biological phosphorous removal, etc.

b) Developing user friendly guidelines for data checking and reconciliation. For example, users should know how to conduct simple mass balances to check the consistency of their data

c) Generating more examples of typical raw wastewater profiles for different parts of the country and for plants serving different types of communities. This type of data could be collected as part of field trials conducted in different parts of the country.

d) Developing an interactive user interface for the Excel fractionator similar to PWSSD

5.3 Steady state vs dynamic modelling

Attendees at the dissemination workshop (Appendix C) were generally most excited by the potential use of dynamic simulation to generate quantitative and plant specific risk analysis data since risk analysis is a major focus of the Green Drop programme. On the other hand, they appeared to be less convinced of the benefits of the Capacity Estimation tool which uses the steady state model.

However, it is important to appreciate that the dynamic simulators have much larger data requirements and a significantly higher learning curve than the steady state model. Furthermore, dynamic models are more sensitive to errors in the input data whereas the steady state model uses averaged inputs so the effect of sampling errors and outliers is minimized, potentially reducing the data reconciliation effort required. In addition, the high cost of the software licenses for the dynamic simulators make them unaffordable for most of the smaller municipalities, whereas the PWSSD programme can be used for free by anyone with Microsoft Office.

Furthermore, the Capacity Estimation tool does have a number of important applications:

1. Many older treatment plants do not have proper design documents and are incurring penalties in the Green Drop audit as a result. The reference group saw the Capacity Estimation tool as a convenient way of generating the missing design capacities.
2. It can also be used to recalculate the capacities of plants in areas where rapid development is occurring in the catchment and where the influent characteristics and/or effluent standards have changed significantly.

3. While less powerful and flexible than the dynamic simulators, the steady state model can still be used to assess whether a plant is being operated optimally and what its limitations are. It can therefore help the operations staff understand their plant better leading to better process control and performance.

For these reasons, the project team and reference group concluded that steady state modelling would be a more a practical entry point for municipalities wishing to build expertise in modelling and the Capacity Estimation tool should be the initial focus of capacity building initiatives. Larger municipalities and organizations which can invest in the dynamic simulators can also apply dynamic modelling to risk assessment.

5.4 Incorporating modelling into Green Drop

Three things were clear from both the case studies and workshop and reference group discussions:

1. Setting up either a steady state or dynamic model is too time consuming to be part of the Green Drop audit itself and any modelling would instead have to be part of the preparatory work undertaken by municipalities leading up to the audit. It is proposed that Capacity Estimation tool could be incorporated into the annual plant audits municipalities are supposed to carry out for their own treatment works. However, the Umhlanga Case study was purely a desk exercise using data that was already available. Additional field trials are required to determine how well this would work in practice. Dynamic modelling is too complex for it to be worth setting up purely for the purposes of an audit, but a model which had been set up and calibrated for other reasons could potentially be used, especially for risk assessment.

2. It is too early to introduce modelling as a mandatory requirement in Green Drop. The workshop participants and reference group felt that the modelling tools should first simply be made available to those municipalities and organizations which are interested in using them. Once the usefulness of the models has been demonstrated in several field trials, DWS can begin a step by step process to gradually introduce modelling into the Green Drop requirements, starting with the awarding of bonus points, as has been done with other aspects of the programme such as risk analysis. This should be brought up for discussion in the next Green Drop planning meeting.

3. Lack of capacity will remain a barrier to introducing any type of modelling into the smaller municipalities for the foreseeable future. However, larger municipalities could potentially earn additional bonus points by partnering with smaller municipalities to assist them (Cross-Pollination).

6 Conclusions and Recommendations

1. Modelling requires a considerable investment in time and skill. In South Africa, only the large metros will be able to develop modelling capacity in the foreseeable future.
2. Steady state modelling is much less demanding than dynamic modelling, and should be introduced first, as part of a programme to develop modelling capacity in the industry.
3. Using dynamic simulators to assess various risk scenarios and provide quantitative plant specific data for Wastewater Risk Abatement Plans is seen as a very promising application of modelling. However, the effort and expertise required to set up and calibrate the dynamic models as well as the high cost of the dynamic simulators means that most municipalities are unlikely to be able to pursue this option in the near future. Other risk assessment tools are already available and simulation would provide a supplement, not a replacement, for these.
4. The steady state capacity estimation model will be particular useful to avoid Green Drop penalties for older plants where proper design reports are not available. It can also be used to predict how plants will respond to increases in population or changes in wastewater characteristics.
5. Although a WTTP model can produce results which are useful for a Green Drop audit, setting up a model is too time consuming to be considered as part of the audit itself. Modelling would have to be completed beforehand as part of the preparatory work. The capacity estimation exercise could potentially be incorporated into the annual plant audits municipalities are supposed to carry our prior to the Green Drop audit. Both steady state and dynamic models can be useful for a range of other purposes besides providing data for the audit.
6. It will not be practical to introduce modelling as a Green Drop requirement in the near future. Rather, the Green Drop programme could adopt a strategy to progressively encourage the development of modelling capacity in order to enhance wastewater treatment practice. This could start with bonus points for the effective use of modelling to meet existing Green Drop goals better, such as capacity estimation or risk assessment. Bonus points can also be awarded for partnerships between larger and smaller municipalities which promote the use of modelling.
7. A specialist division of the Water Institute of Southern Africa should be established to promote and develop WWTP modelling practice.
8. While case-studies presented in this report provided valuable information on the capabilities and limitations of the modelling tools developed, they were essentially desk studies, and there is still a need for a pilot project or field trials to test the usefulness and practicality of the tools in the context of an actual audit.

Three new tools were developed, which are especially aimed at the issues that were expected to be encountered during a Green Drop audit. These are:

1. A steady state model which is suitable for estimating the capacity of a WWTP. This is particularly useful for older WWTPs where the original design documents have been lost, or where the characteristics of the wastewater are markedly different from those used for the design. It can also be used for long-term capacity planning.
2. A probabilistic wastewater fractionator for estimating the characteristics of the influent wastewater from routine plant measurements. This minimises the requirement for extensive additional measurements that would normally be required to characterise the wastewater for modelling purposes.
3. The PWM_SA (Plant Wide Model – South Africa) was introduced as a tool for dynamic modelling using the WEST software platform. This an outcome on of a many years of research carried out at the University of Cape Town and the University of KwaZulu-Natal.

Specific recommendations for the further development of the modelling tools include the following:

a) Developing guidelines on the minimum data requirements for modelling various types of plants with different treatment objectives.

b) Developing user friendly guidelines for data checking and reconciliation.

c) Generating more examples of typical raw wastewater profiles for different parts of the country and for plants serving different types of communities. This type of data could be collected as part of field trials conducted in different parts of the country and should include diurnal variations profiles.

d) Developing a more user friendly interface for the probabilistic fractionator

e) Developing improved methods for characterising wastewaters with a significant industrial contribution.

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Appendix A: Application of PWM_SA model to Darvill WWTP

A.1 Plant Description

The Darvill Wastewater Treatment Plant is a 65 ML/d Biological Nutrient Removal (BNR) plant which treats domestic and industrial wastewater from the city of Pietermaritzburg and discharges treated effluent into the Umsunduzi River. Biological treatment at Darvill consists of an activated sludge plant and anaerobic digestion of the primary sludge. One of the particular challenges at Darvill is that it receives effluent from edible oil processing facilities in the Pietermaritzburg area. Any problems or upsets at the onsite wastewater treatment plants at these facilities can result in high levels of fats, oils and grease (FOG) reaching Darvill.

Over the years, there have been a number of upgrades and changes to the process configuration and operation of the plant in response to increasing flows and organic loads as well as more stringent discharge standards. Figure A.1 shows the current process configuration from the primary to the secondary settlers, which is the focus of the modelling exercise.

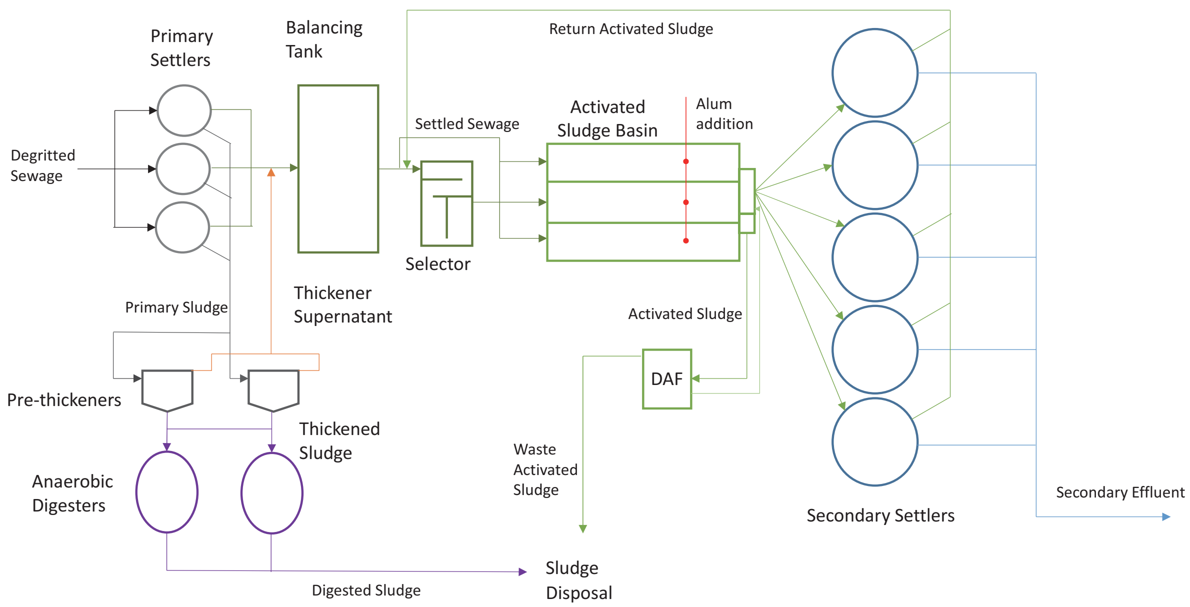


Figure A.1: Process configuration for the Darvill WWTP

The inlet works (not shown) includes diversion of excess flow to a storm water dam, screening and degritting. The raw sewage then enters three primary settlers (PSTs). The PSTs have rotating bridges which skim scum and oil off the surface of the tanks for offsite disposal.

The settled sewage discharges to a 10 000 m^3 balancing tank before being pumped to the activated sludge (AS) plant. The AS plant consists of a 3400 m^3 anoxic/anaerobic basin or selector followed by a three lane aeration basin with a volume of 19 600 m^3. About 4% of the settled sewage is mixed with the return activated sludge (RAS) in the selector while the remaining 96% is sent directly to the aerated channels.

The plant was originally designed with five 75 kW surface aerators per channel with the first section of each channel operated as an unaerated anoxic zone. However, due to the increased COD load, the aeration requirements now exceed the available capacity. Furthermore, Darvill's high and variable loads of FOG (fats, oils and grease) negatively impact aeration efficiency. As a temporary fix, three additional

30 kW aerators had been installed in the first section of each channel (previously the anoxic zone) as shown in Figure A.2.

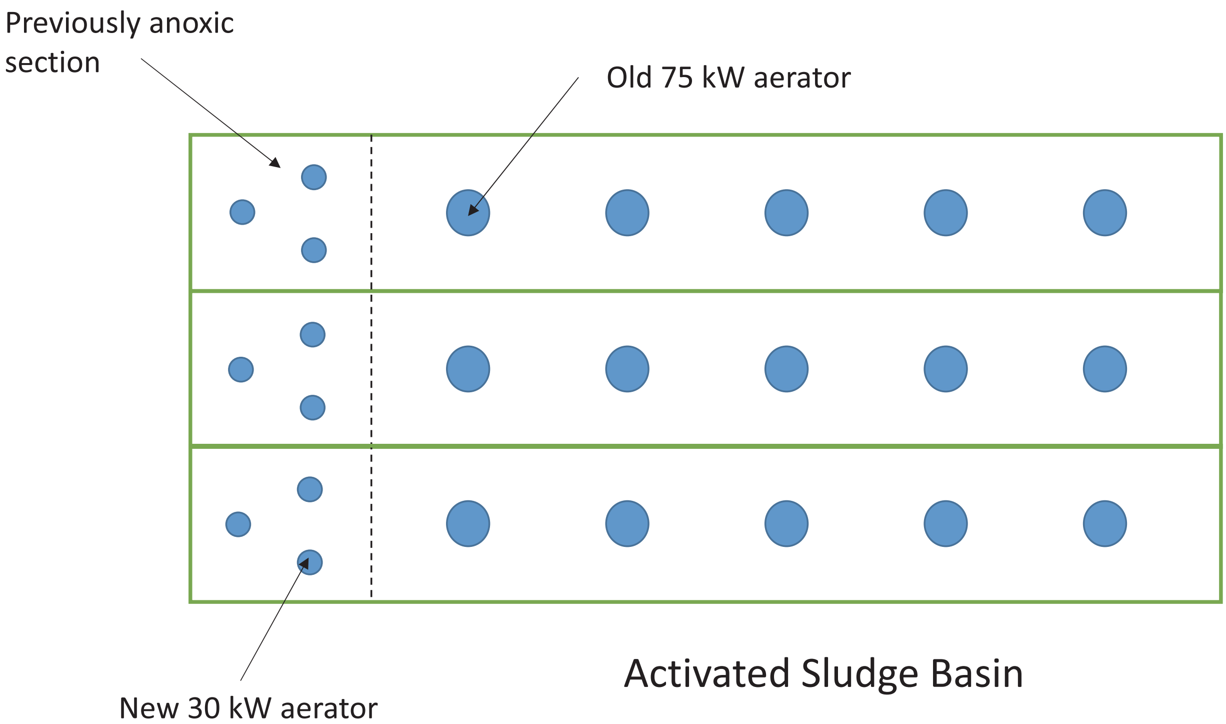


Figure A.2: Aerator locations in the AS basin

The activated sludge plant was originally configured with the bulk of the settled sewage being fed to the anaerobic zone along with a VFA rich stream from the pre-fermentation of the primary sludge, in order to promote the growth of phosphate accumulating organisms for excess phosphorous removal. However, for the time period evaluated in the modelling exercise, the pre-fermenters were no longer in operation and the bulk of the settled sewage was bypassing the anaerobic zone and being fed directly to the aerated zone. Consequently, there was insufficient readily biodegradable COD in the anaerobic zone to support biological excess phosphorous removal and the plant was instead relying on chemical precipitation using alum to achieve low levels of orthophosphate in the effluent. Alum was being dosed into the second to last sections of the aerated channels as shown in Figure A.1.

Mixed liquor from the AS reactor was discharged to five secondary clarifiers. The secondary clarified effluent discharged to a maturation river (not shown) where it was chlorinated prior to final discharge to the Umsunduzi River. Settled sludge from the secondary clarifiers was pumped back to the anoxic/anaerobic selector.

Waste activated sludge was drawn from the end of the AS reactor and pumped to the DAF (dissolved air flotation) plant. The subnatant from the plant was returned to the end of the AS reactor. The thickened sludge was pumped to sludge disposal.

Primary sludge from the primary settlers was pumped to the pre-thickeners. The supernatant from the thickeners was pumped back to the balancing tank. Settled sewage samples were collected after the balancing tank and therefore include the primary settler effluent and thickener supernatant

The thickened primary sludge was pumped to two egg shaped digesters with a combined capacity of 9000 m^3. Part of the gas generated was used to heat the influent to the digesters while the rest was flared. The digested sludge was combined with the WAS before being pumped to sludge disposal.

A.2 Data provided

As in the Phoenix WWTP, the model calibration and validation for Darvill was attempted using only routine monitoring and operational data. The data set provided by Umgeni consisted of flow and composition data for days of low or zero rainfall and normal plant operation recorded between November 2013 to August 2014 and is summarised in Table A.1. The data set included four separate periods of reasonably continuous data with one to two month gaps in between. The largest of these was the period 8 May to 24 June and this subset was used for the initial model calibration. The measurement data provided is summarized in Table A.1.

Table A.1: Data provided

<table>
<tr><td colspan="4">Data periods:
27 November 2013 - 16 January 2014
13 March 2014 - 1 April 2014
8 May 2014 - 24 June 2014*
11-29 August 2014</td></tr>
<tr><td>Raw Sewage</td><td colspan="2">Settled Sewage</td><td>Final Effluent</td></tr>
<tr><td>COD (total)
NH_3 (FSA)
pH
SRP (orthophosphate)
Temperature
TKN – limited
TSS
Rainfall</td><td colspan="2">COD (total)
NH_3 (FSA)
pH
SRP (orthophosphate)
Temperature
TKN – limited
TSS</td><td>COD (total)
NH_3 (FSA)
pH
SRP (orthophosphate)
Temperature
TKN – limited
NO_3 (NO_x)
TSS</td></tr>
<tr><td colspan="2">Solids analysis</td><td colspan="2">Flow balance</td></tr>
<tr><td colspan="2">MLSS
DSVI (mixed liquor)
Sludge age
Raw sludge % TS and % VSS
Pre-thickener % TS and % VS
Digested sludge (#5 and #6) % TS and % VS</td><td colspan="2">Inflow
Settled sewage to balancing tank
Primary sludge to thickeners
Primary thickened sludge to the digesters
Thickeners overflow</td></tr>
<tr><td colspan="2">Dissolved oxygen</td><td colspan="2">Digesters</td></tr>
<tr><td colspan="2">Last two sections of each AS lane (A,B and C)</td><td colspan="2">pH, conductivity, temperature (#5 and #6)
gas composition (CH_4, CO_2, O_2, H_2S)
flow (kg/h)</td></tr>
</table>

* Used for initial model calibration

Unlike the Phoenix case study, most of the data available was grab samples collected between 10 a.m. and 2 p.m. rather than composites. Daily grab samples of raw water are not necessarily sufficiently representative of the plant loads since the raw water composition usually varies substantially over the course of a day. The settled sewage samples were collected after the 10 ML balancing tank which provided some mixing of the feed and damping of the variations in the influent profile. It was reported that the settled sewage grab samples on average agreed within 8% with composite samples collected every one to two weeks (Mluleki Mnguni, Personal Communication, 4 March 2015). It was therefore hoped that grab sample data would be adequate for the calibration of the activated sludge model.

A.3 Influent fractionation

In wastewater treatment modelling, influent characteristics which are typically available only in aggregate measurements such as TSS, COD and TKN, have to be converted to model component concentrations including soluble, particulate, biodegradable and unbiodegradable fractions before they can be inputs to the biological treatment unit models. In the PWM_SA model, each component is assigned a specific stoichiometry in terms of its C, H, O, N and P content. The probabilistic fractionator calculates the fractionated composition of the influent which gives the best agreement with the input measurements subject to a number of constraints and assumptions.

In the Darvill case study, it was decided to model the settled sewage and thickened primary sludge produced by the primary clarifier as two different input streams, each with its own fractionator, because the fractionation of the two different streams, is expected to be different and the currently available models of primary sedimentation and thickening do not account for this. For example, the ISS/TSS ration is expected to be significantly higher in the primary sludge than the settled sewage, whereas the primary clarifier models in WEST assume that all the particulate components have the same settling characteristics. The thickened sludge composition can be calculated from the differences in the measured raw and settled sewage fluxes. Note that the thickener overflow is returned to the balancing tank (Figure A.1) and the settled sewage samples are collected after the balancing tank so settled sewage fluxes include the contribution of the thickener overflow.

The preliminary modelling exercise only considered the activated sludge plant so only the fractionation results for the AS system will be presented in this report. Initial calibration and tuning of the settled sewage fractionator was undertaken using in the Excel version of the fractionator and the optimised parameter values (measurement correlation parameters in Section 2.4) obtained were then transferred to the WEST fractionator.

Table A.2a lists the fractionator parameters used in the WEST settled sewage fractionator.

Table A.2a: Preliminary settled sewage fractionation parameters

Parameter		Description	Default value
f_oho	0.029	Fraction of total COD contributed by OHOs (ordinary heterotrophic organisms)	0.1
f_tss	0.38	Ratio of TSS to total COD	1.05
f_codus	0.056	Un-biodegradable soluble fraction of total COD	0.055
f_codup	0.052	Un-biodegradable particulate fraction of total COD	0.13
f_tkn	0.10	Ratio of TKN to total COD (mg N/mg COD)	0.1
f_fsa	0.82	FSA ratio to TKN	0.75
f_vfa	0.00015	Fraction of total COD contributed by VFAs (taken to be acetic acid)	0.04
f_codf	0.30	Soluble fraction of total COD	0.2
f_tknf	0.088	Ratio of filtered TKN to total COD	0.05
f_tp	0.012	Ratio of total phosphorus to total COD (mg P/mg COD)	0.025
f_orthop	0.0063	Ratio of orthophosphate to total COD (mg P/mg COD)	0.01667
f_tpf	0.51	Ratio of filtered phosphorus to total phosphorus	0.02
f_iss	0.047	Inorganic fraction of total suspended solids	0.15

The other critical inputs for the WEST fractionator are the upper and lower bounds for the component concentrations which constrain the fractionator solutions. The current WEST PWM_SA fractionator requires absolute values for the minimum and maximum component values whereas the Excel fractionator used in the case study used relative limits, specifically: 0.5 > fitted/inferred measurements > 2. The inferred measurements are the actual measurements if available, else the estimates calculated using the fractionator parameters in Table A.2a. The upper and lower limits for the WEST fractionators were selected using the observed range of fitted values in the Excel solution as a guide. The results are summarised in Table A.2b.

Table A2.b: Settled sewage fractionator bounds

Component	Observed range	Lower limit	Upper limit	Default Range
Ammonia (s_NH)	7.5-47	5	60	10-60
Volatile fatty acids (s_VFA)	0.02-0.09	0	10	0-10
Orthophosphate (s_PO4)	0.31-13	0	15	3-15
Soluble unbiodegradable organics (s_U)	4.5-26	3	60	10-60
Soluble fermentable organics (s_F)	15-228	10	500	100-500
Influent unbiodegradable particulate organics (x_U_Inf)	3.8-20	2	400	40-400
Ordinary heterotrophic organisms (x_OHO)	0-14	0	20	2-20
Influent biodegradable particulate organics (x_B_Inf)	32-286	20	900	80-900
Inorganic suspended solids (x_ISS)	2-28	0	500	80-500

Figures A.3a and A.3b shows the fractionator output in terms of agreement between measured and fractionated settled sewage TSS and COD, and the calculated distribution of the organic and particulate components. Figure A.3a also shows the settled sewage flow. Note that the average settled sewage flow for the period 8 May to 24 June was 65.9 ML/d so the plant was operating at just above its hydraulic capacity, although flows were much higher at other times of the year.

There is a good agreement between the TSS measurements (inputs to the fractionator) and the calculated output (Figure A.3a. The fractionator, has more difficulty fitting the large fluctuations in COD especially between 26 May and 13 June. Note that gaps in the measurements correspond to weekends. The fractionator estimates what these values should be either by extrapolating the loads or based on any other available measurements.

Figure A3.b shows that the fractionator calculates that most of the COD is in the form of biodegradable influent particulates (X_B_Inf) and soluble fermentable COD (S_F). X_B_Inf also makes up the bulk of the particulates. Other COD fractions (S_VFA = volatile fatty acids, S_U = soluble unbiodegradable COD component, X_U_Inf = unbiodegradable influent particles) and the inorganic particulate fraction (X_ISS

= inorganic suspended solids) are present in concentrations of less than 20 mg/L each. This is because the fractionator parameters f_codus, f_codup, f_vfa and f_iss are small.

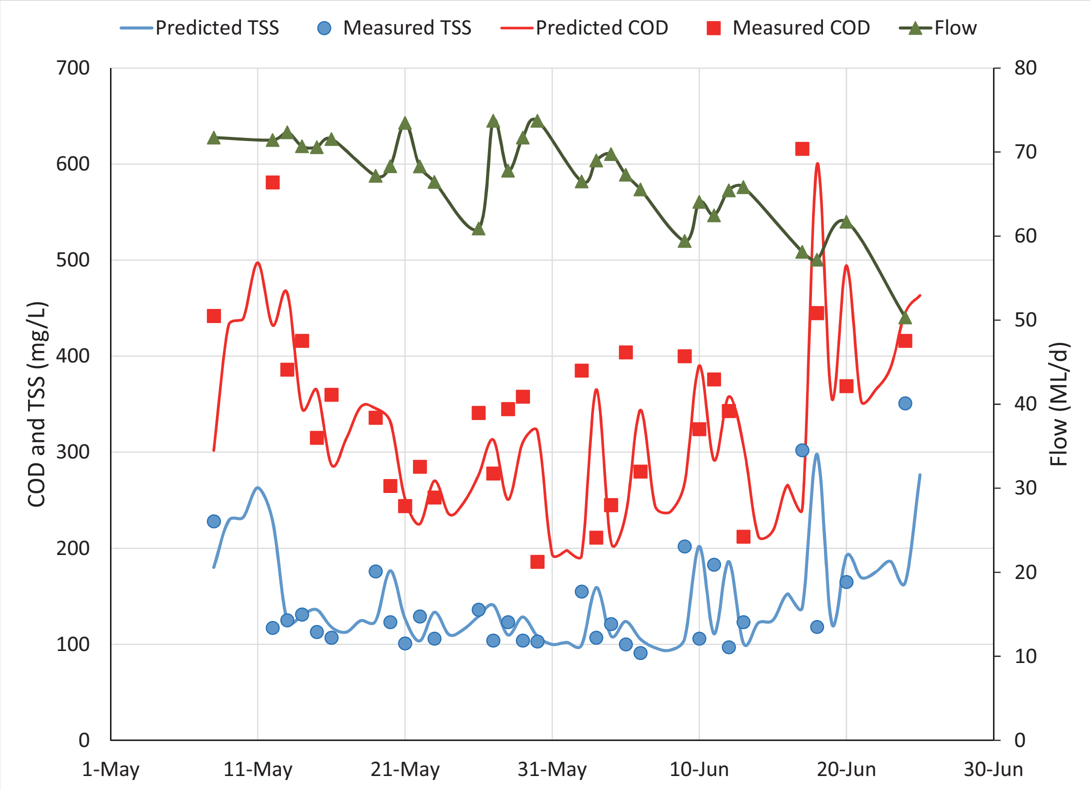


Figure A.3a: Settled sewage flow and measured vs predicted COD and TSS

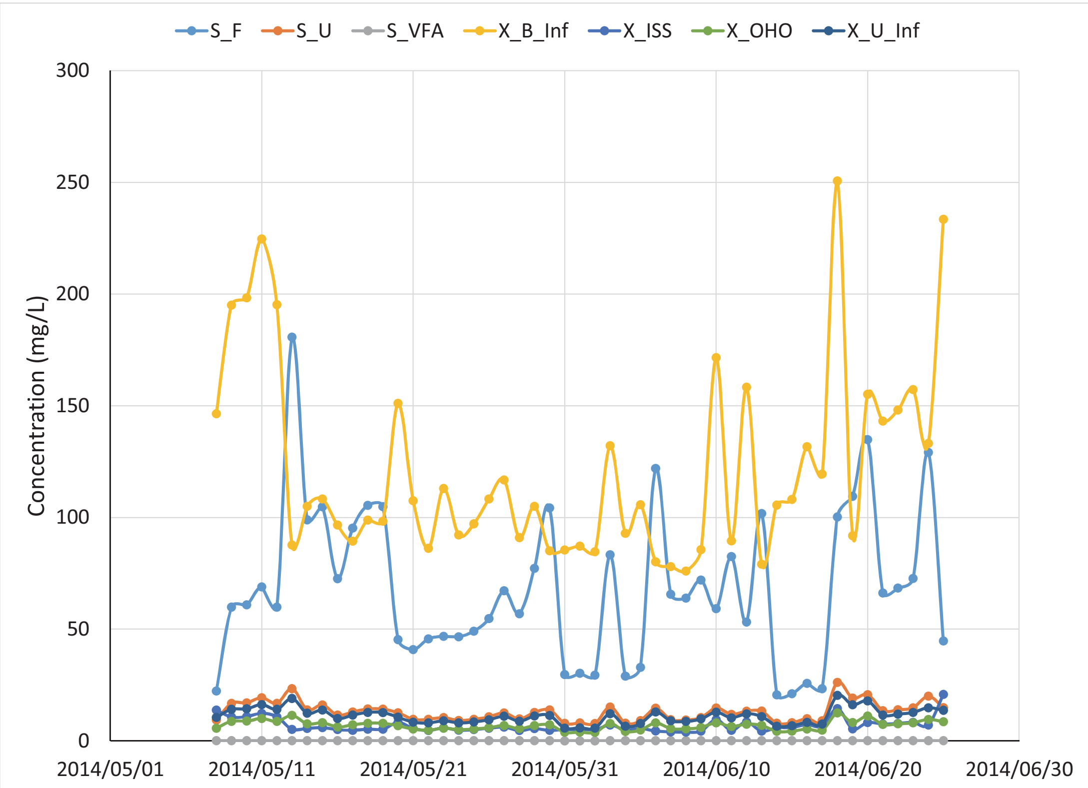


Figure A.3b: Fractionated settled sewage

Figure A.4a compares the agreement between measured and modelled TKN and FSA in the settled sewage while Figure A.4b compares the modelled and measured TKN/COD and FSA/TKN ratios with the corresponding fractionator parameters from Table A.2 (f_tkn and f_fsa respectively).

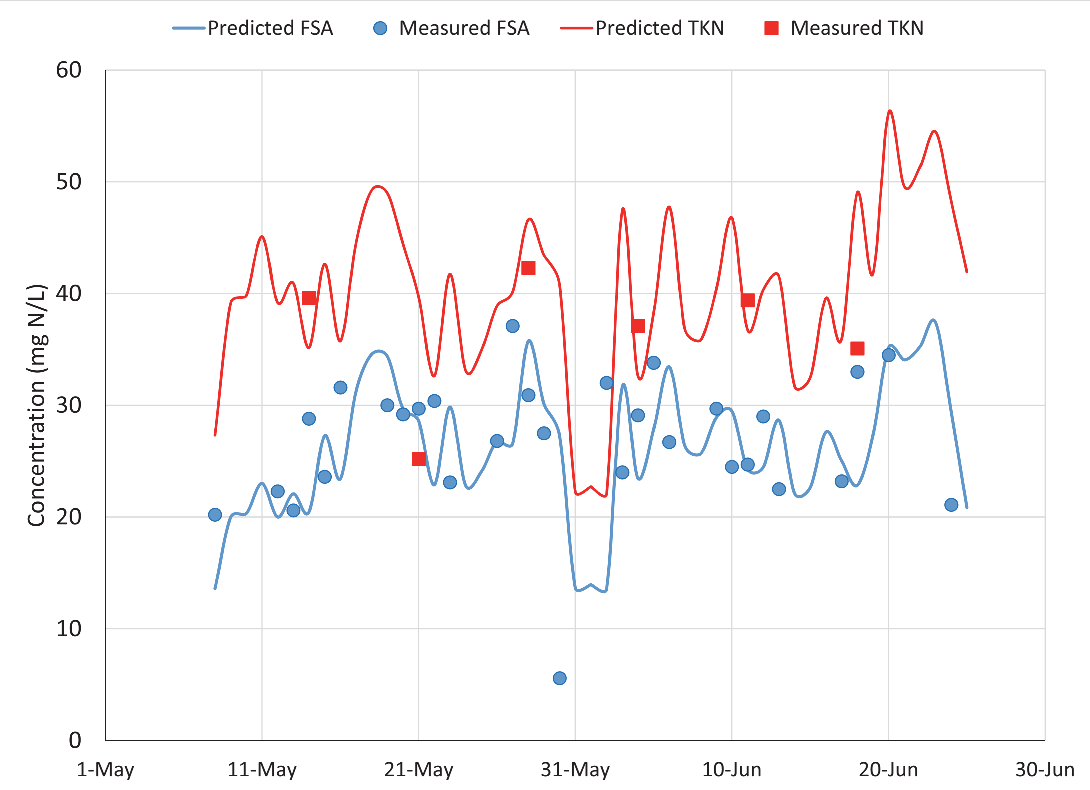


Figure A.4 (a): Measured vs predicted settled sewage TKN and FSA

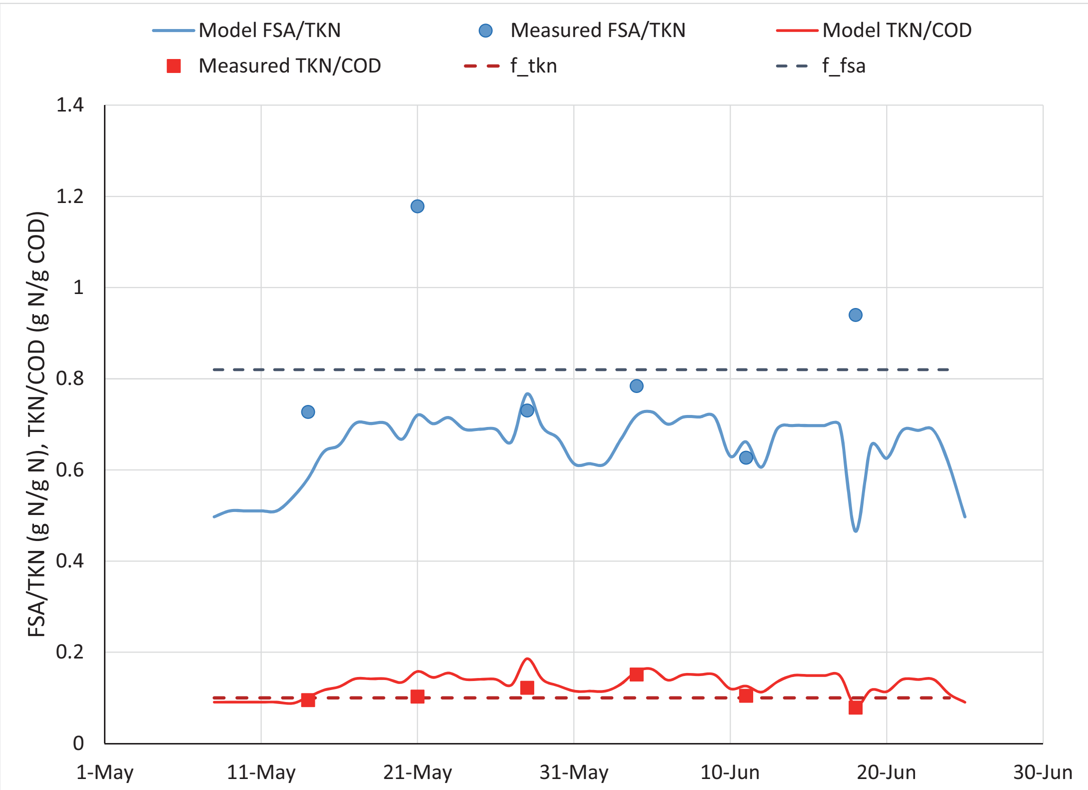


Figure A.4 (b): Modelled vs measured TKN/COD and FSA/TKN ratios

Since there are few actual TKN measurements, the fractionator is generally trying to fit the model results to a TKN estimate based on the COD and FSA measurements. However, since estimates are weighted less than measurements, in practice, the predicted TKN may be primarily determined by the TKN content of the model components which give the best fit to the TSS and COD data. Figure A.4b shows that the predicted TKN/COD and FSA/TKN ratios are respectively higher and lower than the ratios calculated from the data, suggesting that the fractionator may be over estimating the TKN content of the settled sewage.

A.4 WEST model set up

The model plant layout is shown in Figure A.5.The details of the model set up are explained below. The influent is the settled sewage leaving the balancing tank, and the output streams are the secondary effluent and thickened waste activated sludge. The settled sewage is represented as two input blocks, one for daily measurements (flow) and) and one for more intermittent measurements (COD, TSS, FSA, orthophosphate, pH, TKN) because different interpolation settings were used in each set. Both inputs feed into the probabilistic fractionator which calculates the wastewater composition in terms of model components.

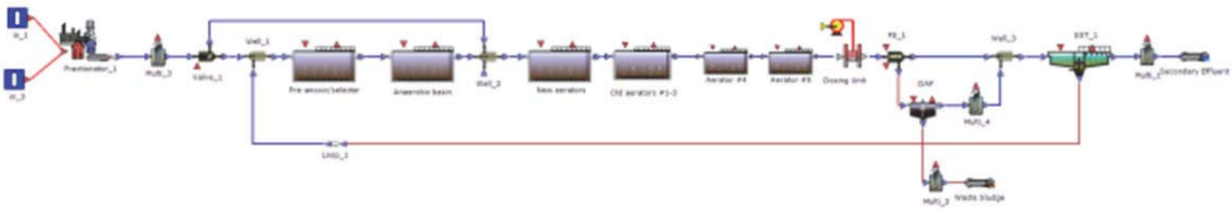


Figure A.5: WEST PWM_SA model set up for Darvill WWTP activated sludge plant

A.4.1 Influent splitting and Anoxic/Anaerobic Selector

The settled sewage is split between the anoxic/anaerobic and aerated reactors using a relative two way splitter with 96% of the flow going to the aeration basin. The return activated sludge (RAS) is returned to the anoxic/anaerobic basin where it is combined with 4% of settled sewage flow. As shown in Figure A.1, the anoxic/anaerobic basin is essentially a baffled channel in four sections. In the model, the first two sections are modelled as a 1 360 m^3 unaerated activated sludge unit (Pre-anoxic/selector) while the last two sections are the 2040 m^3 anaerobic basin.

A.4.2 Activated sludge basins and aeration

The 16 200 m^3 aerated reactor is divided into three parallel channels (A, B and C) as shown in Figure A.2.The original plant design included the facility to raise and lower the first three 75 kW aerators in each channel to vary the aeration rate to achieve a DO set point of 2 mg O/L at the end of each channel. However, in 2014, all aerators including the new 30 kW aerators had to be continuously operated at maximum rate with the 2 mg/L target seldom being achieved.

Since the aerators were all operated continuously at full power, the aerated basin was modelled as a series of activated sludge units (ASUs) with fixed mass transfer coefficient (Kla). Table A.3 summarises the Kla calculations at an average temperature of 20°C. The alpha factors and standard aeration efficiencies (SAE) values were provided by Umgeni Water. The SAE values are typical of low speed aerators (1.5-2.1 kg O_2/kWh, Table 9.2 in Stenstrom and Russo, 2008).

Table A.3: Calculation of initial estimates of aerator Kla's

	Rated power, kW	Alpha factor	Standard aeration efficiency (SAE), kg O_2/kWh	Total volume, m^3	Kla*, 1/d
New aerators	9 x 30	0.55	1.8	3300	164
Old aerators	15 x 75	0.85	1.7	16200	203

* average assuming 80% line power efficiency, 20°C and operating oxygen concentration 0-2.86 mg/L.

Table A.4 lists the model reactor volumes and initial estimates of the Kla's for the activated sludge plant. The aerated basin is modelled as four ASUs in series

i. The first section of the aerated basin where the new aerators have been installed.
ii. The next three sections including the first three rows of the old aerators are modelled as a single unit.
iii. The last two sections including the last two rows of aerators are each modelled as a separate reactor.

The last two sections of the aerated channels are modelled as separate units to facilitate the comparison of model results with DO measurements for these sections. The Old Aerators #1-3 were originally modelled as three separate activated sludge units (ASUs), however, this substantially increased the computational time for the simulations, hence the decision to represent them as a single unit.

Table A.4: Volumes and preliminary aerator parameters in the activated sludge train

Unit	Volume	Kla, 1/d
Pre-anoxic basin/Selector	1360 m^3	0
Anaerobic basin	2040 m^3	0
New Aerators	3300 m^3	164
Old Aerators #1-3	9720 m^3	203
Aerator #4	3240 m^3	203
Aerator #5	3240 m^3	203

A.4.3 Secondary settler

In the preliminary model, the secondary settlers are modelled as a point settler with the underflow set at 60 000 m^3/d. The fraction of suspended solids which is non-settleable, f_ns, is set to match the effluent TSS. An initial estimate of 0.007 was used based on the average effluent TSS/MLSS ratio.

A.4.4 Sludge wasting rate

Waste activated sludge (WAS) is drawn from the end of the aeration basin and sent to the DAF units for thickening. The WAS flowrate is related to sludge age (uncorrected for solids in the clarifier effluent) as follows:

Sludge age (uncorrected) = total reactor volume/wasting rate

An average wasting rate of 2 100 m^3/d corresponding to an average uncorrected sludge age of 10 days was estimated.

A.4.5 DAF

The DAF plant is modelled as a simple efficiency thickening unit with a concentrate flow of 225 m^3/d and a solids removal efficiency e_X = 0.99 (99% of the solids goes to the concentrated sludge). The concentrate flow was selected based on a float volume of 150-300 m^3/d (p. 103 in the Darvill WWTP Operating Manual). The removal efficiency was selected to give a subnantant TSS concentration between 20-70 mg/L.

A.4.6 Alum dosing

The PWM_SA model does not currently include alum dosing for phosphorous removal so this aspect of the plant's performance could not be modelled. However, in addition to precipitating out phosphorous, alum dosing results in the addition of a significant amount of inorganic suspended solids to the process, increasing the overall reactor solids (MLSS). This aspect was modelled by adding a generic dosing unit which added ISS to the outflow from the aerobic basin.

When alum is added, phosphorous precipitates out as $AlPO_4$. Alum for phosphorous removal is typically dosed at a ratio of 1 to 3 metal ion per phosphorous on a molar basis to achieve effluent phosphorous ≥ 0.5 mg/L. Higher doses are required to achieve phosphorous residuals < 0.5 mg/L (Tchobanoglous and Burton, 1991,p. 745). Alum dosing data was not provided however, measured effluent orthophosphate at Darvill was typically < 0.2 mg/L so Al:P dosing ratios were expected to be higher than 3:1.

Excess Al is assumed to precipitate as $Al(OH)_3$. Therefore if x moles Al is added per mole of orthophosphate, the amounts of precipitate formed are (1-x) moles $Al(OH)_3$ per mole $AlPO_4$. The calculations of the ISS added are summarized in Table A.5. The calculations are based on an assumed orthophosphate removal of ~ 2 mg P/L which is the average difference between the measured and modelled orthophosphate concentrations in the effluent, an Al:P ratio of 3:1 which was the starting estimate, and a settled sewage flow of 63.6 ML/d.

Table A.5: Alum dosing calculations for Al:P = 3:1

Orthophosphate removed: 2 mg P/L = 0.65 mmol/L Average settled sewage flow: 63.6 ML/d RAS flow: 60 ML/d Total outflow from the AS basins: 123.6 ML/d	Molecular weights $AlPO_4$: 122 g/mol $Al(OH)_3$: 78.0 g/mol Alum ($Al_2(SO_4)_3.18H_2O$): 666 g/mol
Precipitate formed, mg/L $AlPO_4$: 0.0647 mmol/L = 7.89 mg ISS/L $Al(OH)_3$: 0.156 mmol/L = 12.2 mg ISS/L Total ISS added = 20 mg/L	Assuming dosing flow = 5 m^3/d Dosing solution concentration = 495 000 mg ISS/L Equivalent alum dose = 73 mg/L

A.5 Model Calibration and Preliminary Results

Once the model is set up, the next step is to run the simulation and compare the results to the available data to determine whether the model predictions are reasonable. Three aspects of the model performance were considered:

(i) Predicted vs measured reactor solids

(ii) Predicted vs measured effluent COD

(iii) Predicted vs measured dissolved oxygen profile and its impact on predicted vs measured effluent ammonia and nitrate

PWM_SA does not currently include phosphorous removal by precipitation with alum so this aspect of the plant's performance was not considered. Several different runs were carried out where different parameters were adjusted to try to improve the model's fit to the available measurements. The results of the following runs are discussed in this section:

Table A.6: Summary of dynamic simulation experiments

Run 1	Used initial parameter values presented in Sections A.3 and A.4
Run 2	ISS addition increased to 50 mg/L to better fit MLSS. (Equivalent alum dose of 182 mg/L) Kla's decreased to low values to better fit June effluent FSA data. Kla (new) = 127 /d Kla(old #1 - #3) = 114 /d Kla (old #4) = 107 /d Kla (old #5) = 67 /d
Run 3	Kla's increased to intermediate values to better fit May effluent FSA data. Kla (new) = 144/d Kla(old #1 - #3) = 130/d Kla (old #4) = 121 /d Kla (old #5) = 92/d

A.5.1 Reactor solids

Figures A.6a and A.6b show the calculated vs measured aerobic reactor mixed liquor suspended solids (MLSS) from Runs 1 and 2 as well as the particulate fractions calculated by the model.

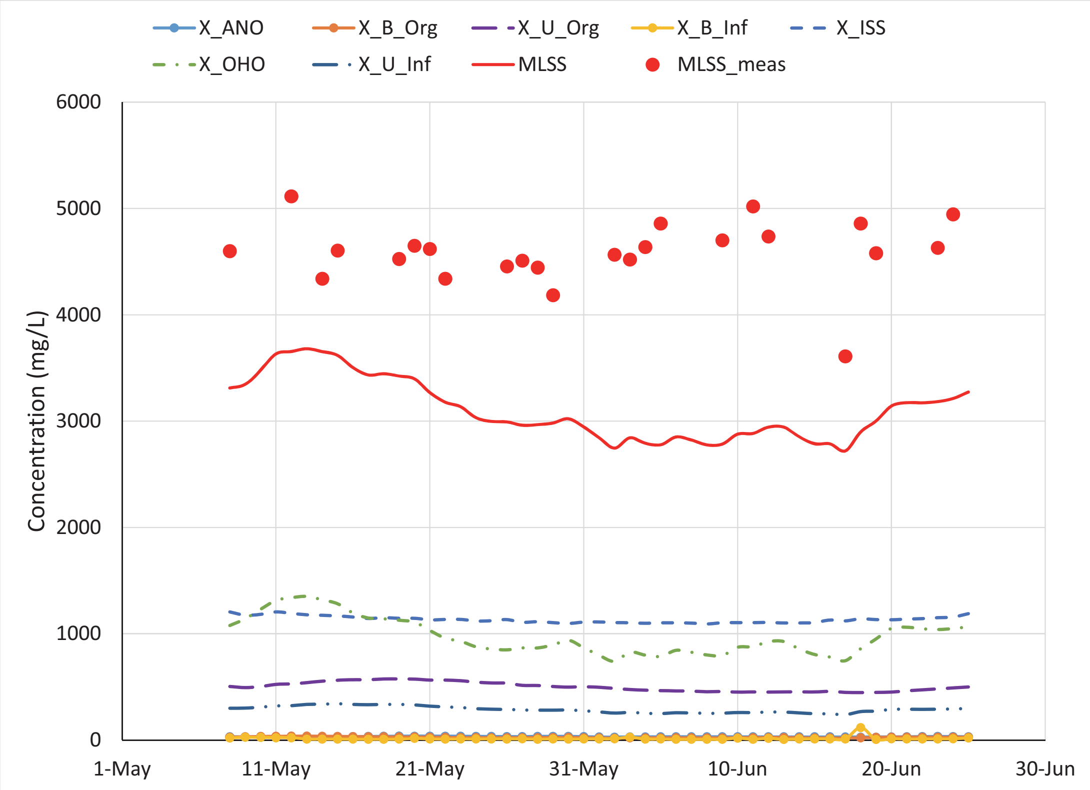


Figure A.6a: Reactor particulates Run 1

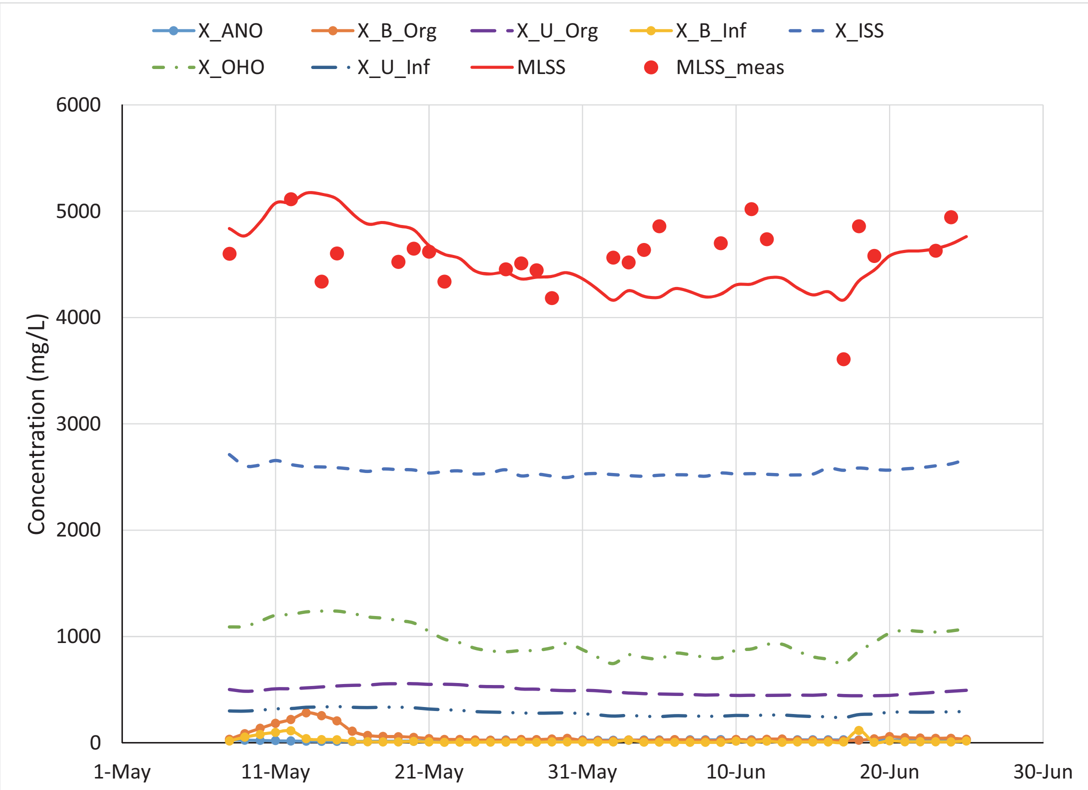


Figure A.6b: Reactor particulates Run 2

Note that in Run 2, ISS, which in this case is mostly alum sludge, has increased to more than half of the aerobic reactor particulates. It is possible that high alum doses are being used to achieve the low effluent orthophosphates observed and also because the plant operating conditions (short sludge ages and low dissolved oxygen levels) tend to produce poorly settling sludge (Darvill Operating Manual) however, the actual alum dosing levels need to be determined to confirm this result.

A.5.2 Effluent COD and TSS

Figures A.7a and A.7b compare the effluent COD and TSS for the two different alum doses (Run 1 and Run 2) with the available measurements.

As in the Phoenix case, a point settler model was used for the secondary clarifiers and so the predicted effluent TSS is a fixed fraction (0.7% in this case) of the MLSS. This simplistic model obviously cannot predict periods of clarifier upset as appeared to occur in June, however, the average performance used to calculate the non-settleable fraction does not appear to be representative of the clarifiers performing well either. Some of the scatter in the measured data may be due to sample timing but it appears that a better clarifier model is required. WEST does include clarifier models which can account for variations in sludge settlability and some settlability data is in fact available for Darvill. However, more detailed information on underflow rates and alum doses is probably also required.

The model effluent COD consists primarily of the particulate component x_OHO (ordinary heterotrophic organisms) and the soluble unbiodegradable fraction (s_U), calculated by the influent fractionator, which is assumed to pass through the plant unchanged. The fluctuations in the predicted COD are primarily due to the s_U fraction.

One interesting aspect of Figure A.7b is that the effluent COD and TSS are numerically close in value. In the absence of metal salts addition, it is usually reasonable to assume that the ISS content of the secondary effluent is very low and the effluent TSS consists mainly of organic particles such as x_OHO which have an fcv ratio (COD/TSS) ~ 1.46. These assumptions can be used to estimate the s_U in the influent from the effluent data. This approach is incorporated into some versions of the Excel fractionator, for example, the fractionator for the Umhlanga case study and was also used in the Phoenix case study.

It was not possible to calculate s_U from the effluent measurements in the Darvill case study where the average COD was only 36 mg COD/L compared to 32 mg TSS/L, so s_U was calculated by the fractionator using only influent data. This can be explained in terms of the high ISS content of the mixed liquor since ISS contributes TSS but COD. Therefore these results tend to support the conclusion that a substantial fraction of the activated sludge is actually alum sludge. However, it should be pointed out that if there is good agreement between measured and modelled effluent COD but poor agreement between measured and modelled TSS, as is the case for the late May-early June data, the measurements do not support the predicted effluent composition (COD and particulate fractions) and may also indicate errors in the influent fractionation.

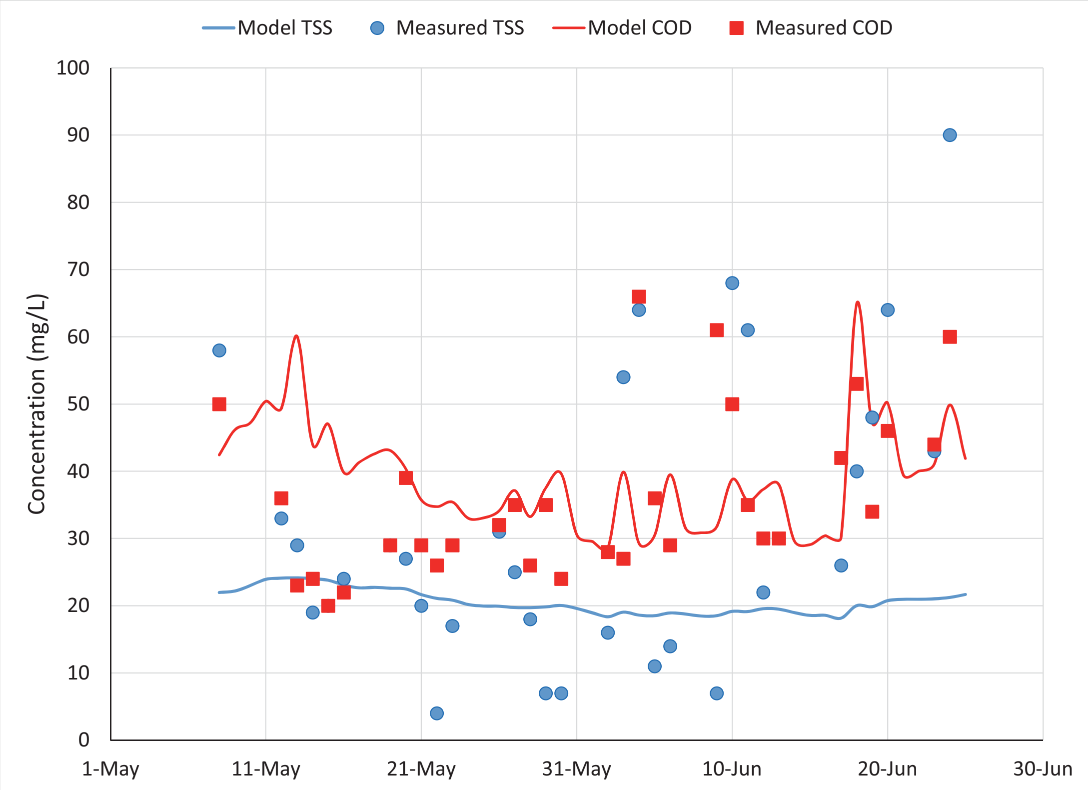


Figure A.7 (a): Run 1 effluent COD and TSS

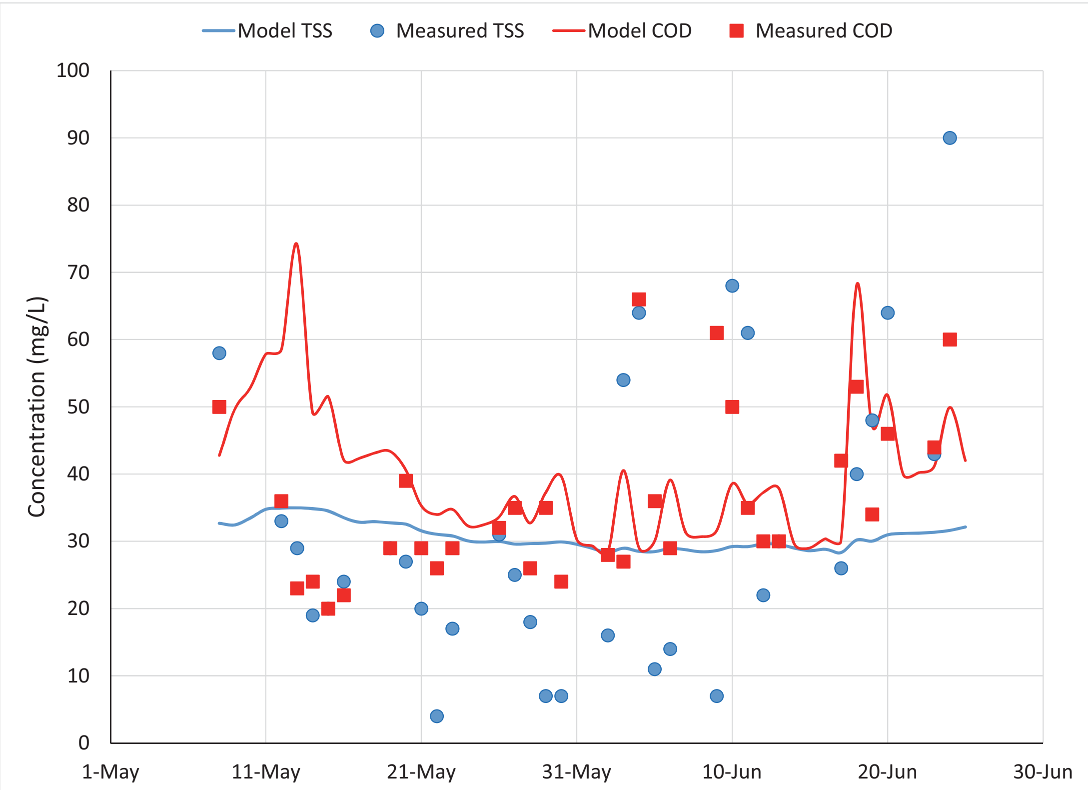


Figure A.7 (b): Run 2 effluent COD and TSS

A.5.2 Dissolved oxygen and nitrification

As noted previously, loads to Darvill generally exceeded its design capacity during the study period and consequently the aerators were all run continuously at full power. Furthermore, the Umgeni staff reported that 2 mg/L DO set point was rarely achieved for any length of time. It was therefore expected that the model would show that the dissolved oxygen levels vary with load. However, this turned out to be very difficult to simulate.

For the purposes of the modelling exercise, it was assumed that the power input and mass transfer coefficient Kla, remained constant for each basin (section A.4.2). However, this was probably not the case since, in reality, Kla varies with process conditions and mixed liquor composition (Stenstrom and Rosso, 2008).

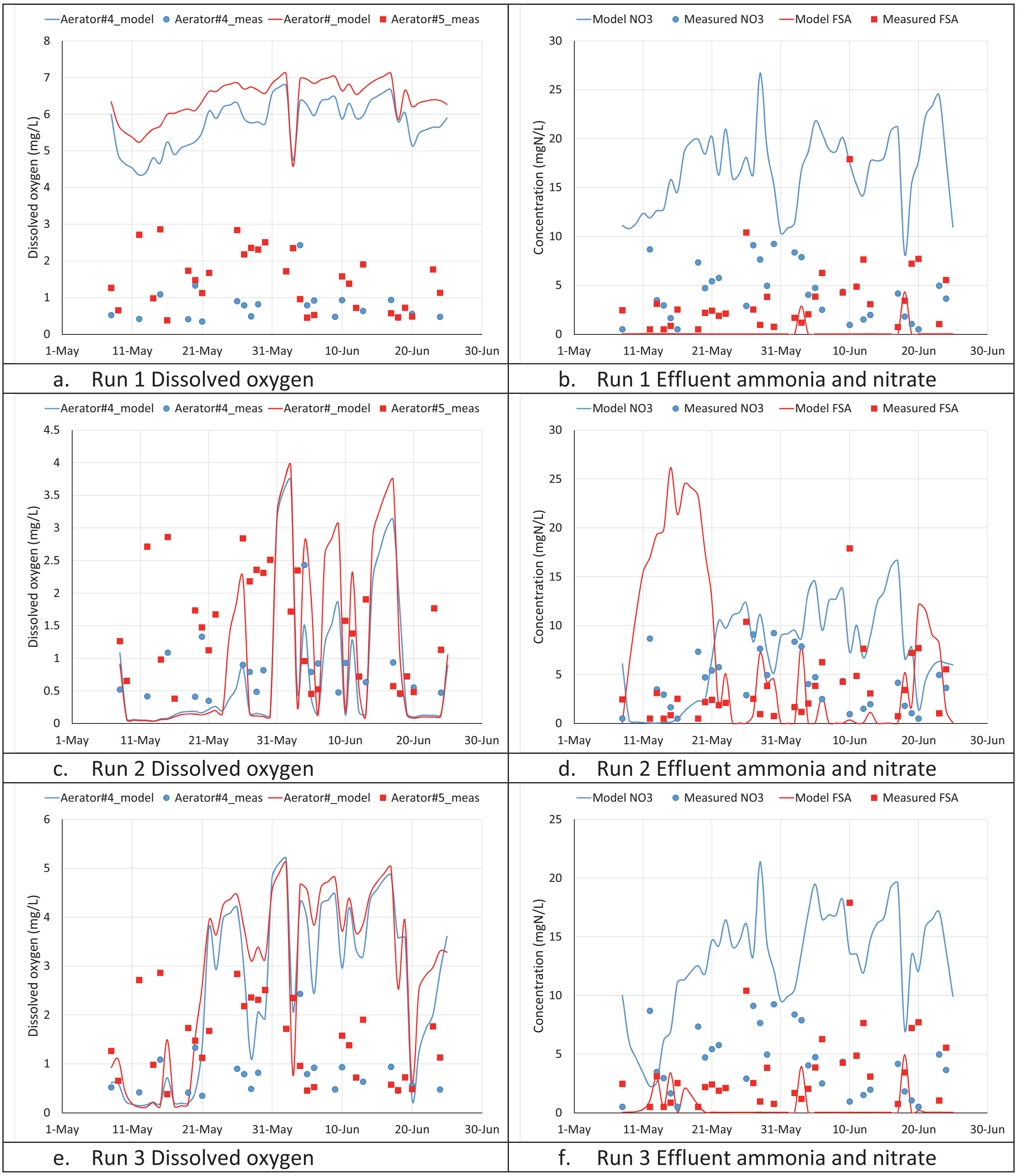
Figures A.8a to A.8f compare the predicted dissolved oxygen and effluent nitrate and ammonia with the available measurements. Effluent free saline ammonia (FSA) is very sensitive to low dissolved oxygen levels and therefore provides a check on the calculated DO profiles with spikes in the predicted effluent ammonia corresponding to DO < 0.5 mg O/L and very low effluent ammonia for higher DO values.

In Figure A.8a, the predicted DO's for Run 1 are 2-4 mg O_2/L greater than the measurements indicating that the initial estimates of Kla were probably too high. It should be pointed out that the dissolved oxygen measurements were not necessarily representative of the average daily values (they were single measurements recorded around 10 a.m. on weekdays). However, the high predicted DO levels in Figure A.8a are not consistent with the observations of the plant staff. Furthermore, there is also no agreement between the measured and predicted effluent nitrogen (Figure A.8b). Factors which could contribute to lower oxygen transfer rates include the presence of organic compounds with surfactant properties, particularly oil, and reduced aeration efficiencies in older equipment.

Figures A.8d to A.8f show the effect of trying to adjust the predicted DO levels down in Runs 2 and 3. Fitting the dissolved oxygen predictions by adjusting the Kla's proved very difficult since the relationship is highly non-linear, therefore the focus was on trying to get a better fit with the effluent ammonia. Run 2 provided a somewhat better fit to the ammonia data after May 27th while Run 3 attempted to fit the model to data for the week of May 11th. What is clear from Figure A.8 is that it is not possible to get a good fit to the entire data set using a single set of mass transfer parameters, i.e. the Kla's appeared to vary over the study period.

Another possibility is that the settled sewage measurements and fractionation were not sufficiently representative of the average load, particularly with respect to total oxygen demand and nitrogen content. The timing of the grab samples is one issue, however, it is also not clear how and if excess FOG reaching the activated sludge basin would be reflected in the available measurements. Further wastewater characterisation studies would be required to resolve these issues, which was beyond the scope of the current project.

The effluent nitrate depends on the balance between nitrification and denitrification. As in the Umhlanga case study, the only pumped recycle is the return activated sludge and Darvill relies on backmixing due to the aerators to recycle nitrate to the anoxic zone. As discussed in Chapter 3, the stirred tank models used for the activated sludge units are not designed for this scenario so one would probably have to add recycle loops to the model to get the denitrification right.



a. Run 1 Dissolved oxygen

b. Run 1 Effluent ammonia and nitrate

c. Run 2 Dissolved oxygen

d. Run 2 Effluent ammonia and nitrate

e. Run 3 Dissolved oxygen

f. Run 3 Effluent ammonia and nitrate

Figure A.8: Dissolved oxygen and effluent ammonia and nitrate

A.6 Discussion

Various other adjustments to the model and influent fractionation were attempted including increasing the weighting of the TKN estimates, increasing the fraction of ISS and S_F in the influent, and adjusting the stoichiometry of X_B_Inf to decrease its nitrogen content (contribution to TKN). However, as in the case of the aeration mass transfer coefficients, adjusting any of the other model parameters improved the fit in some sections of the data while causing it to deteriorate in others.

In summary, it proved impossible to calibrate the model for this particular data set, at least with a single set of parameters. Therefore, the model can also not be used with any confidence to predict the plant behaviour in other scenarios, such as in risk assessment exercises. It was therefore decided to not to pursue this case study any further for the purposes of the current project. Nevertheless, this case study was instructive in terms of the modelling challenges it highlighted:

A.6.1 Type and quality of influent data available

As discussed in the Phoenix case study, the data requirements for modelling are substantially greater than that of compliance monitoring and this can be a significant barrier to model development and calibration. In the Phoenix case study, raw water data was available as three times weekly 24 hour composite samples whereas at Darvill, composite sampling was conducted only once every week or two so the daily grab sample data was used instead.

Composite samples generally provide a better estimate of the daily plant loads for the purpose of model calibration than a single daily grab sample since the concentration of the various determinands in the influent can easily fluctuate by 100% or more during a typical day of operation. The presence of the equalization tank at Darvill would have damped out but not eliminated these fluctuations. For example, Figure A.3a shows large fluctuations in settled sewage COD from one day to the next and it is not clear if this indicates rapid changes in the daily loads or simply samples taken at different times of day. It should be pointed out, however, that composite sample data may be no more reliable than grab sample data if scrupulous sample handling and storage procedures are not adhered to.

This is one of the major reasons that steady state modelling is so much easier to implement than dynamic modelling: in the steady state model, only the average loads are used, and if there are sufficient data points, the averaging process will take care of the uncertainty associated with fluctuations and outliers in the raw data.

The availability and frequency of grab and composite samples will vary from plant to plant depending on plant size, staffing levels, Department of Water Affair's authorisation and the organisation's standard practices, and may affect the modelling goals which can be achieved. The UCT group has been developing a method for estimating daily loads from grab sample data as part of the PWSSD programme (Section 2.2) however this functionality was not yet available at the time of writing this report, and would be much more difficult to apply in a dynamic model.

A.6.2 Oxygen transfer

The efficiency of many of the biological processes is strongly dependent on the dissolved oxygen levels, especially when they drop below 1-2 mg O/L and it is therefore important to get this part of the model right. The simplest case to model is when there is active control of the aerators to maintain the dissolved oxygen levels close to the set point. However, the scenario where the aerator power is (assumed) constant and dissolved oxygen varies with influent total oxygen demand turns out to be very difficult to model. The predicted DO levels are very sensitive to the values assumed for the oxygen transfer coefficients (Klas) and these mostly likely also vary with process conditions. This is obviously going to be a recurring problem when attempting to model plants that are overloaded in terms of the aeration capacity. Many other plants do not have proper DO monitoring and control which will also be a challenge for the models.

A.6.3 Process changes

The plant is no longer operating as it was originally designed or as explained in the operating manual, for example, most of the settled sewage bypasses the anaerobic selector, aerators have been installed in the anoxic zone and phosphorous removal is being achieved using chemical precipitation. This makes it harder to make reasonable assumptions about the plant operation and performance when data is missing.

A.6.4 Process variability and extreme conditions

Processes with highly variable performance are inherently more difficult to model than processes that are relatively stable. Extreme situations are more likely occur which may fall outside the range of validity of the various models and typical modelling assumptions and may also negatively impact model stability. One difficulty encountered in this case study was the behaviour of the phosphate accumulating organisms (PAOs). The process conditions did not favour the development of a robust population of PAOs and the model did indeed show the population dropping to very low levels. The problem was that predicted concentration eventually become negative causing the rest of the simulator solutions to become unstable unless the initial estimates of the PAOs were carefully adjusted at the beginning of each run.

A.6.5 Uncertainties in the influent fractionation

The probabilistic fractionator was developed as a tool to deal with gaps in the data available on the influent composition. If one is able to achieve a reasonable fit between the model and the available measurements, this provides some assurance that the influent fractionation is approximately correct. However, it must be pointed out that the current versions are not designed to deal with uncertainties relating to diurnal variations and grab sample timing.

In the Phoenix case study, which involved a purely domestic wastewater and used composite samples, the fractionator seemed to perform quite well. In the Darvill case study, there was significant uncertainty in both the influent composition and process parameters, so the fractionator performance is difficult to assess. It did appear, however, that it may have been over predicting the settled sewage TKN which would have impacted the nitrogen removal results.

The fractionator tool defaults (set of components, stoichiometry and typical fractions) have been set up based on wastewater characterisation data for typical domestic wastewaters and may not handle wastewaters with a significant industrial component as well. It may therefore be necessary to adjust the PWM_SA components' stoichiometry or possibly even define new industry specific components with their own stoichiometries and degradation kinetics for some plants receiving industrial effluent. This would of course require a substantial modelling effort as well as additional measurements and would not be feasible as part of an audit.

In Darvill's case, a particular concern is the high oil content of the raw water. While domestic wastewater always contains some level of FOG, edible oil industry effluent is a significant fraction of the Darvill influent meaning that oil levels will be higher than typical and quite variable depending on what is happening at the industrial facilities. Most of the oil in the raw water is supposed to be skimmed off in the primary settlers but when concentrations are high in the raw water, some may break through into the settled sewage. Vegetable oil has a higher COD/g and zero nitrogen content compared to the default PWM_SA component stoichiometries as shown in Table A.7.

Another problem with FOG in the influent is that it does not mix well with water so getting representative samples may be challenging. Therefore, in addition to the uncertainties about the sample timing, there is some uncertainty about whether the available COD measurements accurately reflect the COD load at the time they were measured. All of this could have contributed to the difficulties in calibrating the model.

Table A.7: PWM_SA COD components and vegetable oil stoichiometry

	s_U	s_F	x_U_Inf	x_OHO	x_B_Inf	VFA	Vegetable Oil
H/C	1.64	1.99	1.63	1.42	2.27	2	1.85
O/C	0.667	0.604	0.517	0.521	0.583	1	0.130
N/C	0	0.109	0.00237	0.0601	0.135	0	0
P/C	0.0242	0.0113	0.000343	0.0226	0.00406	0	0
MW	25.1	25.6	22.0	23.3	25.6	30.0	15.9
COD/g (fcv)	1.41	1.41	1.67	1.48	1.47	1.07	2.81
N/g (fn)	0	0.0598	0.00151	0.0361	0.0736	0	0

The definitions of the PWM_SA components were provided in Table 2.2. The bulk of the COD in the influent is typically made up of S_F and X_B_Inf as discussed in Section A.3.

A.6.6 Phosphorous removal by precipitation with alum

As discussed in the previous sections, PWM_SA does not currently include dosing with alum. In this case study, the contribution of the alum addition to the inorganic suspended solids was included but the precipitation of phosphorous and its effect on the biological processes and the alkalinity could not be modelled. PWM_SA already includes a speciation sub-model with other precipitation reactions (Section 2.3.1.2), however, the element Al and its precipitates have yet to be added. This is something which can be done in the near future.

A.7 Conclusions and Recommendations

One of the objectives of this case study was to assess the application of the dynamic model to a plant which was overloaded and for which industrial effluent is a significant part of the its influent. While the modelling exercise itself was not successful, it highlighted a number of important challenges relating to modelling such plants as well general issues relating to the type of data available, which need to be addressed in future work.

- The available plant data will not always sufficient for model calibration even with the use of the probabilistic fractionator.
- Model calibration should ideally be carried out using composite measurements for the influent. Since these are not always available, additional tools will have to be developed to estimate daily loads from grab samples if the propose modelling tools are to be widely applied and/or more detailed wastewater characterization studies will have to be carried out on a case by case basis.
- Oxygen transfer coefficients vary with process conditions therefore attempting to model an overloaded aeration system with fixed Kla’s does not work very well. More robust methods for modelling oxygen transfer in overloaded systems need to be developed.
- More work is required on the characterization and modelling of wastewaters with significant industrial effluent content. If the industrial contribution to a municipal wastewater is reasonably consistent, it may be sufficient to simply adjust the PWM_SA component

stoichiometries as necessary, however, if the industrial effluent loads are variable and significantly different in composition to the domestic contribution, then additional model components may have to be defined.

- Precipitation with alum should be added to PWM_SA. The speciation sub-model is already in place and the relevant components, species and reactions just need to be added. Chemical precipitation for phosphorous removal and improved sludge settleability is expected to be commonly practiced at overloaded plants.

Appendix B: Training Workshop

B.1 Summary

A series of six training sessions were conducted for personnel from eThekwini Water and Sanitation and Umgeni Water between May and August 2014. The theme of the workshops was to develop, calibrate use a model of a wastewater treatment plant, with the aim was to have a documented working model of an actual plant by the end of the course. A section of eThekwini's Northern WWTP was chosen as the case study, and 4 of the 6 sessions were conducted at the Training Centre located at the plant. The sessions took place at approximately 2 week intervals, to allow the participants to obtain data or resolve issues identified during the sessions.

B.2 Attendees

eThekwini Water and Sanitation	
Kaverajen Pillay	Kaverajen.pillay@durban.gov.za
Akash Singh	Akash.Singh@durban.gov.za
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Mluleki Mnguni	mluleki.mnguni@umgeni.co.za

B.3 Programme

Date	Topics
11 April 2014	Introduction and planning.
9 May 2014	Overview of ASM models; data requirements; assessment of the Northern WWTP.
21 May 2014	Catchment data and plant data leading to influent characterisation.
6 June 2014	Characterisation of influent wastewater.
23 June 2014	Calibration of influent COD fractionation.
10 July 2014	Calibration of N removal; sensitivity analysis; sludge age calculation
To be scheduled	Assessment of additional assignments

Appendix C: Dissemination Workshop

C.1 List of attendees

Akash Singh	Akash.Singh@durban.gov.za	EWS
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Wade Swannell	Wswannell@hatch.co.za	Hatch_Goba
Jason Browne	JBrowne@hatch.co.za	Hatch_Goba

C.2 Presenters

Chris Brouckaert (CJB): University of KwaZulu-Natal (UKZN)
Nkosinathi Buthelezi (NB): eThekwini Water and Sanitation (EWS)
Barbara Brouckaert (BMB): UKZN
Akash Singh (AS): UKZN/EWS
William Wu (WW) University of Cape Town (UCT)/ Aurecon

C.3 Agenda

Time	Item	Presenter/moderator
10:00	Welcome and Introduction	CJB
10:15	Report back on recent Green Drop seminar	NB
10:30	Steady state model overview	WW
10:45	Discussion / Tea	
11:15	The case study plants (Phoenix & Umhlanga)	AS
11:30	Fractionator overview	CJB
11:45	Umhlanga case study	BMB
12:15	Phoenix case study	AS
12:45	Lunch	
13:30	Potential roles for modelling in a Green Drop audit	AS
13:45	Discussion: • Opportunities and barriers • Institutional capacity issues • Administrative framework • Future research	CJB
14:45	Conclusions and recommendations	CJB

C.4 Presentations

Nkosinathi Buthelezi opened with a report back on a recent seminar on the latest developments in the Green Drop programme. Key points of relevance to the topic of the workshop were that:

- The next audit would take place early in 2016, assessing the period 2014/2015
- The KPAs include KPA5: *Wastewater quality risk management* and KPA7: *Wastewater treatment capacity.*
- The Green Drop scoring provides for bonus points for partnerships with municipalities or other institutions which lead to improvements against one or more Green Drop criteria.
- There is a 10 year plan for ongoing development the Green Drop programme

William Wu presented an overview of the steady-state WWTP modelling tool developed at UCT.

Akash Singh gave an overview of the Phoenix and Umhlanga WWTPs that were the subjects of the case studies undertaken during the project. (He is the area engineer responsible for their operation).

Chris Brouckaert gave an overview of the wastewater *fractionator* developed during the project. This is a tool for determining the composition of the wastewater in terms of model components from routine measurements, which is a critical issue in setting up a model.

Barbara Brouckaert presented a case study of the Umhlanga WWTP, demonstrating the use of the UCT steady state model for capacity estimation.

Akash Singh presented a case study of the Phoenix WWTP, illustrating dynamic modelling using the WEST software from DHI (Danish Hydraulic Institute).

After lunch, **Akash Singh** presented suggestions about how modelling might be included in a Green Drop audit. The principle applications would probably be capacity estimation, and risk analysis. As an example of the latter, he presented a simulation of the impact of a belt-press failure on plant operation.

C.5 Discussion

The discussion was opened by Ayesha Laher, who has been a Green Drop inspector since 2007. She was excited about the potential of modelling to advance the Green Drop process, by expanding the set of tools available. However, modelling is complex, and requires skills that are unlikely to be available outside the major metros. It also is data hungry – many of the smaller municipalities simply do not record the necessary data at present. It would not be practicable for an auditor to carry out the modelling, since he or she only has 2 days for the entire audit. She was particularly excited about the potential for modelling to improve the quality risk assessments, since the whole thrust of the Green Drop programme is mitigation of risks. Models could allow a municipality to improve the level of planning for risk scenarios. So, an auditor would not be able to do the modelling, but would find the results invaluable. Having a collection of models would build up a set of data for WWTP design that is specific to South Africa, which would assist greatly in the future design of treatment plants.

Prof Buckley posed the question of what benefit would there be for the municipality, specifically within the context of a Green Drop audit. Ms Laher replied that the Green Drop in general did not provide incentives other than inspiring municipalities with a sense of ownership and achievement in the improvement of the effectiveness of their treatment plants. So this would apply to modelling also. A case could be made to DWS to include bonus points for a modelling, in order to encourage its adoption. Once it became more established, it weighting could be increased. This is similar to the process for introducing risk management and storm water management into the Green Drop programme.

Dr Naidoo comment that the purpose of the discussion was to establish what value the municipalities saw in modelling, and whether they wanted it taken up by their engineers. A motivation could then be made to regulator to introduce bonus point for modelling. There is a 10 year plan for Green Drop to give the municipalities a longer term view for planning. Modelling could be introduced at the Green Drop training workshop.

Mr Singh said that the only real benefit to the municipality (in the context of the audit alone) would be in being able to meet the existing Green Drop criteria (e.g. capacity estimation) better and with more confidence. However, the concomitant benefits for the municipality would be better understanding of the plants and their operations.

Mr Sampson noted that the benefits of modelling would mostly be on the side of the municipality, the regulator might only gain a little in confidence because a model had been done. However, it was possible to manipulate a model to get a desired result. So how is an assessor to know that the modelling has been done properly?

This led to the question of whether an agency to validate models needed to be established. Ms Laher pointed out that the information used in an audit had to be verifiable, and the outputs of a simulation could not be used if they were open to manipulation.

Prof Ekama felt that it would be a mistake to make modelling simply an administrative exercise, because the main benefit of modelling was that it caused operators to engage more deeply with the operation of their process, which in turn led to understanding how to improve it. The outputs of the steady state model are entirely dependent on the influent wastewater characterisation and the plant configuration. Dr Brouckaert pointed out that the modelling includes the process of deriving the influent characterisation from measured data, and that this step probably needs to be checked by an experienced modeller. It would have to be verified that the monitoring was carried out and that it produced the data that was used in the model. Ms. Laher confirmed that these were aspects that were checked in the audit, on a random sample basis. So the model prediction cannot be used to assess compliance in the audit, but it is a tool which can be used to manage the system to achieve compliance.

The question was raised about how modelling would affect smaller municipalities participating in the Green Drop programme. Would allowance be made for lack of advanced skills? Mr. Sampson mentioned that Green Drop gave recognition to partnerships where larger municipalities assisted smaller ones, and that modelling might be a way of providing such assistance.

The question of how to develop the required skills was then discussed. The options seem to be training offered by universities, by the Green Drop programme itself, university courses, or by a professional organisation, such as a proposed WISA specialist division. Mr Singh explained steps being taken to set up a modelling group in WISA to build capacity in South Africa, (for all purposes, not just Green Drop.) Prof Ekama argued that the professional option would be most effective in the long run, as professional practitioners would bring the best combination of theory and practical experience.

On the question of financing the development of capacity, the big municipalities such as Cape Town, ERWAT and eThekwini would allocate engineers to modelling. However smaller municipalities have much more urgent issues of capacity development. Mr Tshangela pointed out that the Green Drop programme allocates points for training, and one possibility would be to specify modelling as one of the components of this training. So he recommended that we should engage with DWS to introduce modelling training as one of the Green Drop criteria.

No clear recommendation came out of a discussion on how modelling in a Green Drop audit would be financed. Where the modelling was undertaken by the municipality, personnel cost would probably be carried by the municipality itself. Seed money for starting a WISA specialist group is available; WISA has model for the sustained financing of established groups. The question of financing an agency to validate models for audit purposes was not discussed.

After the formal meeting ended, most of the delegates stayed for a more detailed presentation by William Wu on the steady state model.

C.7 Questionnaire

A questionnaire was distributed to the participants to give structure to the discussion, and to capture opinions of the workshop participants.

Total number of responses: 20

Respondents could select more than one answer for each question so totals will add up to > 100%.

Question	Options	Responses	Comments
Should modelling become part of Green Drop audits?	Immediately	3 (15%)	Just for capacity assessment (1) Tools should immediately be provided to those municipalities which have the ability to utilize them (1)
	After trials	10 (50%)	Additional trials required to determine if modelling is practical, in particular considering size and location of plants 1) Increased use of the model will generate feedback which can be used to improve it (1) Modelling would be limited to risk assessment. It would not reflect true compliance (1) More municipalities need to use modelling before (requiring results to be ?) submitted to DWS (1)
	After further research	3 (15%)	Additional research required to simplify modelling for rural plants (1)
	No	5 (25%)	Contribute to bonus points in Training and Risk categories (1) Modelling would be useful to Water Service Authorities and Providers (with sufficient capacity) as a tool for optimisation and upgrades. It cannot be part of an assessment tool at Green Drop (4)
	Undecided*	2 (10%)	It could be used as a tool to get bonus points. It offers a few benefit in achieving the Green Drop (1) Lack of capacity, skills and data sets. Audit information must be verified and credible. (1)
The comments tended to be more revealing than the answers to the questions. Without comments it was not clear what applications people did or did not see for the modelling within the Green Drop context. Only one respondent specifically stated that capacity estimation could immediately be used in the Green Drop process. One respondent thought modelling should only be used in risk assessment and two thought it could be used to earn bonus points but should not be part of the compulsory requirements. The majority of respondents (11 or 55%) felt that (field) trials and/or more research was required before modelling could be formally included in the Green Drop requirements. The need to simplify the modelling process for rural plants was specifically mentioned. Respondents generally seemed to feel the modelling tools would be useful to municipalities that had the capacity to use them whether or not they believed modelling should be part of Green Drop. A general concern among all respondents were lack of capacity in many municipalities. One respondent raised concerns about the verifiability of the results.			

* Indicates a category added during analysis of the results

Question	**Options**	**Responses**	
Are the proposed approach and tools adequate?	Fractionator		Comments
	Yes	14 (70%)	More than adequate, perhaps too complex for Green Drop (1) Ready for now (1) Excellent tools (1) Smaller WSAs have limited data and fractionator would assist with capacity verification (1)
	No	3 (15%)	Not for Green Drop but OK for municipal use (1)
	No response*	3 (15%)	Fractionator complex (to use?) and based on complex mathematical principles (1) – *presumably considered a disadvantage*
	Steady state model		Comments
	Yes	15 (75%)	Same as for yes responses for Fractionator
	No	2 (10%)	More data needs to be generated to include different regions (1) – *assumed to refer to the limited number of pre-loaded raw water profiles available in the SS model.* Not for Green Drop but OK for municipal use (1)
	No response*	3 (15%)	Dynamic model preferable to SS model (1)

The majority of respondents felt that the Fractionator and Steady State Model were adequate and ready to use although one respondent was concerned they might be too complex for the Green Drop process. One respondent specifically mentioned the fractionator as tool for addressing data gaps for smaller WSAs. Issues raised by the respondents who felt the tools were not adequate or were not sure, were the need for more data on raw water fractionation from different regions and the complexity of the fractionator in particular. One respondent felt the dynamic model was a better tool. Another stated that the models were adequate for the municipalities but not for use in Green Drop.

The responses to this question taken together with the answers to the previous question reflect the spectrum of views on the tools presented and their potential role in Green Drop. Twelve out of 15 (80%) respondents who thought that modelling could be used in Green Drop immediately or after trials/further research, also felt that the fractionator and steady state model tools were adequate. Of the 3 remaining respondents who thought that modelling might have a future in Green Drop but that the tools were not ready, one comment was unfortunately illegible, one specifically mentioned the need for more data from different regions and wanted more research, and the last wanted more trials by the municipalities (during which the adequacy of the tools would presumably be assessed).

Of the 7 respondents who did not see a compulsory role for modelling in Green Drop or who were undecided on question one, only one appeared to doubt the general usefulness of the tools presented. The other 6 felt the tools should be given to the municipalities who had the capacity to use them with two specifically mentioning the possibility of using them to earn bonus points. One of the six thought that the steady state model would be useful to the municipalities but was apparently unconvinced about the fractionator.

Question	Options	Responses	Comments
Who should do the modelling?	Municipality (or Water Board)	18 (90%)	Both municipalities and consultants should undertake the modelling – they bring different skills sets (1) Municipalities with sufficient capacity should undertake modelling otherwise consultants should be used (6) Consultants would provide training and support (3)
	Consultants	13 (65%)	
	DWS*	2 (10%)	
	Universities*	1 (5%)	
Discussion The largest majority (18 or 90%) of participants saw modelling as being undertaken by the municipalities but a smaller majority (13 or 65%) also saw a role for consultants. Seven of the respondents made comments to the effect that modelling should be undertaken by those municipalities who were able to do so with consultants providing training and support or undertaking the modelling on behalf of the smaller municipalities which did not have the resources to do it themselves. Two respondents suggested DWS should also be involved and one thought that the Universities would undertake some of the work.			
Should models be independently validated?	DWS	4 (20%)	Validate capacities (1)
	Consultant	5 (25%)	Independent, qualified consultants (2)
	Yes, not specified*	2 (10%)	Necessary but difficult (1)
	Group of experts or Agency*	3 (15%)	Professionals and universities involved (1) Set up by WISA or the regulator (1) Need to validate modellers' inputs (1)
	UKZN/UCT*	2 (10%)	
	No*	2 (10%)	Because did not think it should be formally part of Green Drop (1) Difficult and impractical. Unnecessary especially if only for bonus points (1)
	No response*	3 (15%)	
Fifteen (75%) of respondents thought that there should be some type of model validation, 2 (10%) said there should not and 3 (15%) provided no response on this question. Based on the group discussion and two written comments in the surveys, "model validation" probably meant validation of the model inputs to most participants, although one respondent indicated that capacity estimation results needed to be validated. Some participants appeared to be unsure how model validation would be carried out with 2 respondents (10%) not specifying who should undertake it, another respondent commenting that it would be impractical another that it would be difficult to do. With respect to who should carry out the modelling, the original options were a) DWS, b) consultants and c) alternate suggestion. Four (20%) specified DWS and 5 (25%) consultants. Alternate suggestions were UKZN/UCT (2 or 10%) or a specialist group/agency/committee comprising experts including professionals and the universities involved (3 or 15%).			

How should modelling capacity be developed?	Courses	12 (60%)	WRC funded courses presented in all provinces by UCT/UKZN (1) Continue with courses and training already being undertaken by Universities and include them in the WISA specialist group (1) Interactions to include hands-on use of models and data input (1)
	WISA specialist group	13 (65%)	
	DWS training*	2 (10%)	Model verification might be included in GDS assessor accreditation (1) Online training on GDS site (1)
	University seminars*	1 (5%)	
	No response*	1 (5%)	
The options considered were courses and training offered through the Universities, a WISA Specialist group and training offered through the Green Drop programme. The most popular option (13 or 65%) was the establishment of a WISA Specialist group although a majority (12 or 60%) also saw a continuing role for the Universities which are already offering courses and training to both their own students (future practitioners) and some municipalities. Only two respondents suggested training through DWS but they had some interesting suggestions, namely that model verification could be part of the GD assessor accreditation and that online training could be offered through the GDS site.			
How should tools be maintained and developed?	UKZN / UCT	15 (75%)	Tools development by UKZN/UCT. Other organizations which use the tools may develop their own interfaces. (1) With funding by the WRC and Metro Municipalities (1) Initially maintained by Universities. Should have download site with latest pristine copies. (1) UKZN/UCT should lead initiative and be custodians of software and knowledge (1)
	WISA specialist group	13 (65%)	GDS specialist should be appointed (1)
	Consultant	5 (25%)	All stakeholders combined (1)
	DWS*	2 (10%)	Custodians (1) Tools will eventually move to GDS site. Should have the latest pristine copy available for download (1)
	No response*	1 (5%)	
The biggest group of respondents (15 or 75%) thought that the UKZN/UCT should continue to take the lead in the development and maintenance of the modelling tools however two respondents saw DWS becoming the eventual custodian with tools being downloadable from the Green Drop site. A majority of respondents (13 or 65%) also saw a WISA specialist (modelling) group being involved with one respondent suggesting that GDS specialist be specifically appointment. Five (25%) respondents also saw a role for consultants with one respondent commenting that all stakeholders need to be involved.			

How can modelling be financed?	DWS*	3 (15%)	
	WRC*	5 (25%)	Workshops and projects (1)
	WISA*	1 (5%)	Self-financed through events (1)
	Municipalities*	9 (45%)	Finance specific events (2) Municipalities fund themselves (1) Personnel and time (1)
	Universities and private organizations*	1 (5%)	R & D budgets (1)
	No response*	8 (40%)	
This question had the highest rate of no response (8 or 40%) but the biggest group of responses was that modelling should be funded by the municipalities. However, it was usually not clear what the respondents meant by this. There are two aspects to the funding question – one is who would finance the development of tools and field trials to assess them as well as capacity development and training, etc., while the other is who would bear the cost of the modelling work as a part of regular Green Drop audits. Without the inclusion of relevant comments, it was generally not clear which aspect of funding the respondent is referring to. In the municipalities group, two respondents were clearly referring to capacity building (events) while only one referred to undertaking the modelling (provide personnel and time). Five (25%) respondents thought funding should come with WRC and this would presumably for capacity development and further research into the development, use and usefulness of modelling. Similarly funding from WISA, Universities and other private organizations would be for courses and events but it is not clear what DWS funding would be for.			

C.8 Overall Conclusions

The clearest results from the surveys was that 70-75% of participants thought that the modelling tools presented would be useful to those municipalities which are able to use them and 80% felt that modelling was ultimately the responsibility of the municipalities. However, there was less certainty about whether and how modelling should be incorporated into Green Drop. Several participants noted that many municipalities will not have the capacity to undertake modelling and will have to outsource it while even the larger municipalities will require assistance and support.

Only one participant identified the capacity estimation tool as being immediately ready for Green Drop which is the application which the project team believed was most ready for application. Several participants thought that modelling should only be used to earn bonus points and some felt that the risk assessment was the most promising application. More than half felt that more trials/research was required. Concerns raised included lack of capacity in the municipalities and lack of verifiability of the results.

Seventy five percent of participants felt there should be some type of model verification but there was no consensus on how this would work. The majority of participants assumed an ongoing role for the Universities in capacity development and training as well as modelling and tools development but the majority also favoured the establishment of a WISA Specialist professional group. Funding for these efforts was assumed to come from WRC and the major municipalities while the WISA group was assumed to be self-funded. Participants seemed less clear how funding for the actual inclusion of modelling in Green Drop would work.

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